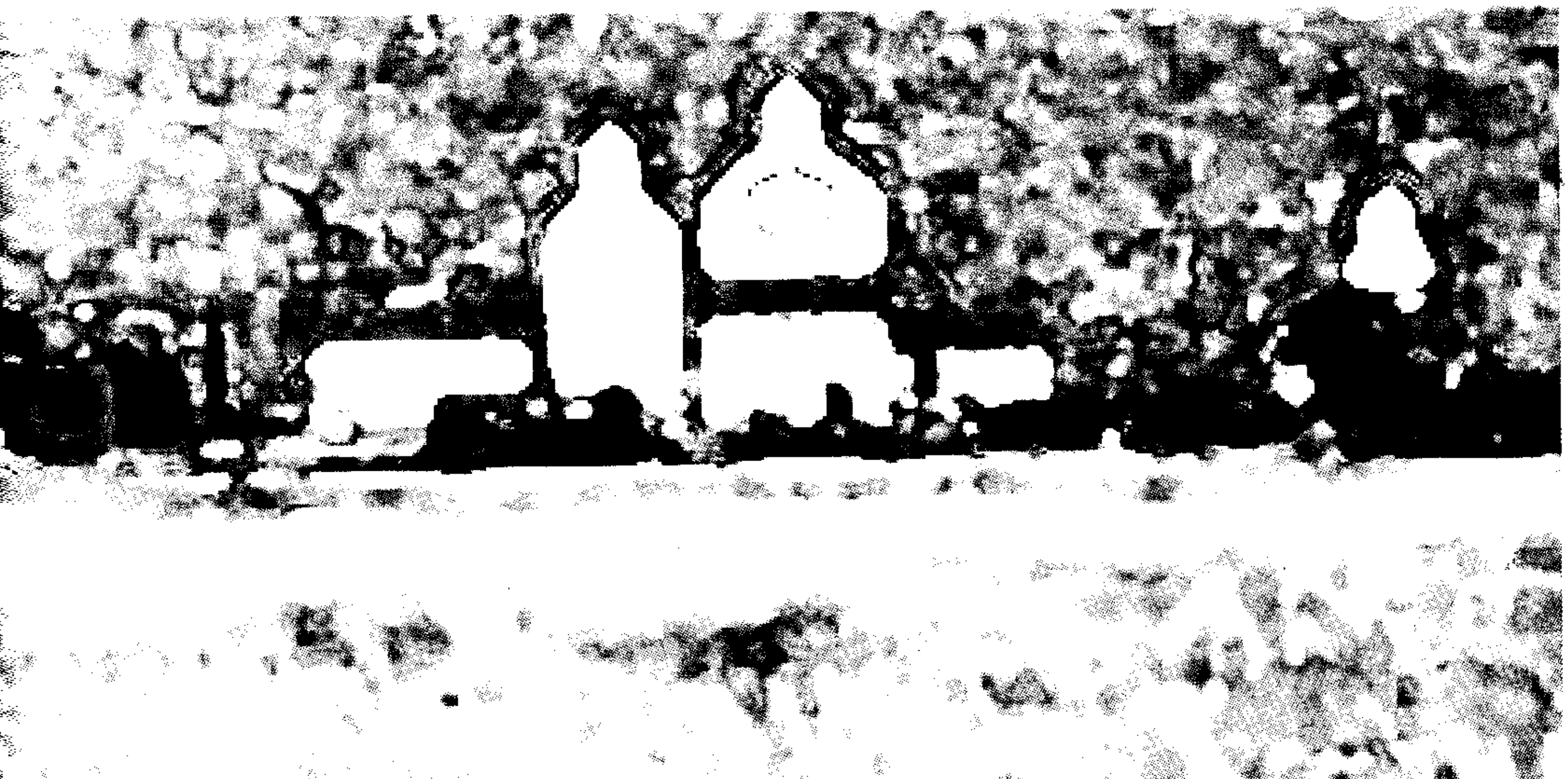
Extension Report 62

Diffuse Sources of Nitrogen Related to Water Quality Protection in the Northern Great Plains



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Introduction

Nitrogen (N) is an element commonly found in many different chemical forms. It is an essential component of amino acids, which are the fundamental chemical compounds from which all life is composed. The nitrate (NO_3) form is the predominant source of N extracted by plant roots from soils. Therefore, it is beneficial that an adequate supply of NO_3 exists in the soil profile during plant growth. As with many chemicals, some of the forms of N can be toxic to certain life forms and may override the beneficial effects.

Excessive amounts of NO_3 ingested by humans and animals contribute to methemoglobinemia, a disorder that reduces the oxygen transported by red blood cells. Studies have tenuously linked nitrate in the diet with other human disorders such as gastric cancer. Crop production problems such as lodging of small grains and reduced quality (e.g. barley, sugar beets, potatoes) are also associated with excessive levels of NO_3. The presence of NO_3 in the environment may have both positive and negative consequences in terms of plant growth. Bumper yields of many crops are associated with high levels of soil NO_3 that may lead to excessive aquatic plant growth if translocated to streams or lakes.

Management of the environment in ways that favor the positive benefits of N is obviously a worthy goal. However, the complexity of natural systems challenges our ability to accomplish that goal. Much has been learned regarding the role of N in cropping systems and the environment. Some sources of N contamination can be directly identified (point source), such as effluent from a municipal sewage system. However, most N sources of contamination are less easily identified and occur as small diffuse areas (nonpoint source), such as private septic systems. Regional and national inventories of water resources indicate that most nitrogen contamination occurs from diffuse sources, and the majority of these are related to agricultural activity.

Many studies have shown that diffuse sources of pollution are most directly influenced by interrelationships between local environmental and anthropogenic factors. These studies have also shown that the factors are often subject to substantial variation within small areas. Most practitioners of nonpoint source pollution control agree that effective management practices must be tailored to local conditions. In the case of N, it is necessary to address diffuse sources as they relate to the complexities of the N cycle and soil/crop management.

Identification of diffuse sources of N and their control is addressed within this document by relating existing knowledge of processes that govern N in the environment. Based on this knowledge, a systematic methodology for assessment of potential for contamination of both groundwater and surface water is presented. Recommendations for control of diffuse sources of N should be related to the potential for contamination (i. e., assessment results). The recommendations within this document are based on published results from scientific studies that address N in the environment.

CHAPTER 1

Diffuse Sources of Nitrogen

Status of Nitrogen in Water Resources

Groundwater

In a nationwide study of water wells used for drinking, NO_3 was present both at detectable levels and levels that exceed health standards much more often than pesticides (EPA Staff, 1990). It was estimated from this study that nationwide, 1.2% and 2.4% of the community water system wells and rural domestic wells, respectively, exceed the 10 parts per million (ppm) nitrate-nitrogen (NO_3-N) EPA drinking water standard. Results of water well studies in the prairie provinces of Canada indicate that in the early 1940s NO_3-N levels greater than 10 ppm occurred in nearly 20% of the wells, and these frequencies have not changed (Harker et al., 1997).

In North Dakota several data sets exist related to groundwater monitoring studies. Roberts (1990) reported on three databases that the North Dakota Department of Health maintains. The data show that 11%, 11.5%, and 9% of well water samples from the Groundwater Archive, Microbiological Testing, and Dairy Farm databases, respectively, exceeded 10 ppm NO_3-N. The Madison and Brunett (1985) analysis of 25 years of USGS groundwater data showed that about 5% of the 7387 samples recorded for North Dakota aquifers exceeded 10 ppm NO_3-N. Domayahn (1995) reported that NO_3-N exceeded 10 ppm in 14% of the water samples taken from 683 wells selected from a statistical grid covering North Dakota. Radig and Bartelson (1993; 1995a; 1995b; 1996) and Bartelson and Gunnerson (1998) reported that NO_3-N exceeded 10 ppm in 6% of their well water samples in 1992, 2% in 1993, 3% in 1994, 5% in 1995, and 7% in 1996.

Surface water

A recent study of 20 U. S. watersheds revealed relatively low concentrations of N in surface water, particularly NO_3 (Mueller et al., 1995). NO_3-N concentrations from streams in the Red River Valley watershed had a median value of about 0.5 ppm. All values were less than 10 ppm, but some were high enough to contribute to eutrophication. Compared to streams of the other 19 watersheds studied in the U.S., both ammonia (NH_3) and NO_3 concentrations in the streams of the Red River watershed were lower than average.

An analysis of water quality data from streams in the Red River basin over the period 1970 to 1990 by Tornes and Brigham (1994) reported that the highest median concentrations of NO_3 occurred in the Pembina, Red, and Sheyenne Rivers. NO_3-N concentrations in the Red River over this period averaged about 0.4 ppm. The median NH_3 concentrations in streams of the Red River basin were generally less than 0.2 ppm. The sum of organic-N and inorganic-N (total-N) ranged from below detection (0.1 ppm) to 20 ppm. Most of the N detected in the streams of the Red River was present as a component of organic chemicals dissolved in the water.

NO_3-N plus nitrite nitrogen (NO_2-N) concentrations measured during a two-year period on the 8 major rivers in North Dakota (Cannonball, Heart, James, Knife, Missouri, Red, Sheyenne, and Souris Rivers) were less than 1 ppm (Berkas, 1993). The interim standard for class I streams in North Dakota is 1 ppm for NO3-N plus NO2-N (NDDH Staff, 1991). No trends occurred in NO_3 concentrations during the monitoring period.

The importance of N to aquatic organism growth associated with eutrophication is influenced by phosphorus (P). It is generally accepted that in eutrophic lakes a N to P ratio that is less than 10:1 indicates a "N-limited" system (Forsberg, 1980; Thomann and Mueller, 1987). In other words, N availability controls biomass growth when it is the limiting nutrient, and small additions of the limiting nutrient to the aquatic system will result in increased production. Many lakes in the eastern U.S. with symptoms of eutrophication have been determined to be P-limited. However, in the northern prairies where minerals containing P remain relatively abundant in soil parent material compared to more weathered materials in the eastern U.S., the potential for N-limited conditions is greater. N limitation in northern prairie lakes has been observed by investigators in South Dakota (German et al., 1991) and North Dakota (Shubert, 1980). Despite the observation of N-limited conditions in Devils Lake, total-N concentrations have been reported to be equivalent to hypereutrophic systems (Sando and Lent, 1995).

N levels in northern prairie wetlands, lakes, and tributaries have been observed to vary seasonally (Lent, 1994; Sando and Lent, 1995; Lent and Zainhofsky, 1995), but long-term trends have not been detected. German et al. (1991) observed that fluctuations in the ratio between N and P were great enough to change from N-limited to P-limited conditions among seasons. Generally the highest concentrations of NO_2-N plus NO_3-N are found during spring runoff but are lowered significantly by biological activity as temperatures increase later in the spring and summer. Total N concentrations in northern prairie lakes are lowest in the fall, increase in the winter, remain the same or decrease in the spring, and increase in the summer (Sando and Lent, 1995). During periods of high total-N concentration, organic N (aquatic organisms) is the predominant form in the summer and NH_3 (decomposing organic material) in the winter.

Natural Sources of Nitrogen in the Environment

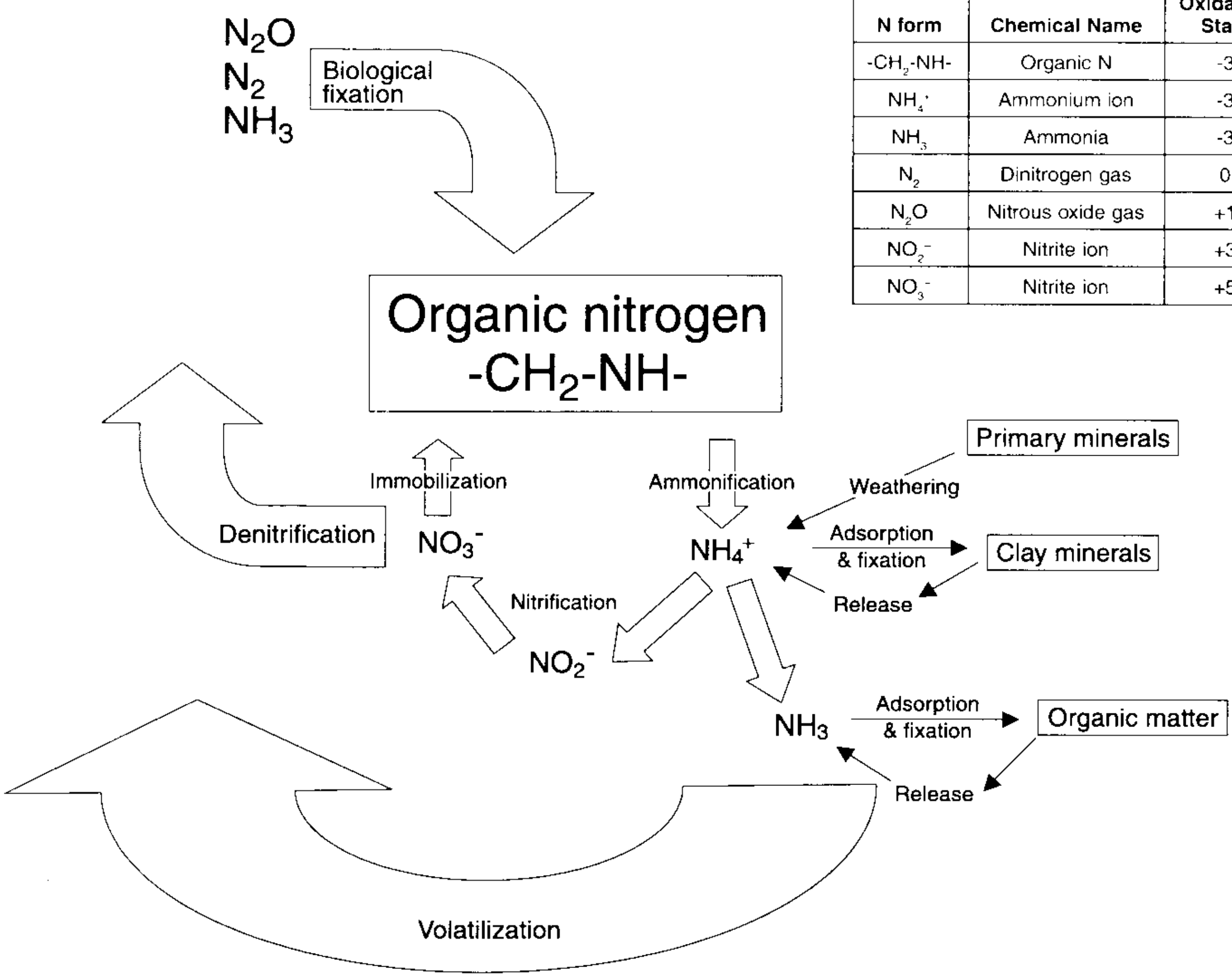
N is distributed among the earth's atmosphere, biosphere, lithosphere, and hydrosphere (Stevenson, 1965; 1982). The processes of exchange among the pools of nitrogen are commonly referred to as the nitrogen cycle (Fig. 1). The natural transformation of N constitutes change from an oxidation state of -3 to +5. Ninety-eight percent of the total-N is present in the lithosphere, which includes geologic materials and soils. N contained in primary (igneous) rock of the earth's crust and mantle is estimated to be approximately 50 times the amount present in the atmosphere. Despite the relatively small portion of total N held in the atmosphere, approximately 35,000 tons of N occurs in the atmosphere above every acre of land. In comparison, an average prairie topsoil contains about 3 tons of N per acre. The most active zone of N transformation is the biosphere, where the majority of N is present as a component of myriad organic compounds. However, the amount of N flowing through biospheric cycles and interacting with the atmosphere accounts for only 2% of the earth's total-N (Hauck and Tanji, 1982).

Mineralization and immobilization

Oxidation of the organic form of N releases inorganic forms of N and is referred to as mineralization. The reverse of mineralization occurs when inorganic N is incorporated into biologic tissue in a reduced oxidation state and is referred to as immobilization. Both of these processes involve the exchange of eight electrons that results in gaseous, mineral, and organic compounds composed of N with different oxidation states (Fig. 1).

Mineralization and immobilization occur simultaneously and provide continuous movement between the organic and inorganic pools of N (Jansson and Persson, 1982).

As carbon (C) and N cycle through these processes, they are gradually incorporated into organic material that is quite resistant to degradation (Bremner, 1965) and is considered the humus component of the soil. Mineralization of humus is quite slow compared to fresh organic residues but is a continual source of NH_3 and NO_3.



N form	Chemical Name	Oxidation State
$-CH_2-NH-$	Organic N	-3
NH_4^+	Ammonium ion	-3
NH_3	Ammonia	-3
N_2	Dinitrogen gas	0
N_2O	Nitrous oxide gas	+1
NO_2^-	Nitrite ion	+3
NO_3^-	Nitrite ion	+5

Figure 1. A general representation of the nitrogen cycle and oxidation state of nitrogen.

The ratio between C and N (C/N) in decomposing residues influences the net balance between these two processes (Bartholmew, 1965; Jansson and Persson, 1982). It has been generally assumed that when decomposing organic matter has a high C/N ratio, microorganisms absorb all N mineralized and little inorganic N accumulates. Decomposing materials require a C/N ratio less than 20 to 25 before appreciable amounts of inorganic N are expected to accumulate (Harmsen and Kolenbrander, 1965). The C/N ratio can be misleading, because it assumes that all C and N in the decomposing material is equally available or degradable. Organic materials in soils are heterogeneous; consequently, each organic material has a different resistance to degradation (Bartholomew, 1965; Jansson and Persson, 1982). Smith and Peterson (1982) state that many studies show that crop residue degradation is often not limited by N, and the practice of adding N to increase the rate of residue degradation may not be effective under many circumstances.

Ammonification and nitrification

Mineralization of organic N includes two processes, ammonification and nitrification (Stevenson, 1982). Ammonification (Fig. 1) describes a group of enzymatic reactions utilized by heterotrophic organisms to extract energy from organic materials (Ladd and Jackson, 1982). These reactions also release the ammonium ion (NH_4^+) into the environment. Enzymes break down complex N compounds such as proteins or relatively simple compounds such as amino acids. The enzymes involved may be hydrolases, oxidases, deaminases, and lyases and may originate from plant, animal, or microbial sources. The product of ammonification, NH_4^+, is oxidized through a series of reactions to nitrate (NO_3^-), also known as nitrification (Fig. 1).

Although nitrification is not the only process that contributes NO_3 to the soil, it is the predominant process (Schmidt, 1982). The organisms responsible for nitrification are autotrophic bacteria that utilize the energy released when NH_4^+ and NO_2^- are oxidized. The genus *Nitrosomonas* oxidizes NH_4^+ to NO_2^- and the genus *Nitrobacter* oxidizes NO_2^- to NO_3^-. Plants or microorganisms will use NO_3 to synthesize organic material as it becomes available and the cycle begins again.

Factors affecting mineralization and immobilization

Because many types of reactions and organisms are involved with mineralization and immobilization, changes in environmental conditions have varying impacts on the forms and quantity of soil N at any given time. Low temperatures or dry conditions usually will slow the rate of immobilization but ultimately have little effect on the magnitude of net immobilization after extensive decomposition has occurred (Bartholomew, 1965). Moist conditions contribute to faster rates of mineralization and immobilization of N; therefore, the release and uptake of N by microbes from plant residue will be much quicker if residue is incorporated into the soil compared to being left on the drier surface. Immobilization of N in soil organic matter or humus under anaerobic conditions results in lower net immobilization of N compared to aerobic conditions. This explains why peat has a relatively low N content. High soil pH and low cation exchange capacity (CEC) will inhibit nitrification, because these conditions are favorable to the formation of NH_3, which is toxic to nitrifying bacteria (Schmidt, 1982). Of the nitrifiers, NH_4 oxidizers are least affected by NH_3, which contributes to NO_2 accumulation in some calcareous soils (Stevenson, 1982).

Fixation and adsorption

Biological fixation

The accumulation of N in soil bears a close relationship with organic matter. A C to N ratio of 10 to 12 for organic matter of the surface soil is considered typical for temperate regions. The major source of N that accumulates in soils is biological fixation (Fig. 1) of elemental-N (N_2) from the atmosphere (Stevenson, 1965). Organisms capable of fixing N_2 are thought to have developed early in Earth's history, probably at the time that carbon dioxide (CO_2) reduction capabilities were developed. Blue-green algae is one of the survivors of these early organisms that persist in our present day environment (Stevenson, 1965). These autotrophic organisms are often one of the first to colonize barren areas and begin the process of soil development.

The symbiotic relationship between certain N fixing bacteria and plants is thought to have developed over time through a series of stages that began with a casual association of free-living bacteria and plant root surfaces (Stevenson, 1965). The sophisticated relationship between plants of the family Leguminosae (legumes) and bacteria of the genus *Rhizobia* is the culmination of millions of years of evolution. By invading root hairs of legumes, the *Rhizobia* bacteria are provided with nutrition from the plant in exchange for supplying NH_4 to the plant by reducing N_2 from the atmosphere (Nutman, 1965). Legumes have the potential to fix as much as 500 lbs/ac/yr of N (Vincent, 1965). Although the legume/*Rhizobia* symbiotic relationship is the predominant biological fixation process, other plants, such as angiosperms, also participate in this type of a symbiosis. The importance of nonlegume N fixing plants has been noted as soil formation and plant establishment progress on new glacial moraines in Alaska (Stevenson, 1965).

Nonsymbiotic organisms such as heterotrophic bacteria from the genuses *Azotobacter* and *Clostridia* and autotrophic blue-green algae also fix atmospheric N_2 (Jensen, 1965). *Azotobacter* are aerobic organisms while *Clostridia* are anaerobic. Both are commonly found in most soils. The contribution of nonsymbiotic N fixing bacteria to soil N may be significant in certain environments such as semi-deciduous forests (Allison, 1965). Blue-green algae prefer wet soil and aquatic environments. Their contribution of fixed-N can be quite important (Jensen, 1965).

Ammonium adsorption and fixation

Adsorption of NH_4 occurs on clay minerals and organic matter (Fig. 1). The NH_4^+ ion will compete with other cations for negative exchange sites on clay minerals and organic matter. Exchangeable NH_4 will participate in exchange reactions, which are electrostatic in nature and dependent on ionic charge, radius, and hydration (Mortland and Wolcott, 1965; Nommik and Vahtras, 1982). Electrostatically sorbed NH_4 is available to microbes and plants but is essentially immobile with respect to water movement. The lack of exchange capacity in coarse textured soils decreases the potential for NH_4 sorption, which increases the potential for NH_3 volatilization.

Silicate clay minerals also sorb NH_4 ions on interlayer positions, rendering some of these ions unavailable to normal exchange reactions (Nommik, 1965; Nommik and Vahtras, 1982). This type of sorbed NH_4 is defined as "fixed-NH_4" because it is not readily available to microorganisms or plants. Several studies have noted a high correlation between potassium (K) and fixed-NH_4 (Nommik and Vahtras, 1982). Research shows that clay minerals fix only certain cations and they may influence the fixation of each other. The fixation of NH_4 and K seem to be related to the similarity in size of the cation and the opening in the clay hexagon. Because of the "snug fit" with NH_4^+ and K^+ ions, the clay layers are allowed to closely approach each other and are tightly bound, thus trapping the NH_4 and K (Nommik, 1965). If K is introduced to the clay mineral before the NH_4, the amount of NH_4 fixed by the clay will be reduced or vice versa.

The amount of NH_4 fixed on clay minerals will vary according to the type of clay mineral and the conditions of fixation. Montmorillonite will allow the NH_4^+ ion into the interlayer space but does not fix it under moist conditions (Mortland and Wolcott, 1965). Both drying and freezing result in removal of interlayer water, which will produce fixation of NH_4 in montmorillonite. The availability of fixed-NH_4 has been reported the lowest in vermiculite and highest in montmorillonite (Nommik, 1965). The content of fixed-NH_4 in clay minerals from native materials has been reported to be illite > montmorillonite > kaolinite. Native shales with high amounts of illite, vermiculite, or mica have been reported with very high amounts of N as fixed-NH_4. Based on fixed-NH_4 data from various native materials, it is estimated that the amount in the Earth's crust exceeds the amount of elemental-N in the atmosphere (Nommik, 1965).

Young and Aldag (1982) report that fixed-NH_4 in soils range from 0 to >1,000 ppm, with the lowest values for sandy surface soils and the highest for clayey subsurface soils. In some soils the amount of N fixed as NH_4 exceeds the amount of readily available N. Soils in South Dakota formed from Pierre shale have inherited high fixed-NH_4 from the parent material. In the central U.S., 4 to 8% of the N in surface soils has been estimated to be fixed-NH_4 (Nommik, 1965). Usually the relative amount of fixed-NH_4 was found to increase with soil depth, which accounts for the observation of decreased C/N ratios with depth. Significant errors in the determination of organic N are likely if large amounts of fixed-NH_4 are present and total-N is calculated from total-N determinations using the standard Kjeldahl method (Bremner, 1965). This is because the Kjeldahl method measures the NH_3 released by acid digestion of organic matter but cannot discern the difference between other sources of NH_3.

Oxidation of fixed-NH_4 to other forms of N may have important environmental consequences. Young and Aldag (1982) indicated that under some circumstances the source for nitrous oxide in soil and water might be oxidation of fixed-NH_4 rather than denitrification of NO_3. Hendry et al. (1984) determined that elevated NO_3 concentrations in groundwater from an area in Alberta were due to the release of NO_3 as the fixed-NH_4 present in glacial till was oxidized.

Ammonia fixation

NH_3 is adsorbed by soil organic matter (Fig. 1) and is considered fixed, because it is not readily available (Nommik, 1965; Nommik and Vahtras, 1982). Mortland and Wolcott (1982) suggest that NH_3 fixation by organic matter principally involves entrapment during the course of oxidative condensation reactions. In the alkaline pH range, these reactions are autocatalytic and rapid. Studies have shown that NH_3 fixation in dry organic soils is positively correlated to C content (Nommik and Vahtras, 1982). Fixation of NH_3 by organic matter is increased by simultaneous oxidation, whereas preoxidation suppresses fixation. Research has shown that less NH_3 fixation occurs in well-oxidized muck soils (Nommik, 1965).

Volatilization and gaseous loss

Denitrification of nitrate

When oxygen is depleted in the soil, facultative anaerobic bacteria use NO_3 as an alternate electron acceptor for their respiratory requirements. It is generally agreed that the reduction of NO_3 during denitrification (Fig. 1) follows the sequence nitrate (NO_3^-), nitrite (NO_2^-), nitrous oxide (N_2O), elemental-N (N_2), in that order (Broadbent and Clark, 1965). Under natural conditions, N_2O often volatilizes to the atmosphere before reduction to N_2. Through volatilization of N_2O and N_2, the denitrification process removes N from soils (Broadbent and Clark, 1965; Firestone, 1982), particularly in wetlands (Buresh and Patrick, 1981; Patrick, 1982; Jorgensen, 1989) and riparian areas (Hanson et al., 1994; Groffman et al., 1996; Bragan et al., 1997a, b; Gold et al., 1998; Horwath et al., 1998; Jacinthe et al., 1998). Evidence of denitrification in groundwater aquifers has also been demonstrated (Postma et al., 1991; Knighton and Albus, 1992; Mayer et al., 1992; Gerla et al., 1994; Goebel et al., 1994; Patch and Padmanabhan, 1994; Bohlke and Denver, 1995; Bates and Spalding, 1998).

Denitrification also depends on the presence of an oxidizable substrate to furnish the energy required by the denitrifying bacteria (Broadbent and Clark, 1965). Organic C usually serves as the electron donor in the denitrification process and in most cases is the limiting factor (Firestone, 1982; Groffman et al., 1996). Several investigators have found that denitrification often occurs in isolated pockets of intense activity that are related to accumulations of decomposable organic material (Firestone, 1982; Gold et al., 1998; Jacinthe et al., 1998). It is not uncommon for denitrification and nitrification to occur simultaneously in a soil profile with extreme spatial variation in organic material and oxygen content (Patrick, 1982; Jacinthe et al., 1998). In shallow aquifers, the energy source for denitrification appears to be organic C (Groffman et al., 1996; Mayer et al., 1992; Bragan et al., 1997b; Bates and Spalding, 1998). However, evidence from some studies indicates that when denitrification occurs in deep aquifers, iron sulfides may serve as the energy source (Postma et al., 1991; Korom, 1992; Bohlke and Denver, 1995) More specific to North Dakota, Korom et al. (1996) have demonstrated that pyrite associated with shale fragments in the Elk Valley aquifer serve as a probable source of energy for denitrification.

Ammonia volatilization

In addition to denitrification, volatilization of NH_3 is another source of gaseous loss of N from the biosphere (Fig.1). The process of ammonification was discussed previously as a component of organic matter mineralization. Another pathway of ammonification may occur under highly reduced conditions or low oxidation potentials as observed in some wetland soils. Under these conditions, NO_3 has been observed to be reduced to NH_4 rather than taking the denitrification pathway (Buresh and Patrick, 1981; Jorgensen, 1989). In soil and water, equilibrium exists between the NH_4^+ ion and NH_3, with NH_3 being the more volatile form. NH_3 losses are favored in soils with high temperatures, high pH and low cation exchange capacities (Allison, 1965).

Some volatilization of NH_3 is expected when organic materials with N contents higher than 2% decompose, particularly if these materials decompose on the soil surface (Allison, 1965). Worldwide volatilization of NH_3 from animal waste decomposition has been estimated as 2-6 teragrams per year (Tg/yr) [Tg = 10^{12} grams] from wild animals and 20-35 Tg/yr from domestic animals (Stevenson, 1982). The significance of NH_3 volatilization from animal waste is put into perspective when compared to 4-12 Tg/yr from combustion of fossil fuel.

Transport of diffuse nitrogen by water

The process of N transport by water is integral to water resource contamination and protection. Commonly, the influence of groundwater and surface water hydrodynamics on N contamination receives most of the attention. However, contributions of N to the environment via precipitation can be an important source under some circumstances (Legg and Meisinger, 1982). In many cases research has shown that the amount of N contained in precipitation exceeds the amount of soluble N lost in runoff. Stevenson (1982) suggests that for natural plant communities, N contained in precipitation may be enough to balance leaching and denitrification losses. The N in precipitation occurs as NH_3 and NO_3, the former generally accounting for the greatest portion of N. The amount of N deposited from precipitation varies with the world climatological regions as follows: tropics > humid temperate > semiarid (Brady, 1974). In temperate regions the total amount of N added to the soil from precipitation appears to average about 5 lbs/ac/yr. However, areas near feedlots in the Midwest have been found to receive > 30 lbs/ac/yr of N. Studies indicate that the fraction of N added annually to soils in the corn-belt is equivalent to 3-10% of the amount of N applied as fertilizer (Tabatabai, 1983). Tornes and Brigham (1994) calculated that about 1.3 lb/ac/yr N is added to the soil by atmospheric deposition in the watershed of the Red River.

Groundwater and nitrogen

Of the different forms of N, nitrate (NO_3^-) has the greatest potential to move with water through soils and geological materials because of its high solubility and negative charge. The high mobility of NO_3 and its propensity for leaching through soils has been recognized since the mid -1800s (Harmsen and Kolenbrander, 1965). On the other hand, the NH_4^+ ion is adsorbed and fixed on cation exchange sites, thereby rendering it immobile except in soils with extremely low cation exchange capacities (Nommik, 1965; Nommik and Vahtras, 1982). NO_3 occurrence is closely related to the movement of soil water (Allison, 1965; Young and Aldag, 1982) and its presence is subject to the same extreme variations observed in field soil water content (Legg and Meisinger, 1982). Other than plant uptake, leaching of NO_3 from soils is the most important mechanism of removal.

Movement of water and solutes, such as NO_3, may be mathematically described as a combination of diffusion and convection (Gardner, 1965). NO_3 leaching may be estimated using these equations; however, general assumptions of steady state conditions need to be applied, which rarely occur in field soils. When water and solute flow are estimated over longer periods, such as seasonal wetting and drying cycles, the average distribution of water and solutes in the soil more closely reflects steady state conditions (Gardner, 1965). By disregarding perturbations in water content and solute concentrations caused by intermittent applications of water (irrigation and rain events), steady state may be assumed without impairing predictions of solute leaching (Nielsen et al., 1982). Legg and Meisinger (1982) concur that N budget models based on the assumption of steady state are useful for making long-term (years) N balance predictions.

The high spatial and temporal variability in field soils that affect water and solute movement has been demonstrated rather conclusively (Nielsen et al., 1982). For at least a century, scientists have been aware that cracks, fissures, clay lenses, worm holes, root channels, textural variations and differences in soil solution properties contribute to non-uniform displacement of water and solutes under field conditions. Schuh (1993) found that the spatial arrangement of microtopographic lows (< 1 inch difference in elevation) were a major factor in predicting water and solute movement. Sampling and monitoring field plots to adequately account for variation in soil parameters have challenged scientists. Deterministic methods that do not account for this variability have not been successful in accurately predicting field-scale results (Nielsen et al., 1982). Geostatistical methods, such as kriging, combined with soil mapping information help to improve predictions of water movement and NO_3 leaching over larger areas such as fields or watersheds.

In general NO_3 leaching is expected to be minimal in the Great Plains or other semi-arid areas compared to humid areas, because water and NO_3 seldom move to depths beyond the rooting zone (Allison, 1965; Scarsbrook, 1965; Harmsen and Kolenbrander, 1965; Power, 1970; Stewart et al., 1975). In fact, the low potential for NO_3 leaching in the northern Great Plains is the reason why residual soil NO_3 can be related to crop yields (Swenson et al., 1979; Stanford, 1982). In medium textured soils, NO_3 leaching is not likely to occur during seasons of active plant growth; therefore, the most probable time of leaching is during non-crop periods, such as in late fall and early spring when soils are not frozen (Allison, 1965). A study of 13 different soil types in North Dakota at 16 sites that varied from fine to coarse textures (Swenson et al., 1979) identified significant NO_3 leaching beyond the root zone only in a coarse textured soil during an exceptionally wet year (26 inches of precipitation). At all other sites, most of the NO_3 was found in the rooting zone above the 36-inch depth. Cassel et al. (1971) stated that the greatest potential for NO_3 leaching to groundwater in North Dakota occurs in the spring on coarse textured soils with shallow water tables.

Surface water and nitrogen

Knowledge of the mechanics of surface water contamination is quite extensive compared to groundwater contamination (Daniel et al., 1991). The main reason for the disparity is related to the relative ease of access to study surface water compared to groundwater. However, the complexities of water flow, chemical reactions, and ecological interactions still create tremendous challenges in determining the exact nature and extent of surface water contamination related to diffuse sources. Unlike groundwater, contamination of surface water from diffuse N sources is also directly related to forms of N other than NO_3. Stewart et al. (1975) stated that sediment is the greatest pollutant of surface water. Attached to sediment, among other things, are organic forms of N and NH_4, which account for the largest percentage of the total-N translocated by water (Dean, 1983). Stewart and Woolhiser (1976) state that the erosion hazard and potential impacts to water resources is much greater for less productive soils in the USDA Capability classes of III and IV compared to the more productive soils in classes I and II.

Because of the close association of certain forms of N with sediment carried by runoff water, the dynamics of particle detachment, translocation, and deposition are important in determining the fate of N in water resources. Increased concentration of N and other contaminants in sediments during the transport process (i.e. enrichment) is commonly reported and directly related to surface water flow dynamics (Stewart et al., 1975; Baker and Laflen, 1983; Dean, 1983). The practical implication of the enrichment process is that reductions in erosion do not result in an equal reduction of nutrient loading to water resources (Stewart et al., 1975; Wineman et al., 1979).

Detachment and translocation of sediment is influenced by many factors that have been accounted for in the Universal Soil Loss Equation (Wischmeier and Smith, 1978; Renard et al., 1997). By quantifying certain soil factors, surface cover, and climatic conditions, the long-term average amount of soil eroded by water can be estimated. The same principles may be applied to determine the translocation of nutrients and their potential impact on water resources (Wischmeier, 1976).

The most accurate estimates of eroded sediment cannot be assumed to be a measure of loading to water resources, because only a portion of the sediment actually reaches a stream or lake (Boyce, 1975; Renfro, 1975; Campbell, 1985). For example, it was observed in an Illinois study that most eroded sediment was deposited on low lying hillslope positions and not delivered to water resources (Stall, 1985). In general, a greater percentage of eroded sediment is delivered to streams in smaller watersheds with steep slopes and fine textured soils.

The source of eroded sediment plays an important role in determining impacts on water resources. Translocation of sediment as aggregates of primary particles (sand, silt, clay) plays a critical role in the erosion and sedimentation processes (Foster and Meyer, 1975). Organic matter promotes the formation of soil aggregates and is essential to their stability (Brady, 1974). Meyer (1985) found soil erodibility to be negatively correlated to clay and organic matter content. Organic matter and clay have been determined to be intimately associated with respect to many soil chemical and physical characteristics (Bremner, 1965). Because N is an integral component of organic matter, its environmental fate is strongly influenced by soil aggregate and clay particle behavior.

The relative amount of sediment originating from the surface soil with respect to that originating from subsurface materials will vary and has a definite bearing on N delivery to water resources. Eroded sediment originating from the soil surface is likely to contain higher amounts of adsorbed chemicals compared to sediment eroded from deeper soil layers (Meyer et al., 1975). However, nutrients also generally associate with the clay particle size fraction due to its chemical reactivity and affinity for organic matter. Deeper subsoils and geologic materials with high clay content have been reported with relatively high amounts of adsorbed N (Nommik, 1965; Young and Aldag, 1982; Hendry et al., 1984). Under these circumstances, gully or channel erosion of deeper soil and geologic materials may be an important source for N transport to surface water resources.

Sediment detachment and translocation controlled by the energetic force of rainfall is interill erosion (Foster and Meyer, 1975). In some studies, interill erosion has been found to account for 10 to 20% of the total amount of material eroded (Vandaele and Posen, 1995). When the flow of runoff water begins to concentrate, sediment is detached and translocated by the energy of turbulent shear developed along specific paths known as rills (Foster and Meyer, 1975; Loewenherz-Lawrence, 1994; Nearing and Parker, 1994). Vandaele and Posen (1995) attributed over 30% of the total amount of eroded sediment to rill erosion.

The processes of interill and rill erosion are closely related to soil aggregation (Foster and Meyer, 1975; Meyer, 1985) and are accounted for by the Universal Soil Loss Equation (Wischmeier and Smith, 1978; Renard et al., 1997). Eroded aggregates and associated nutrients behave as coarse particles, if the aggregates remain intact during detachment and transport (Foster and Meyer, 1975). In this case, nutrients will not move very far from where they are eroded. However, if aggregates are substantially degraded to their primary particles, the clay particles and associated nutrients will travel much greater distances with increased potential for water contamination. Streams from watersheds where interill and rill erosion are predominant generally show a positive correlation between sediment concentration and discharge (Flaxman, 1975).

When the rills become significantly deep (> 1ft), they are classed as gullies and cannot be accounted for by the USLE (Wischmeier and Smith, 1978). Gully erosion and other forms of sediment detachment and translocation due to channel flow were responsible for over 50% of the sediment delivered from a watershed in Oregon (Anderson, 1975). Streams within watersheds where gully and channel erosion is predominant usually have an unpredictable relationship between sediment concentrations and discharge (Flaxman, 1975).

It has been suggested that although materials initially translocated and deposited do not immediately impact water resources, their potential for detachment and translocation may be increased during subsequent erosive events (Heede, 1975; Gallart, 1995). Sharpley (1985) concluded that aggregate destruction during translocation caused increased bioavailability of certain nutrients, which could increase eutrophication of receiving streams. However, Meyer (1985) found no additional breakdown during transport of aggregates eroded from fine silt and clay soils. On the northern Great Plains, wind eroded sediments deposited in ditches and depressions were observed to function as source N contamination due to the rapid release of significant quantities of NO_3 (Cihacek et al., 1992; 1993).

The environmental impact of N contained within transported particles(particulate-N) compared to N transported as dissolved NO_3 is not firmly established. Baker et al. (1979) suggested that the influence of particulate-N on water quality would be minimal due to burial by subsequent depositional events. However, Lent (1994) found that through the process of benthic recycling, the predominant source of nutrients in a northern prairie lake (Devils Lake) was bottom sediments.

NO_3 is part of the total-N translocated to streams. Most of the NO_3 is dissolved in runoff and is generally only a small fraction of the total-N translocated (Burwell et al., 1977; Schepers et al., 1980; Gumbs et al., 1985). However, the NO_3 load in runoff is of great importance with respect to water quality because of its bioavailability and mobility. Accurate quantification of NO_3 loading to water resources is problematic due to the many biologic processes that consume or release NO_3 in runoff water (Seigly et al., 1993). NO_3 concentrations in streams often increase as stream discharge increases (Nelson et al., 1979; Mueller et al., 1997). In North Dakota this phenomenon has been observed to occur in some streams during spring melt and runoff (Tornes and Brigham, 1994; Lent and Zainhofsky, 1995; Williams-Sether et al., 1996). In other water resources, such as wetlands, lakes, or streams draining them, the association between NO_3 concentrations and discharge is not as predictable (Lent and Zainhofsky, 1995; Williams-Sether et al., 1996).

Soil patterns related to nitrogen sources and movement

Nielson et al. (1982) stressed that natural variability of soils needs to be adequately characterized before N fate can be predicted with confidence. For example, soils on lower landscape positions in Colorado were found to have significantly greater water content and NO_3 below the root zone compared to soils upslope (Evans et al., 1994). Comparing soils at different landscape positions in Saskatchewan, NO_3 concentrations in the soil immediately below the rooting zone decreased significantly going downslope due to increased leaching (Farell et al., 1996). Creed et al. (1996) demonstrated the need to include soils-landscape characterization in three-dimensional modeling routines to accurately predict N release to a stream from a watershed in Ontario. Soil characteristics have been used with various modeling routines to determine N leaching (Geleta et al., 1994; Jabro et al., 1995). Modeling routines combined with geographical distribution of soil properties have been used to determine areas of groundwater vulnerability to N (Khakural and Robert, 1993; Shaffer and Wylie, 1993).

N losses due to denitrification have been demonstrated to follow a predictable pattern of increase with nearness to wetlands (Patrick, 1982) or riparian areas (Hansen et al., 1994; Groffman, et al., 1996; Bragan et al., 1997a,b; Gold et al., 1998). These studies show denitrification increases to be directly related to soil water regime and the presence of a readily oxidizable C source. These soil characteristics generally change with soil type, thus allowing delineation of areas prone to denitrification losses as opposed to leaching losses.

Soil characteristics that relate to water movement and NO_3 leaching occur in patterns on landscapes that may be determined by referring to a soil survey (Seelig, 1993). Consideration of soil properties is often a component of assessment for groundwater vulnerability (Volk, 1990; Cates and Madison, 1991; Luther, 1992; Seelig, 1994). Some studies have shown that the chemical nature of the NO_3 ion increases the likelihood of movement via preferential pathways compared to other contaminants such as pesticides (Andreini and Steenhuis, 1988; Czapar et al., 1994). It has been suggested that the correlation between preferential flow and soil structure noted in some studies (Quisenberry et al., 1993; Flury et al., 1994) may also be identified by referring to soil survey information.

Knuteson et al. (1989) and Seelig and Richardson (1994) demonstrated that areas of focused recharge may be identified using soil morphologic and chemical characteristics. Areas of recharge are surface locations that contribute water to groundwater (Freeze and Cherry, 1979). Greater potential for groundwater contamination in areas of recharge is expected compared to other areas on the landscape (Seelig, 1994). Low annual precipitation and high potential evapotranspiration allow only limited groundwater recharge from extensive areas on the landscape in the northern Great Plains (Stewart et al., 1975). Groundwater recharge in this region has been described as depression focused (Lissey, 1971; Knuteson et al., 1989). Depressional topographic positions that accumulate runoff water (focused) are most likely to receive enough water to exceed evapotranspiration, thus increasing the potential for water movement beyond the root zone of growing plants. Baer et al. (1993) and Delin and Landon (1993) determined that preferential flow was more likely to occur through soils in depressional or lower landscape positions in Missouri. A similar observation was made for microtopographical (< 1 inch difference in elevation) depressions in North Dakota (Schuh and Klinkebiel, 1994). A chloride tracer study conducted in North Dakota confirmed that substantial flow of water and solutes from the soil profile occurred rapidly in depressional areas during spring melt (Derby and Knighton, 1997).

Anthropogenic Effects on Nitrogen in the Environment

Man's effect on N in water resources needs to be viewed in terms of sources and transformation processes. As previously discussed, processes of N transformation are complex and lead to highly variable sources of N as related to water resource contamination. The impact of man's activities on the N cycle adds just another factor to an already complicated array of interrelated processes. The difficulty in establishing man's contribution to specific cases of N contamination lies in the fact that N contaminants, whether from natural sources or manmade, are chemically the same. Unlike most pesticides and other synthesized chemicals, the mere presence of N in water resources cannot be unequivocally assigned to man's activities. However, in general it can be stated that man's activities have changed the balance of N on the earth (Stevenson, 1982).

It has been suggested that concentrations of NO_3-N greater than 3 ppm in groundwater represent contamination influenced by man's activities. This value was arbitrarily selected for comparison of groundwater at the national level and should not be applied at the local level without critical study (Madison and Brunett, 1985). Natural concentrations of NO_3-N in groundwater substantially greater than 3 ppm are well documented (Power et al., 1974; Hendry et al., 1984).

Upon further analysis of the data from the National Drinking Water Well study, EPA Staff (1992) concluded that both environmental conditions and human activities contribute to contamination of drinking water wells and no single factor alone can be used as a predictor. They suggested that localized site-specific assessments appear to be necessary to obtain adequate evaluations of sensitivity to contamination. Other studies of national scope with respect to groundwater and N contamination have found similar results and conclusions (Madison and Brunett, 1985; Nielsen and Lee, 1987; Fedkiw, 1991).

Fedkiw (1991) and Spalding and Exner (1993) reviewed many groundwater studies that indicated farm well contamination with N was often associated with sources such as barns, barnyards, septic systems, feedlots, silos, buried organic material, fertilizer storage and handling sites. Many studies have shown a positive correlation between N contamination and well characteristics such as age, shallowness, size, porous unsealed casing, and placement in depressions (Madison and Brunett, 1985; Wallrabenstein and Baker, 1992; Barnett and Howard, 1993; Rudolf and Goss, 1993).

As in other U. S. studies, results indicate that the source for NO_3 in contaminated wells in North Dakota is most closely related to the condition of the well or activities around it (Domayahn, 1995; Radig and Bartelson, 1993; 1995a, b; 1996; Bartelson and Gunnerson; 1998). However, studies also show that not all incidences of NO_3 contamination are clearly definable. Consequently, diffuse sources such as leachate from cultivated fields should be considered as plausible alternatives under certain circumstances, even in low vulnerability areas like North Dakota.

Increased nutrient levels in surface water resources can often be related to anthropogenic activities, particularly of an agricultural nature (Artola et al., 1995; Fausey et al., 1995; Dodds et al., 1996; Mattikalli and Richards, 1996; Randall et al., 1997). Jordan et al. (1997) found a positive correlation between percentage cropland of a watershed and the NO_3 concentrations in the watershed streams. It has been estimated that over 90% of the N reaching U.S. streams is from diffuse sources, of which over 80% is related to agricultural activities (Keeney, 1982). Nutrient concentrations in streams studied by the USGS (Mueller et al., 1995) were significantly correlated with landuse. High concentrations of NH_3 often exceeded the chronic exposure level (2.1 ppm) and were associated with urban areas. Generally, nutrient concentrations in streams associated with agricultural landuse were higher compared to streams associated with undeveloped areas.

Considering the variations that occur in surface water conditions and the factors that influence them, Menzel (1983) cautions against assigning eutrophication symptoms exclusively to anthropogenic activities. Evidence presented from writings of naturalists in the 1800s indicates that problems associated with eutrophication were observed at that time. Paleolimnological studies on prairie lake sediments in southern Saskatchewan and Manitoba indicate that eutrophic conditions existed prior to settlement of the region (Allan et al., 1980).

Cropping systems and water contamination from nitrogen

Groundwater

Stewart et al.'s (1975) analysis of groundwater vulnerability to NO_3 contamination from agricultural sources indicates significant regional differences based on soil and climatic patterns. The northern prairies and many western states have the least vulnerability. Similar conclusions were drawn from a study of vulnerability based on agricultural chemical usage and the DRASTIC groundwater vulnerability model (Nielsen and Lee, 1987). It appears that groundwater beneath irrigated, row cropped areas with well drained soils and permeable vadose zones is most impacted in Minnesota, Nebraska, Arizona, California, and Washington (Spalding and Exner, 1993).

The relationship between groundwater contamination with N and cropping systems is far from completely known. Notwithstanding the lack of all the details, some general observations are useful. Evidence shows that undisturbed natural systems or permanently vegetated uncultivated systems usually utilize N and water with greater efficiency than cultivated crops, thus less N is available for vertical movement through the soil profile (Harmsen and Kolenbrander, 1965; Power, 1970; Evans et al., 1994; Izaurralde et al., 1995). Mueller et al. (1995) found the greatest incidences of high NO_3 concentrations in wells surrounded by agricultural landuse compared to urban or undeveloped landuse.

Northern prairie farming systems that include a large component of corn in the crop rotation have greater potential to allow NO_3 movement beyond the rooting system compared to small grains (Power et al., 1970). This difference is attributed to greater and later-season N requirements of corn compared to small grain (Stewart et al., 1975; Olson and Kurtz, 1982). Corn is generally the prevalent crop in areas where agricultural management is associated with elevated NO_3 in groundwater in Minnesota (Anderson, 1989; Wall et al., 1992), Nebraska (Ferguson, 1990), South Dakota (Goodman et al., 1992), Pennsylvania (Hall and Risser, 1992), Iowa (Hallberg, 1985; Hallberg et al., 1993), and North Carolina (Gilliam et al., 1996). Data from North Dakota studies indicate that corn production may be associated with elevated NO_3 in groundwater in some areas (Montgomery et al., 1988; Knighton and Albus, 1993; Derby et al., 1994; Albus and Knighton, 1998) but not in other areas (Cowdery and Goff, 1994; Cowdery, 1997).

Keeney (1982) points out that other crops such as potatoes are likely to have increased potential for NO_3 leaching due to their water and N needs. Landon et al. (1993) reported that NO_3 levels in a shallow aquifer were positively correlated with the amount of N fertilizer applied to potatoes. In California, orchards are associated with elevated levels of N in the groundwater (Legg and Meisinger, 1982; Madison and Brunett, 1985).

The fate of N in soils is not only influenced by the type of crop grown, but also by the combination of crops grown in rotation. Izaurralde et al. (1995) found less N movement below a cereal-hay rotation compared to a continuous cereal rotation due to greater synchrony between NO_3 release and plant uptake. Continuous corn has been found to have the greater potential for leaching losses of N compared to a corn-soybean rotation (Kanwar et al., 1990; Hergert et al., 1993). Barry et al. (1993) predicted that adding hay to the rotation of corn-soybean-wheat would increase the amount of NO_3 leached through soils in Ontario.

Surface water

Many studies have shown the linkage between runoff, erosion, and nutrient losses (Beyerlein and Donigian, 1979; Baker and Laflen, 1983). Because the type of crop grown influences the factors that control runoff and erosion, nutrient losses will vary with different crops (Wischmeier, 1976; Wischmeier and Smith, 1978). The importance of specific crops changes as other factors that influence runoff also change. For example, Nelson et al. (1979) found that in years of average precipitation small grain crops and pasture had less runoff and erosion compared to row crops. However, small grains were generally grown in areas with greater slopes compared to row crops, and in years of above average rainfall, greater runoff and erosion was observed from the areas with predominantly small grain crops. Meals (1992) noted that N concentrations in streams with adjacent riparian areas cropped with corn was greater compared to areas of pasture or hay. Logan (1987) observed greater losses of NO_3 from corn and soybeans to tile drains compared to wheat. Research indicates that higher losses of nutrients from corn and potatoes compared to small grains and hay crops is also related to greater nutrient requirements (Stewart et al., 1975; Coote et al., 1979) combined with lower water and N use efficiency (Hatfield et al., 1993; Randall et al., 1997). Rotations of deep-rooted crops such as sunflower, safflower, alfalfa, and sweet clover will scavenge water and nitrogen that escapes the root zone of shallower rooted crops such as small grains (Brun and Worcester, 1975; Peel, 1998).

Tillage and water contamination with nitrogen

Groundwater

It is relatively well established that cultivation enhances mineralization of organic matter and the release of N (Stevenson, 1965; Bartholmew, 1965; Legg and Meisenger, 1982). Harmson and Kolenbrander (1965) state that a general characteristic of permanent vegetative cover is an established rhizosphere that has a high capacity to utilize or immobilize available N compared to cultivated crops. Studies of water and N movement confirm that greater availability of water and N under cultivated management have lead to increased water content and N in deeper soil layers compared to natural uncultivated conditions (Legg and Meisinger, 1982) even in semi-arid climates (Power, 1970; Evans et al., 1994). Comparative studies of different land uses confirm that groundwater underlying cultivated soils generally contains higher levels of NO_3 compared to natural uncultivated soils (Postma, 1991; Goodman et al., 1992; Rudolph and Goss, 1993; Dasika and Atwater, 1995; Mueller et al., 1995).

N and water movement changes from one type of tillage system to another. Hansen and Djurhuus (1997) determined that fall plowing on coarse-textured soils increased NO_3 leaching compared to spring tillage. Tillage causes disruption of macropore continuity, while reduced tillage promotes soil aggregate and macropore stability (Edwards et al., 1988; Francis et al., 1988; Steenhuis et al., 1990). Some studies have found that improved percolation of water through macropores of soils under reduced tillage cause increased NO_3 leaching and groundwater contamination (Logan, 1987; Bischoff et al., 1990; Kanwar et al., 1990; Izaurralde et al., 1995). Although Jones et al. (1995) observed greater runoff from no-till fields in the Southern Plains, they also observed that reduced evapotranspiration from these fields resulted in increased percolation of water and NO_3 to depths beyond the rooting zone. Sharpley and Smith (1994) also found greater potential for NO_3 to leach in the Southern Plains under no-till wheat compared to conventional tillage or grass. Other studies have shown that denitrification and immobilization in soils under reduced tillage decrease the amount of available N, thus contributing to less leaching of NO_3 compared to conventional tillage (Randall and Iragavarapu, 1995).

Surface water

Tillage that leaves the soil surface unprotected during erosive periods causes profound increases in runoff, erosion, and nutrient losses from fields (Wischmeier and Smith, 1978; Roose and Masson, 1985; Wicherek and Bernard, 1995). Tillage is indirectly related to surface water quality due to its impact on the delivery of runoff and eroded sediment to streams and lakes (Coote et al., 1979). When native grass was converted to conventional tillage, the total-N in runoff was increased by 2.5 times (Sharpley and Smith, 1994). In an area of untilled pristine tall grass prairie, the total-N in local streams averaged 0.15 ppm, well within the range of oligotrophic conditions (Dodds et al., 1996).

The major input of N to water resources is generally associated with sediment eroded from land surrounding streams or lakes (Burwell et al., 1977; Cihacek et al., 1994). Most studies show that fields with reduced tillage or no tillage contribute much less total-N to local water resources compared to fields tilled by more conventional methods (Beyerlein and Donigian, 1979; Sharpley and Smith, 1994). However, the input of NO_3-N, which is the most soluble component of total-N, to local water resources has often been observed to be greater from fields with reduced tillage compared to conventional tillage (Dick and Daniel, 1987; Gilliam and Hoyt, 1987). In the southern Great Plains, it has been observed that no-tilled soils generate greater runoff with less soil and total-N losses compared to conventionally tilled soils due to a surface soil crust maintained under no-till (Jones et al., 1995).

Minimum tillage studied at many sites in midwestern and eastern states induced increased infiltration, which led to significant reductions in runoff compared to conventional tillage (Edwards and Amerman, 1984; Dick et al., 1986; Mielke et al., 1986; Baker, 1987a; Donigian and Carsel, 1987; Edwards et al., 1988; Francis et al., 1988; Hatfield and Prueger, 1993; Hall and Mumma, 1994). The same studies showed that increased infiltration under reduced tillage produced a greater risk of groundwater pollution, particularly from NO_3. However, in areas with extensive systems of tile drainage, much of the water and NO_3 that percolates beyond the crop rooting zone is returned to surface water via tile outlets (Baker, 1987b; Logan, 1987; Keim et al., 1989; Keeney and Deluca, 1993; Czapar et al., 1994).

Although many studies indicate that reduced tillage may simply substitute one water quality problem with another, there is evidence that indicates many soil environments under reduced tillage promote smaller quantities of residual soil N after harvest (Wagger et al., 1993; Randall and Iragavarapu, 1995; Albus and Knighton, 1998). If the amount of available N can be consistently maintained at low levels in the soil, the opportunity for translocation to either groundwater or surface water resources is concomitantly lowered.

The effects of tillage systems on erosion, runoff, and nutrient losses may be subjugated by other factors. For instance, no-till has been demonstrated to be far more successful on well drained soils compared to wetter soils

(Edwards and Amerman, 1984; Schepers, 1987). Logan et al. (1994) found little difference in runoff and soil loss between conservation and conventional tillage on heavy textured soil types. Differences in bulk density, aggregate stability, and surficial soil chemistry between no-till and conventional tillage systems were of much greater significance for certain types of soils compared to others (Rhoton et al., 1993). Conservation tillage significantly increased biomass accumulation in a Kentucky soil but did not cause a significant change in biomass in a Canadian soil (Smith and Blevins, 1987). It is apparent from the wide range in results from tillage experiments that the effects of tillage on water quality are extremely variable, and recommendations regarding water quality protection need to be adjusted to site specific factors (National Research Council, 1993).

As a practical matter, it must be remembered that in many environments the bulk of sediment and nutrient load delivered to surface water originates from a small fraction of the watershed (Stewart et al., 1975; Campbell, 1985; Mostaghimi et al., 1997) during a few extreme weather events (Walter et al., 1979; Langdale and Leonard, 1983; Edwards, 1985; Vandaele and Posen, 1995; Oenema and Roest, 1998) or spring melt (Schepers et al., 1980). Under these circumstances the effectiveness of conservation measures in reducing runoff and nutrient losses depends on their application to critical areas (Johnson, 1979; Sharpley and Smith, 1994). Tufford et al. (1998) suggest that landuse changes farther than 150 m from the stream channel have little significant effect on stream nutrient concentrations.

Summer fallow and water contamination with nitrogen

Groundwater

In the northern Great Plains, the practice of summer fallowing has been a regular component of many cropping rotations (NDASS Staff, 1998). Fallowing ensures that a portion of the farm will yield a crop, because of the soil NO_3 and water that is carried over from the fallow (unplanted) year to spring planting of the following year. Although this practice lowers the risk of a complete crop failure, studies have shown that fallowing also lowers crop-water-use efficiency due to deep percolation of water below the rooting zone (Haas and Willis, 1962; Black et al., 1974; Halvorson and Black, 1974). The fallowed soil not only loses a portion of the precipitation that infiltrates the surface, but also a portion of the solutes present in the soil, such as NO_3 released by mineralization (Custer, 1976; Doering and Sandoval, 1976).

NO_3 leaching from summer fallowed fields was recognized in the late 1800s (Harmsen and Kolenbrander, 1965). Research has demonstrated rather conclusively that summer fallow contributes to a more regular NO_3 leaching pattern in the northern Great Plains compared to continuous cropping systems (Power, 1970; Cassel et al., 1971; Hedlin and Cho, 1974; Campbell et al., 1984; Izaurralde et al., 1995). Although NO_3 leaching from fallowed fields with coarse textures is more likely to occur, significant loss of N was observed under a fine textured soil in a semi-arid climate (Patra and Rego, 1997). In Montana, results from a drinking water well study indicated that summer fallowing contributed to groundwater NO_3 under certain climatic, physiographic, and soil conditions (Bauder et al., 1993).

Surface water

The effect of summer fallow in the northern Great Plains on surface water resources is related to soil erosion via water movement, which has been discussed previously. Summer fallow maintained through the application of regular tillage leaves the soil surface with little protection during the erosive periods of the year and contributes to excessive runoff of water and sediments. When developing the Universal Soil Loss Equation, Wischmeier and Smith (1978) used erosion losses from clean-tilled continuous fallow as the baseline to compare all other forms of management and cover. With respect to soil erosion, it is the worst-case scenario. Higher content of available N in summer fallowed soils (Hedlin and Cho, 1974; Swenson et al., 1979) coupled with greater potential for soil erosion on this type of summer fallow would be expected to cause high losses of N to surface water resources. Cassel et al. (1971) stated that losses of N from summer fallow fields could be greater than losses of fertilizer N applied to cropped fields. On the other hand, summer fallow maintained through chemical control of weeds allows stubble from previous crops to protect the soil during erosive periods. The amount of protection is related to the type and quality of stubble left standing. This type of summer fallow would have losses of N to surface water comparable to continuously farmed fields with conservation tillage systems.

Fertilizer applications and water contamination with nitrogen

Groundwater

Crops, worldwide, are more deficient in N than any other element, which limits yield and quality (Viets, 1965). Declining soil organic matter content (Hedlin and Cho, 1974; Bauer and Black, 1983) and high yielding crops have contributed to a general increase of N fertilizer use in agriculture (Hauck and Tanji, 1982). N fertilizer use increased in North Dakota from 14 tons (as N) in 1940 (TVA Staff, 1941) to about 595,000 tons in 1997 (North Dakota Ag. Statistics Staff, 1998). Hedlin and Cho (1974) calculated that long-term N removal by crops exceeded the amount applied on the Canadian prairies despite the rather large quantities of commercial fertilizer applied.

At equilibrium, the N removed by crops must come from external sources, which in modern agriculture is predominantly applied fertilizer (Stevenson, 1965). Research indicates that over time, equilibrium between mineralization and immobilization is attained in most farming systems (Hauck and Tanji, 1982). The level of total N that is maintained at equilibrium under a specific farming system will vary with soil properties (Hedlin and Cho, 1974). The assumption of steady state or equilibrium between mineralization and immobilization of N in soils under long-term farming systems has been used by many investigators to estimate N losses due to volatilization and leaching (Hauck and Tanji, 1982; Legg and Meisinger, 1982; Barry et al., 1993; Gross et al., 1993).

Nitrogen-use-efficiency (NUE) of fertilizer applications of most cropping systems rarely averages greater than 50% (Allison, 1965; Keeney, 1982; Schindler and Knighton, 1994, 1999) and generally decreases with increased amounts of N applied (Harmsen and Kolenbrander, 1965). Depending on other environmental conditions, low NUE may contribute to groundwater contamination as documented in Germany (Van der Ploeg et al., 1997). Keeney (1982) stated that improving NUE is a critical factor in reducing environmental impacts of N. N fertilizer inputs that more closely match plant needs, such as split applications, slow-release formulations, avoidance of fall application, setting realistic yield goals, and proper crediting of N sources, are general recommendations to improve N-use-efficiency (Scarsbrook, 1965; Keeney, 1982; Wall et al., 1992; Yadav and Wall, 1998). Application of N late in the growing season is particularly effective with cereal crops because more of the N is used to produce grain as opposed to unnecessary vegetative growth (Olson and Kurtz, 1982). Less leaching of NO_3 was observed from a grazed pasture (unfertilized) interseeded with a legume compared to a fertilized pasture even though the economic returns were comparable (Owens et al., 1994).

Some studies report regional connections between landuse, agricultural systems and N levels in groundwater (Mueller et al., 1995). Burkart and Kolpin (1993a,b) found a correlation between NO_3 detections in well water and nearness of wells to irrigation, fertilizer plants, and golf courses. However, no relationship was found with N application rates, which agrees with some studies (Wallrabenstein and Baker, 1992) but not others (EPA Staff, 1992). Spalding and Exner (1993) concluded that lack of correlation between fertilizer application rates and groundwater contamination with N could be explained either by denitrification in the southeastern U.S. or interception of leachate by tile drains in the eastern part of the midwestern U.S.

In the northern prairies of North America, there is little evidence to show that low NUE contributes substantially to groundwater contamination. Harmsen and Kolenbrander (1965), Power (1970), and Swenson et al. (1979) suggest that in semi-arid regions, the factor of greatest influence appears to be the rainfall or lack thereof. N movement in this region has been determined to be significantly influenced by local recharge and discharge areas (Schuh and Klinkebiel, 1994; Schuh et al., 1997). Studies in North Dakota indicate that some vertical movement of N is related to rapid preferential flow through macropores (Derby and Knighton, 1992; Derby et al., 1994; Schuh and Klinkebiel, 1994; Derby and Knighton, 1997; Schuh et al., 1997). However, most of the N was observed to move much slower and remains in the root zone (Schuh and Klinkebiel, 1994; Schuh et al., 1997). Although available N often exists in northern Great Plains soils, evidence indicates that NO_3 leaching beyond the root zone only occurs in years of above average precipitation (Power, 1970; Swenson et al., 1979; Chang and Entz, 1996). Stanford (1982) and Fanning (1988) stress the importance of accounting for the residual NO_3 in soils of semi-arid regions by regular soil testing.

Application of N fertilizer does not necessarily increase the potential for contamination. Campbell et al. (1984; 1993) demonstrated that in a wheat rotation, fields fertilized at recommended rates had less NO_3 leaching losses compared to non-fertilized fields due to improved plant growth that resulted in greater water and nutrient uptake. Available N was observed to accumulate no deeper than the upper 3 feet of a soil under a fertilized pasture in a semi-arid region and NO_3 amounts at deeper depths were comparable to an unfertilized pasture (Power, 1970). Kanwar et al. (1985) showed the same rate of fertilizer placed in different locations (surface and subsurface) resulted in greater leaching losses from surface-applied N due to preferential flow of water through macropores.

N management recommendations intended to reduce NO_3 leaching are not always effective or compatible with economic yields. Excessive NO_3 leaching to groundwater was observed to continue even after N management practices were adopted (Melvin et al., 1993; Gilliam et al., 1996). Viets (1965) and Scarsbrook (1965) found that under some circumstances split applications and slow release fertilizers showed no greater NUE than the more conventional one-time application of standard N-fertilizer. Keeney (1982) points out that the desire for economic efficiency often requires above-optimum fertilizer applications. In a long-term study, the N leached from corn under conventional tillage and no-till systems fertilized at the same rate was similar (about 20% of the amount applied), but the average yield was significantly greater under conventional tillage (Randall and Iragavarapu, 1995).

Surface water

Many studies have shown that a positive correlation often exists between the amount of agricultural land in a watershed and the concentration of N in watershed streams

(Mattikalli and Richards, 1996; Jordan et al., 1997). This phenomenon may be attributed to the application of fertilizer to agricultural fields (Coote et al., 1979; Mattikalli and Richards, 1996; Mueller et al., 1997), but not always (Schepers et al., 1980; Lindau et al., 1997). As previously discussed, many studies indicate that as N fertilizer applications increase, the efficiency of plant utilization of N decreases. This phenomenon may translate to N losses to surface water resources. However, because the fate of N in the environment is dependent on many processes, a direct correlation with water contamination often cannot be demonstrated. In fact, surface water contamination may be decreased by increased N fertilizer applications (Stewart et al., 1975; Campbell et al., 1984,1993). The reason for this counter-intuitive observation is improved plant growth that results in more plant residue that improves water infiltration, thus decreasing runoff and soil losses.

Organic waste and water contamination with nitrogen

Groundwater

Septage (human waste)

Drainage from septic systems has been identified as a possible source for elevated NO_3 in groundwater (Madison and Brunett, 1985; Fedkiw, 1991). Review of many groundwater studies revealed high correlation between domestic wells and septic systems in the northeastern U.S. but poor correlation in areas farther west (Spalding and Exner, 1993). A study in an unsewered area of Wisconsin indicated that the plumes from septic systems were too narrow to be regularly intercepted by domestic wells. Through the use of bacteriological data Felton (1991), concluded that human waste entered groundwater during the spring in a karst (weathered limestone) catchment. Incidences of elevated NO_3 in groundwater from the Oakes and Ypsilanti areas in North Dakota were suspected to be due to drainage from septic systems (NDDH, 1988).

It has been observed in studies of septic systems that most N leaving the drainfield is NH_4 but is quickly oxidized to NO_3 and susceptible to leaching (Hantzsche and Finnemore, 1992; DeSimone and Howes, 1998). Gerritse et al.(1995) found that most NO_3 had leached within 10 m of the septic drainfield. It has been estimated that a family of four will typically contribute 33 kilograms (kg) of N to the waste system per year. Over a comparable area, this is 200 times the amount of N that is added to the soil through mineralization and atmospheric deposition (Hantzsche and Finnemore, 1992). Based on assumptions of 25% loss due to denitrification and groundwater recharge related to rainfall, areas of greatest vulnerability to N contamination from septic systems are those with low rainfall and high development density.

Animal waste

Feeding cattle results in 25 to 30% of the N in the crop being removed from the soil as marketed products (Allison, 1965). About 74% of the N contained in feed for cows will be found in fresh manure. Nearness to feedlots has been linked to higher concentrations of NO_3 in drinking water wells in some studies (Barnett and Howard, 1993); however, other studies have found no significant relationship between the two (Burkart and Kolpin, 1993a,b). In a Missouri study, 75% of wells that exceeded 10 ppm of NO_3-N were within 450 ft of livestock yards, but a strong correlation could not be demonstrated [$R = 0.44$] (Sievers and Fulhage, 1991).

Many investigators have observed that NO_3 levels decrease significantly in the soil below the compacted layer that occurs in the surface 12 inches (Schuman and McCalla, 1975; Norstadt and Duke, 1982; Dantzman et al., 1983). Alego et al. (1972) found that soil NO_3 levels in the surface 2 ft of a feedlot were only slightly higher than in adjacent cropped fields. Below the 4-ft depth, NO_3 concentrations were the same beneath the feedlot and cropped fields. Some studies indicate that denitrification plays an important role in this phenomenon and is related to anaerobic conditions that develop in the surface layer of soils in many feedlots (Stevenson, 1982; Sweeten, 1990).

The amount of infiltration of water through the feedlot surface has a direct bearing on whether NO_3 can be transported to groundwater below. The surface of a feedlot is actually a series of layers with different densities. Mielke et al. (1974) found the density of the top manure layer to be 40 to 58 lbs/ft^3. The compacted manure/soil layer immediately below had a density of 62 to106 lbs/ft^3. Compacted layers of manure and soil usually provide a seal that reduces water infiltration to less than 0.002 in/hr (Mielke et al., 1974; Mielke and Mazurak, 1976). Schuman and McCalla (1975) concluded that NO_3 leaching was insignificant when percolation rates were this low. Significant leaching of NO_3 from feedlots is most likely to occur on coarse-textured soils, and when compacted conditions are not maintained due to low stocking rates, frequent disturbance of the compacted layers, or abandonment (Keeney, 1982; Sweeten, 1990).

Land applications of organic waste

Organic wastes are a source of nutrients for plant uptake, but many factors must be considered to ensure they are utilized efficiently without negative environmental effects. Waste materials from humans or animals contain aliphatic and aromatic amines that may form nitrosamines when they react with nitrous acid. Nitrosamines are carcinogenic and pose a health hazard; however, no evidence has shown that this reaction occurs to any extent in natural soils (Stevenson, 1982). Translocation of N as NO_3 from organic wastes is a major environmental concern (National Research Council, 1993).

Long-term studies have shown that NUE of crops is significantly greater for manufactured fertilizer compared to manure (Allison, 1965). The N not mineralized in the first year after application of manure generally becomes part of the soil organic matter and releases available N relatively slowly (Bartholomew, 1965). Barry et al. (1993) used a 50% credit for manure applied in the spring in Ontario to develop a N budget that showed good agreement between leaching estimates and actual N loading in tile drains.

Because manures from various sources decompose at different rates, it is necessary to determine the decomposition pattern or decay series for proper fertilization (Smith and Peterson, 1982). Constant annual manure applications that supply enough N for crop demands will ultimately cause excessive fertilization; therefore, decreasing amounts need to be applied each year (Smith and Peterson, 1982). In a long-term study on a loamy soil in Saskatchewan, recommended constant rates of manure application were found to cause regular leaching of NO_3 under irrigation and increased potential for leaching under dryland farming systems (Chang and Entz, 1996; Chang and Janzen, 1996). This study demonstrated that mineralization of manure over many years causes eventual underestimation of the N credit, which leads to increased N leaching. Chang and Entz (1996) estimated that to prevent NO_3 leaching under long-term manure applications, no more than 14 Mg/ha/yr manure (138 kg/ha/yr of N) should be applied.

The method of application or form of organic waste may make a measurable difference with respect to leaching losses. Liquid manure injected into the soil compared to surface applications resulted in the greatest loading rates of nutrients and bacteria to tile drainage on a soil in Minnesota (Foran et al., 1993). No significant loading to tile drainage was observed when manure was applied as a solid. In a California study similar results were found where liquid manure increased mineralization, crop uptake, leaching, and gaseous losses, but solid manure increased soil organic N (Legg and Meisinger, 1982).

Organic wastes from agricultural product processing often have high decomposition rates, which deplete the supply of oxygen and promote anaerobic conditions that cause denitrification (Smith and Peterson, 1982). This process can be manipulated to protect groundwater from NO_3 by irrigation applications with the waste products that promote the development of anaerobic conditions when needed.

Land applications of sewage sludge should be aged, dewatered, and applied to the soil surface for the best protection of groundwater (Smith and Peterson, 1982). It has been estimated that 50 to 75% of NH_4 in the sludge will volatilize if surface applied on sandy and clay soils, respectively. Excessive production of NO_3 from nitrification of land applied sludge may be managed by the addition of organic C.

Surface water

N contamination to surface water becomes an issue when organic wastes are subject to movement via surface runoff. Many studies have shown the connection between animal waste and the quality of certain surface water resources (Gunsalus et al., 1992; McCoy and Summers, 1992; Meals, 1992; Schlagel, 1992; Edwards et al., 1996; Jennings et al., 1998). The location of organic wastes with respect to runoff water obviously is of importance, particularly when that waste is concentrated in small areas (Fedkiw, 1992). Management of organic wastes and the runoff from these areas has a significant effect on the amount of N that is mobilized and transported to surface water resources. Well-designed storage facilities that are properly located, regular maintenance of the animal yard, and diversion of runoff reduce the potential for contamination of surface water from these facilities (Boyles, 1988).

Land application is the other critical part of animal waste management that has potential to result in N contamination of surface water (Fulhage, 1992). The transport of land-applied organic wastes is controlled by the principles of surface water runoff and soil erosion as has been previously discussed. The potential for transport of N to water resources from animal wastes applied to the fields, as with other N sources, is subject to many different factors, which makes it extremely variable within watersheds (Oenema and Roest, 1998) and among watersheds (Nelson et al., 1996). However, studies indicate that greater losses of N occur from fields fertilized with animal waste as compared to inorganic fertilizer (Stewart et al., 1975; Coote et al., 1979; Edwards and Daniel, 1994). Generally, application of organic waste to more erodible soils will result in increased transport offsite and greater potential for contamination of water resources. Usually increased application amounts also lead to increased losses; however, this is not always the case. Under some circumstances exceptionally high amounts of organic waste on the soil surface may increase infiltration of water and decrease runoff and transport of the waste (Walter et al., 1987). However, most studies show that incorporation of animal waste into the soil after application reduces the potential for translocation by surface water.

The principles of nutrient supply and demand should be followed when applying animal waste to avoid excessive buildup of nutrients in soils. Farm operations studied locally (Legg and Montgomery, 1993) and nationally (Trachtenberg and Ogg, 1994) that utilize animal waste as a fertilizer generally apply N at levels greater than crop requirements, resulting in surplus supplies in the soil. As discussed previously, testing of soil and animal waste for nutrients at regular intervals is required to determine proper application rates. It is equally important to base animal waste applications on a balance between N and P. Animal waste application rates based solely on N have resulted in high P levels in some soils (Walter et al., 1987; Stewart, 1992) that contribute to surface water problems.

Irrigation and water contamination with nitrogen

Groundwater

As stated previously, evidence shows that in some places groundwater contamination with NO_3 may be related to irrigation of crops (Madison and Brunett, 1985; Anderson, 1989; Burkart and Kolpin, 1993a,b; Spalding and Exner, 1993). Anderson (1989) found NO_3 levels in shallow surficial aquifers to be significantly higher under irrigated fields compared to either dryland agricultural fields or natural uncultivated areas. No significant difference was determined in aquifer NO_3 levels below dryland agriculture and natural areas. Madison and Brunett (1985) and Hallberg et al. (1993) concluded that no single factor adequately explains the incidence of N contamination of groundwater. Knighton and Albus (1993) reported that N in deep soil cores indicated that leaching through the root zone had occurred under dryland farming prior to initiation of irrigation. Irrigation, therefore, cannot be directly linked to NO_3 in groundwater without consideration of other factors that influence N fate in soils and the geologic materials below. Irrigation often occurs where coarse-textured soils and geologic materials and shallow groundwater also increase the potential for groundwater contamination.

Coarse-textured soils in drier climates, not viewed as being particularly valuable for dryland-agriculture, are often excellent for irrigation because of their internal hydraulic characteristics. From an agronomic point of view, soils that absorb and conduct water relatively easily are more desirable for irrigation as opposed to those that do not (Franzen et al., 1996). These soils have higher hydraulic conductivities, which allow water to percolate faster and deeper into the soil compared to finer textured soils. Many crops that are irrigated, such as potatoes, are also quite intolerant of low soil water contents, so maintening an optimal agronomic environment results in greater potential for vertical water movement (Keeney, 1982). When conditions that maximize vertical water movement through the soil profile are combined with the presence of a mobile chemical such as NO_3, the potential for groundwater contamination increases (Geleta et al., 1994).

If management methods allow large quantities of NO_3 to exist in the soil profile for extended periods, the opportunity for N movement with percolating water increases. Higher mineralization rates under irrigation management have been reported in several studies (Harmsen and Kolenbrander, 1965). An increase in groundwater NO_3 was attributed to an improved environment for mineralization and greater percolation of water after the first year of irrigation (Derby and Knighton, 1992). Long-term studies in Alberta found that higher applications of manure combined with irrigation resulted in regular leaching losses of NO_3 compared to non-irrigated soils with lower applications of manure (Chang and Janzen, 1996; Chang and Entz, 1996). Many high-value crops, such as potatoes and vegetables, have large nutrient requirements and shallow rooting depths, which further increase the potential for groundwater contamination (Keeney, 1982).

Although N management of irrigated crops has been shown to be important in reducing potential for N leaching (Spalding and Kitchen, 1988; Watts et al., 1993; Derby et al., 1994), research also indicates that water management is generally the most critical factor (Montgomery et al, 1990; Fermanich et al., 1993; Han et al., 1995; Sexton et al., 1996). Ayars and Phene (1993) found that drip irrigation of cotton was more efficient with respect to N and water use, resulting in reduced potential for groundwater contamination compared to furrow irrigation. Using model predictions, furrow irrigation was determined to have greater potential for NO_3 leaching compared to sprinkler irrigation for sorghum, corn, and wheat (Geleta et al., 1994). In North Dakota, studies of deficit irrigation water scheduling for corn grown on coarse-textured soils show that N contamination of groundwater is reduced significantly by ensuring that the soil profile is depleted of both residual N and water in the fall (Montgomery et al., 1990). A study that modeled irrigated corn growth in Minnesota predicted that NO_3 leaching could be reduced significantly with little impact on yield by not applying irrigation water until as little as 30% of the available water remained in the rooting zone (Pang et al., 1998).

Surface water

The processes that affect N availability, mobility, and translocation to surface water under irrigated agricultural systems are the same as those discussed in previous sections. The importance of some of the factors that influence these processes may become largely a function of the application of irrigation water. For example, soils that are consistently maintained at field capacity are more likely to generate runoff during rain events compared to similar soils allowed to dry down to lower water contents. If applications of irrigation water cause surface runoff or subsurface drainage that outlets to surface water, the potential for N contamination exists. The processes that affect losses of N to surface water under irrigation are no less complicated than under other forms of agricultural management as observed earlier.

Conditions common to many irrigated fields that should be considered in terms of N losses were previously discussed in detail with respect to groundwater. The following conditions may also be related to surface water contamination:

1) Sprinkler irrigation systems usually generate much less surface runoff than gravity irrigation systems. Decreased potential for surface water contamination via runoff from fields with sprinkler irrigation is a reasonable expectation.

2) Irrigation is often applied on relatively flat areas of well-drained soils that often have coarse textures (Scherer et al., 1996). Due to high permeability and low slope gradients, these soils do not generate much surface runoff, particularly under sprinkler irrigation systems. This would reduce expectations of N losses to surface water.

3) Irrigated crops generally receive higher inputs of fertilizer due to increased yield potential compared to nonirrigated crops. The efficiency of crop utilization of nutrient inputs usually decreases as inputs increase, causing higher residual N levels in the soil (Harmsen and Kollenbrander, 1965). In addition, the improved water regime for crops brought about by irrigation also has been related to increased release of organic N via mineralization. The increased availability of soil N caused by these processes could contribute to greater losses of N from irrigated fields to surface water if runoff occurs.

4) As mentioned in point number 2, coarse textured soils are often selected for irrigation. Although runoff is less likely to be initiated on these soils, when runoff does occur, coarse-textured soils, which lack the stabilizing influence of clay particles (Meyer, 1985) and organic matter, are generally less resistant to detachment by the erosive forces of water. Under such circumstances, losses of N are likely to be significant.

5) If tile drains are used to control the water table under irrigated fields, surface water may be contaminated by tile line return flow. Although tile drainage is rare in North Dakota, NO_3 loading to surface water has been observed in areas where tile drains have been installed (J.T. Moraghan, personal communication).

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CHAPTER 2

Assessment for Contamination of Water Resources from Diffuse Nitrogen Sources

Potential for Groundwater Contamination with Nitrogen

Assessment factors

Factors that affect the fate of N in the environment are many and their interrelationships complex. Several attempts have been made to model the processes through which these factors operate to predict N translocation under cropped systems (Geleta et al., 1994; Jabro et al., 1995; Han et al., 1995; Madramootoo et al., 1995; Van der Ploeg et al., 1995; Yiridoe et al., 1997; Pang et al., 1998), natural systems (Creed et al., 1996), or due to certain physical characteristics (Li and Ghodrati, 1995). Some of these models have been combined to estimate groundwater vulnerability over large regions (Khakural and Robert, 1993; Shaffer and Wylie, 1993; Wylie, et al., 1995).

Review of studies related to N and groundwater reveal complexity that is unique to a specific site, but also some predictable patterns with respect to certain factors. In general, groundwater contamination is controlled by: 1) contaminant mobility; 2) contaminant availability; and 3) accessibility of the water resource. Research shows that some factors consistently exert significant control over the processes that affect mobility, availability, and accessibility. Combinations of these factors have been used to develop several different types of groundwater assessment systems (Aller et al., 1985; Trojan and Perry, 1988; Cates and Madison, 1991; Pettyjohn et al., 1991; Seelig, 1994).

Natural factors

The mobility of a contaminant with respect to groundwater is related to chemical properties that affect ease of transport with water and adsorption to soil particles. The NO_3^- ion is the most mobile form of N in water because of its high solubility and negative charge. The NH_4^+ ion, however, is not as soluble and adsorbs to soil clays and organic matter due to its positive charge. Studies show that the relatively immobile NH_4 ion rarely reaches levels of concern in groundwater compared to NO_3, which consistently poses health problems in a certain percentage of domestic water wells. Natural factors that favor the mineralization of N will increase the potential for groundwater contamination.

Some research studies have shown an inverse relationship between the concentration of NO_3 in groundwater and the presence of shale fragments (Hendry et al., 1984; Korom et al., 1996). The shale content of aquifers may be an important factor affecting vulnerability, but further research is required before this factor can be applied with any level of confidence.

Soil aeration

Well aerated soils would be expected to provide an environment favorable to mineralization of N to NO_3 with only minimal loss through denitrification. Mueller et al. (1995) found a positive correlation between well drained soils and NO_3 in groundwater. Gerla et al. (1994) found a similar relationship with coarse textured soils. Although immobilization and plant uptake will counterbalance the NO_3 released by mineralization, soils with internal drainage characteristics that allow good aeration have potential to serve as sources for the NO_3 form of N. **Soils with internal drainage characterized as well, somewhat excessively, or excessively drained are considered to contribute to groundwater vulnerability with respect to NO_3 contamination (Table 1).**

Soil texture

Soil texture influences mobility, availability, and accessibility of N in several ways. As mentioned in the previous paragraph, ease of drainage through coarse-textured soils often results in well-aerated conditions that favor the NO_3 form of N. Nommik (1965) states that fixation of NH_3 and ammonium may be a significant factor in preventing NO_3 leaching under certain soil and climatic conditions. Fixation and adsorption of NH_3 and NH_4 is primarily controlled by the amount of clay and organic matter in the soil. Coarse-textured soils characteristically have only low amounts of

Table 1. Natural factors that contribute to groundwater vulnerability to nitrogen.

Natural Factors	Criteria
Soil aeration	Soils that are well, somewhat excessively, and excessively drained
Soil texture	Soils that are classified with a sandy, sandy-skeletal, or fragmental USDA family particle size
Depth to aquifer	Less than 50 feet from the surface to the top of the saturated zone in the aquifer
Hydrologic recharge area	Greater than 30 inches to the $CaCO_3$ in the soil profile

both, thus limiting their ability to retain either NH_3 or NH_4. Under these conditions, NH_3 losses from volatilization are likely to be high, but leaching losses of NO_3 formed by nitrification may be limited due to the sensitivity of *Nitrobacter sp* to NH_3. However, any NO_3 formed by nitrification would be available to be quickly leached from the soil profile. Aquifers underlying coarse-textured soils are more accessible to surface contaminants, because their infiltration and permeability allow quicker and deeper penetration of water compared to finer textured soils. **Soils that are classified in the USDA family particle size groups of sandy, sandy-skeletal, or fragmental should be considered to be relatively vulnerable to NO_3 leaching (Table 1).**

Depth to aquifer

The accessibility of contaminants to groundwater resources can be related to the distance from the ground surface to the top of the aquifer (Aller et al., 1985; Trojan and Perry, 1988; Cates and Madison, 1991; Pettyjohn et al., 1991; Seelig, 1994). **An aquifer within 50 ft of the ground surface should be considered vulnerable to NO_3 contamination (Table 1).** Some studies indicate that extremely shallow groundwater depths may be linked to high rates of denitrification that reduce the potential for contamination from NO_3 (Burkart and Kolpin, 1993b; Spalding and Exner, 1993). The effects of denitrification on the availability of NO_3 are accounted for within the aeration factor.

Groundwater recharge area

The depth to a zone of calcium carbonate ($CaCO_3$) accumulation in a soil profile is related to vertical water movement through the soil profile and is controlled by climate and landscape position (Buol et al., 1973). Deeper depths to $CaCO_3$ indicate greater leaching due to downward movement of water. Although St. Arnaud (1979) pointed out that the depth of $CaCO_3$ accumulation in soils was influenced by other factors, such as the partial pressure of CO_2 and carbonate ion concentration, he determined that the depth of $CaCO_3$ accumulation was deepest under soils with the greatest amount of water percolation. Seelig (1994) used soil survey information to relate the depth of $CaCO_3$ leaching to groundwater recharge areas (Table 1). **When $CaCO_3$ is not present in the upper 30 inches of the soil profile, this soil should be considered as a recharge site that would be vulnerable to groundwater contamination.**

Anthropogenic factors

The previously discussed natural factors provide an estimate of aquifer vulnerability as defined by Pettyjohn et al. (1991). In addition, human activities can be categorized into a group of factors that also affect the potential for N contamination of groundwater. These factors mostly influence the availability of N with respect to groundwater contamination and have less influence on N mobility and groundwater accessibility. When anthropogenic factors are considered along with natural factors, an assessment is defined as one of groundwater sensitivity (Pettyjohn et al., 1991).

Concentrated human activity

Many activities of man affect the availability of N. Some activities that may be related to groundwater contamination are: 1) facilities related to animal production; 2) production, storage and use of synthetic fertilizers; 3) facilities related to human waste disposal; and 4) location and condition of water wells (Madison and Brunett, 1985; Barnett and Howard, 1991; Wallrabenstein and Baker, 1992; Rudolf and Goss, 1993). However, studies also show that although relationships between these activities and NO_3 in groundwater often exist, in many cases the relationship of proximity is difficult to establish with confidence (Burkart and Kolpin, 1993a,b). **Boundaries of 1) city limits and 2) business property, recreational facilities, and human inhabitance (i.e. farmsteads) outside city limits delimit areas of concentrated human activity where NO_3 availability is increased for groundwater contamination (Table 2).**

Cultivated land

Cultivation of soils contributes to greater availability of NO_3 through increased mineralization and greater mobility due to higher soil water contents compared to soils left under native grasses or trees (Stevenson, 1965; Bartholomew, 1965; Power, 1970; Legg and Meisinger, 1982; Evans et al., 1994). It has also been demonstrated that cultivated crops generally are not as efficient users of nutrients and water compared to native vegetation

Table 2. Anthropogenic factors that contribute to increased availability of nitrogen.

Anthropogenic factors	Criteria
Concentrated human activity	Areas within city limits, or within boundaries of business or inhabitance outside of city limits
Cultivated land	Predominant landuse requires manipulation of soil surface for the purpose of growing crops
Crops	Rotations with corn, potatoes, and vegetable crops
Summer fallow	Rotations that include idle periods for the purpose of storing water and nitrogen in the soil
Irrigation	Areas that receive water applications in addition to that received from natural rainfall

(Harmsen and Kolenbrander, 1965; Power, 1970; Evans et al., 1994; Izaurralde et al., 1995). Many other factors will ultimately determine whether groundwater under cultivated soils will become contaminated with NO_3. **Areas where manipulation of the soil for the purpose of growing crops is the predominant land use are considered to have increased availability of NO_3 for groundwater contamination (Table 2).**

Crops

Differences in plant uptake of nutrients and water use affect availability of NO_3 in the soil during the growing season. Corn, potatoes, and vegetable crops have growth patterns that increase the availability compared to other crops like small grains (Power, 1970; Stewart et al., 1975; Keeney, 1982; Olson and Kurtz, 1982). **Fields that are rotated with corn, potatoes, and vegetable crops are considered as having increased availability of NO_3 for groundwater contamination (Table 2).**

Summer fallow

The practice of summer fallow (black or chem-fallow) contributes to increased availability and mobility of NO_3 in the soil due to a build-up of NO_3 from mineralization during the uncropped year and low water-use efficiency that causes deep percolation of unused water (Haas and Willis, 1962; Black et al., 1974; Halvorson and Black, 1974). Bauder et al. (1993) found a strong link between summer fallow and NO_3 concentrations in water wells. **Fields that include idle periods between crop rotations for the purpose of storing nitrogen and water in the soil are considered to have increased availability of NO_3 for groundwater contamination (Table 2).**

Irrigation

Many studies have reported elevated NO_3 concentrations in groundwater beneath irrigated land compared to other land uses (Anderson, 1989; Montgomery et al., 1988; Burkart and Kolpin, 1993a,b). However, Cowdery and Goff (1994) and Cowdery (1997) found that this relationship is not evident in all irrigated areas. Despite the uncertainty of linking increased NO_3 in groundwater beneath all irrigated fields, there are several good reasons why irrigation increases the potential for NO_3 contamination. Keeney (1982) points out that the large nutrient requirements and shallow rooting depths of many irrigated crops increases the potential for NO_3 leaching. It has generally been found that increased N application rates result in greater quantities of residual NO_3, because of low NUE (Allison, 1965; Harmsen and Kolenbrander, 1965; and Schindler and Knighton, 1994;1999).

Although the application of irrigation water in amounts greater than crop needs are not encouraged or practiced in general, additional amounts of water applied over the rainfall received will increase the potential for vertical water movement and leaching (Geleta et al., 1994). Most investigators recognize irrigation water management as the most critical factor in determining the extent of NO_3 leaching (Montgomery et al., 1990; Fermanich et al., 1993; Han et al., 1995; Sexton et al., 1996).

Given the general conditions of N and water under an irrigated environment, the potential for NO_3 leaching to occur must be considered greater during the growing season compared to the general conditions under a dryland environment. The extent to which the potential is increased will vary substantially with different crop rotations, soils, geology, climate, etc. **Although research results vary with respect to the effects of irrigation on groundwater quality, in general fields that receive applications of water in addition to amounts received through natural rainfall are considered to have increased availability of NO_3 for groundwater contamination (Table 2).**

Determination of aquifer sensitivity to nitrogen

First, determine the location of glacial and alluvial aquifers. This can be accomplished by referring to the County Groundwater Studies Report published by the North Dakota Water Commission. If the location in question is underlain by one of these types of aquifers, the natural factors should be assessed to determine the vulnerability.

Aquifer sensitivity is found by combining the results of the vulnerability determination with the assessment of the anthropogenic factors. The factors outlined in the preceeding section can be combined in many different ways to determine groundwater vulnerability and/or sensitivity as demonstrated by Seelig (1994). From a practical point of view, it is desirable to limit the number of categories. However, the system also must be composed of enough discretely different categories to assist with practical management decisions. As discussed previously, there are many ways to assign weight to the factors or combine them into an overall sensitivity rating. One possible method of determining aquifer sensitivity to NO_3 contamination is described in Appendix I.

Potential for Surface Water Contamination with Nitrogen

Assessment factors

As mentioned previously, the many factors that affect the fate of N in the environment are interwoven into complex relationships. Modeling of nutrient transport to surface water resources on a watershed scale has been used to predict impacts to water quality for many years

Anthropogenic factors

Human activities can be categorized into a group of factors that also affect the potential for N to contaminate surface water resources. The effect of land drainage via road ditches has already been discussed with respect to the natural drainage network and is most logically accounted for with the natural drains under the proximity factor. Anthropogenic factors (Table 4) are considered important because of their effects on N availability. Anthropogenic factors are considered together with the natural factors that define vulnerability to estimate the overall sensitivity of a given surface water resource to N contamination.

Areas of concentrated human activity

When confined in small areas (towns etc.), the activities of man lead to conditions that increase the accessibility and availability of N to surface water runoff. Mueller et al. (1995) reported that high NH_3 levels in streams were significantly correlated with proximity to urban areas. Nelson et al. (1979) found that septic system discharge to a nearby stream contributed significantly to nutrient loads particularly in drier years. After review of research and monitoring studies, Fedkiw (1991, 1992) concluded that strong evidence supported the contention that waste from animal production facilities have played a role in observed increases in N concentrations in surface water near those facilities. **Boundaries of 1) city limits and, 2) business property, recreational facilities, and human inhabitance (i.e. farmsteads) outside city limits delimit areas of concentrated human activity where NO_3 availability is increased for surface water contamination (Table 4).**

Table 4. Anthropogenic factors that contribute to increased availability of nitrogen.

Anthropogenic factors	Criteria
Concentrated human activity	Areas within city limits, or within boundaries of business or inhabitance outside of city limits
Cultivated land	Predominant landuse requires manipulation of soil surface for the purpose of growing crops
Tilled summer fallow	Rotations that include idle periods for the purpose of storing water and nitrogen in the soil and weeds are controlled by tillage
Rangeland, pastureland, and grazed forestland	Predominant landuse entails grazing of domesticated animals

Cultivated land

Mueller et al. (1995) found that surface water from watersheds where agriculture was the main landuse had higher nutrient concentrations compared to streams from undeveloped watersheds (Mueller et al., 1995). These results are supported by many other studies. Cultivated soils contribute to greater availability of NO_3 through increased mineralization (Stevenson, 1965; Bartholomew, 1965). It has also been demonstrated that cultivated crops generally are not as efficient users of nutrients compared to native vegetation (Harmsen and Kolenbrander, 1965). Cultivation or tillage exposes the soil surface to the erosive power of surface runoff, which causes soil detachment and removal (Wischmeier and Smith, 1978).

Wischmeier and Smith (1978) showed that erosion from cultivated soils is dependent on many factors, of which the type of crop and management of its residue has a significant effect on soil erosion. Crop rotations that maintain residue on the surface during the erosive periods will reduce soil erosion, which has water quality benefits (Wischmeier, 1976). Compared to conventional tillage systems, sediment and total-N losses are much smaller for reduced tillage or no-tillage, which also results in less total-N that reaches water resources (Beyerlein and Donigian, 1979; Sharpley and Smith, 1994). However, water resources receiving runoff from reduced tillage fields often have greater concentrations of NO_3 compared to runoff from conventionally tilled fields (Dick and Daniel, 1987; Gilliam and Hoyt, 1987).

Although the benefits of certain types of cropping systems are recognized, the evidence continues to show that much greater losses of N occur from cropland compared to less intensive landuse. Maintaining a protective cover on the soil surface is recognized as generally being a benefit to water quality with respect to N. **Areas where manipulation of the soil for the purpose of growing crops is the predominant land use are considered to have increased potential to serve as a source of N for surface water contamination compared to areas of less intense activity, such as rangeland (Table 4).**

Cropland with tilled summer fallow

Soil maintained bare of vegetation by tillage is the most vulnerable to erosive energy and generates the greatest soil losses. N associated with the eroded sediment substantially exceeds the N that may be lost in the solution (Dean, 1983). The combination of stored NO_3 in summer fallow (Hedlin and Cho, 1974; Swenson et al., 1979) plus large losses of N with eroded sediment increases the availability of N to water resources. **Fields that are rotated with tilled summer fallow for the purpose of storing nitrogen and water in the soil are considered to have greater potential to serve as a source of N for contamination of surface water compared to fields where tilled summer fallow is not practiced (Table 4).**

Rangeland, pastureland, and grazed forestland

Evidence shows that undisturbed natural systems or permanently vegetated systems utilize N with greater efficiency than cultivated crops (Harmsen and Kolenbrander, 1965; Evans et al., 1994), which results in less NO_3 available to surface runoff. Wischmeier (1975) used the universal soil loss equation to determine runoff and erosion from permanently vegetated sites, which was substantially less compared to disturbed soils. Sharpley and Smith (1994) found that conversion of grassland to conventional tillage caused a 2.5-fold increase in total-N in stream water.

The water quality benefits of certain landuses have been described previously. Under conditions beneficial to water quality, vegetation utilizes N efficiently, allowing only minimal quantities of residual NO_3 to exist in the soil. Little impact from sediment derived N occurs to surface water under these conditions because runoff is minimized due to soil structure that promotes high infiltration and soils are well protected by vegetation and residue. Water often reaches streams in these areas as groundwater flow. Minimal amounts of N were found in streams from watersheds dominated by undisturbed forestland (Mulholland and Hill, 1997) and grassland (Dodds et al., 1996) and was almost entirely of soluble form.

It is recognized that grazing causes soil disturbance within permanently vegetated areas, and the disturbance increases with intensity of grazing. Consequently, overgrazing of pastureland and rangeland contributes to soil degradation and erosion (Stoddert et al., 1975). Whether soil erosion occurs from cropland or permanently vegetated areas, the water quality impacts remain the same as discussed in previous sections. **Rangeland, pastureland, and grazed forestland have greater potential to serve as a source of N for contamination of surface water compared to ungrazed areas (Table 4).**

Determination of surface water sensitivity to nitrogen

The determination of surface water vulnerability to N must first be completed before sensitivity determination can be done. The six natural factors used to assess potential contamination of surface water with N must be combined to arrive at an overall vulnerability rating.

Surface water sensitivity to N contamination is determined by combining the results of the vulnerability assessment with the results of the assessment for N availability due to human activities. As discussed previously, there are many ways to assign weight to the factors or combine them into an overall sensitivity rating. One possible method of determining aquifer sensitivity to NO_3 contamination is described in Appendix II.

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CHAPTER 3

Management Recommendations for Water Resource Protection

A Perspective on Nitrogen Management for Water Resource Protection

As previously discussed, the connection between N and anthropogenic activities was known well before the 20th century. However, environmental consequences of this relationship have not been quite as obvious compared to agronomic causes and effects. People in the 19th century knew that the process of N leaching from soils had negative effects on crop yields, but they did not suspect that leached N would contribute to degradation of their local wells. As science has advanced our knowledge in the 20th century, we now realize that the consequences of N losses encompass a much broader area than just agronomic concerns. We also understand that the impacts of N losses are closely connected to anthropogenic activities, but that the connection is complicated and not easily described or predicted.

Research has shown that many factors affect water quality of a given water resource and the effects of management vary from one area to another (Magette et al., 1988; Goodman et al., 1992; Anderson et al., 1993; Ward et al., 1993; Logan et al., 1994). In other words, local conditions influence the effectiveness of management systems designed to control nonpoint source pollution, so specific management practices cannot be assumed to produce the same effects under all circumstances (Christensen, 1983). For example, many investigators have demonstrated that alternative practices that reduce potential for water contamination also maintain yields at levels comparable to those produced using conventional management practices (Schweizer, 1988; Montgomery et al., 1990; Ayars and Phene, 1993; Martin et al., 1993; Nokes et al., 1993; Watts et al., 1993a). Reduced tillage is often an alternative management practice used to accomplish both water quality protection and yield enhancement due to increased storage of soil water (Unger, 1986). However, Deibert et al. (1986) and Tanaka (1989) found that in the northern Great Plains enhanced yields did not occur in wheat crops after fallow even though increased soil water storage was detected in soils under reduced tillage. Other studies have shown that alternative management practices may reduce but not eliminate N leaching losses (Randall et al., 1993), and in some cases alternative management practices can reduce leaching losses of N to acceptable levels only if yields are reduced (Melvin et al., 1993).

Management practices used to protect water resources from nonpoint source pollution were first recognized in Public Law 92-500, known as the 1972 Federal Water Pollution Act. These practices were required to meet criteria of effectiveness with respect to water protection, economics, and social acceptance (Bailey and Waddell, 1979; Haith and Loeher, 1979; Johnson, 1979). Management practices that meet these criteria have been coined as "best management practices" or BMPs. Determining whether certain management practices meet the effectiveness criteria as outlined in Public Law 92-500 has been a daunting task due to the changes in management impacts as natural environmental factors change across the landscape (Johnson, 1979; Baker and Johnson, 1983; Daniel et al., 1991; Park et al., 1994).

Nonpoint source pollution control through adoption of various management practices depends on knowledge of the interaction with local environmental conditions (Stewart et al., 1975; Dean and Mulkey, 1979; Christensen, 1983). Soil survey information is critical to the understanding of these interactions (Frere, 1976; Johnson, 1979). Soil type has been identified by several investigators as the key to success or failure of certain management practices (Edwards and Amerman, 1984; Schepers, 1987; Logan et al., 1994). The assessment of water resources for N contamination discussed in the previous section is based on soils information, which can be directly related to management recommendations outlined below. It is not known whether the following recommended practices meet the effectiveness criteria for BMPs as defined in Public Law 92-500. However, analysis of research results cited earlier combined with knowledge of environmental conditions in North Dakota indicate each of the recommended practices should lower the potential for N contamination of water resources under certain circumstances.

Types of Management Practices for Nitrogen Control

Recommendations to reduce the potential for N to contaminate water resources may be grouped into the following general categories: 1) farmstead and/or household practices; 2) fertilizer application practices; 3) soil and land management 4) irrigation management. Many of the recommended practices are beneficial to both groundwater and surface water. However, this is not always the case, and under some circumstances, practices used to otect surface water may have negative impacts on oundwater or vice versa.

armstead and/or household anagement practices

Positive correlations between areas of concentrated uman activity, both rural and urban, have been demonrated by many investigators (Fedkiw, 1991). Farmstead nd/or household practices have been recommended argely to address groundwater contamination and wells Weston, 1994; Nowatzki et al., 1996; Nowatzki et al. 998a,b,c). Many of these recommendations may also enefit surface water resources, because they reduce the availability of N to runoff. The following practices are recommended to reduce water resource contamination from areas of concentrated human activity such as farmsteads or residential areas:

1) **Water wells should be constructed according to modern standards as outlined in Article 33-18 of the North Dakota Century Code, "Water Well Construction and Water Well Pump Installation", (ND Dept. of Health) to prevent surface water infiltration through seams, cracks, or holes in the casing.** They should not be located in depressional areas or low landscape positions that receive surface water runoff. Wells should be located at least: 1) 50 feet from privy pits, cesspools, septic tanks, sewage filtration fields, and silage storage areas; 2) 100 feet from livestock operations such as barnyards or feedlots; and 3) 250 feet from livestock manure storage areas, or locations where milkhouse waste-water is discharged. In order to ensure that surface water does not enter the top of the well, the casing should extend at least 12 inches above the ground elevation. In areas subject to flooding the well casing should extend at least 24 inches above the highest known flooding elevation and be surrounded with earth fill.

2) **Wells no longer used or abandoned should be sealed according to appropriate methods as outlined in NDSU Extension publication AE-996, A Guide to Plugging Abandoned Wells.**

3) **Septic system installation should conform with standard design, siting, and construction requirements as outlined in NDSU Extension Service publication AE-892, Individual Home Sewage Treatment Systems, to ensure proper waste disposal and reduce the availability of N.**

4) **Septic systems should be maintained through regular inspections and avoiding dumping excessive amounts of grease, oil, or caustic chemicals that will plug or damage the system.** Generally the solids in most septic tanks should be pumped out every two to three years.

5) **Storage of any type of N fertilizer should be located in an area protected from excessive surface runoff or water infiltration.** A fertilizer storage area should have an impermeable surface from which runoff is diverted. In the case of commercial fertilizers, they should be stored in areas where the integrity of packaging or container can be maintained and where spills or leaks can be easily detected and managed.

6) **The surface of an animal yard or feedlot should be maintained by allowing the compacted layer of manure immediately above the soil surface to remain undisturbed.** This 3- to 4-inch layer serves as a seal with respect to NO_3 leaching. Only when the area is no longer used for animal production, should all the manure be removed from the surface.

7) **Animal manure (liquid or dry) should be stored in properly designed facilities that are protected from excessive runoff, flooding, or overflow conditions that would allow contamination of surface water.** Proper design and location of animal waste facilities may be determined from the Midwest Planning Service Bulletin 18, ***Livestock Waste Facilities Handbook.***

Fertilizer application practices

Keeney (1982) has suggested that the primary relationship between water resources and the application of N fertilizer is related to the NUE of plants. As discussed in previous sections, agricultural crops are inherently less efficient users of N compared to native vegetation. Research has shown that under agricultural systems greater amounts of N are available to move to water resources. It is important then, to ensure that fertilization methods that improve the efficiency of N uptake by agricultural crops be applied to the greatest extent possible. N inputs to water resources from agricultural areas will probably never be comparable to inputs from areas of native vegetation; however, adopting certain practices related to fertilizer application may help maintain acceptable N loading rates. The following fertilizer application practices should improve the efficiency of N uptake by agricultural crops. Not all of these practices will be effective under all circumstances, but at least some of them can be expected to minimize nitrogen contamination of water resources in any given situation. Referring to the results of water resource assessment described earlier will help to determine which of the recommended practices would be most useful.

1) **The level of residual soil NO_3 should be determined by analysis of soil samples taken from each cropped field.** Sampling should be done using accepted procedures such as those outlined in NDSU Extension Service circular SF-990, Soil Sampling for Fertilizer Recommendations. In addition to proper sampling, application recommendations

that result in efficient N use also depend on reasonable yield goals. If estimated yields are attained in one out of 10 years, then over-fertilization and lower N efficiency will also have occurred in nine out of 10 years. N applications based on results of soil testing are beneficial to both surface water and groundwater.

2) **N applications should be managed very carefully on coarse textured soils due to the high potential for leaching losses.** Fall applications are not recommended on coarse-textured soils. Split applications of N should be considered on these soils to ensure that adequate N is available to the plant during critical growth stages. This practice helps reduce the potential for N losses during less critical periods of plant growth. Applications are commonly split between a preplant application and a topdress or sidedress application during early vegetative growth. Another practice that may be beneficial under these conditions is the use of slow-release N fertilizer or nitrification inhibitors. Management practices that improve the efficiency of N uptake on coarse textured soils are particularly important to groundwater protection.

3) **Fall applications of anhydrous NH_3 and urea should be delayed as long as possible.** When soil temperatures are above 45° F microbial conversion of NH_3 to NO_3 (nitrification) is relatively rapid; consequently the potential for leaching losses increases. Some research has shown the recovery of fall-applied NH_3 the following spring may be as low as 16% if applied in mid September and as high as 90% if applied the beginning of November. Proper timing of fall-applied NH_3 is important to both surface and groundwater protection.

4) **Animal manure used as a fertilizer should be analyzed to determine its nutrient value.** As with commercial fertilizer, manure applications should be guided by test results and reasonable yield goals. Unlike commercial fertilizer, the rate of decay or mineralization of animal manure must also be accounted for to estimate the amount of N released each cropping season after initial application. These practices are important to both surface water and groundwater protection.

5) **Animal manure applied to the soil surface should be immediately incorporated or injected on soils with slopes greater than 6%.** Injection of liquid manure is not recommended into coarse textured soils that overlay shallow groundwater resources.

6) **Avoid animal manure applications on frozen ground or during excessively wet periods.** Manure should never be applied any closer than 25 to 30 feet from a stream or lake or within 200 feet if the soil is frozen.

Soil and water management practices

Research has shown that many soil management practices, such as tillage, that are beneficial to agronomic production have detrimental effects with respect to water resources. Research also shows that both agronomic and environmental effects of soil management practices are closely related to local soil properties and other environmental conditions. Soil management practices are many and varied. The most beneficial use of these practices that balances agronomic requirements with environmental protection will depend on knowledge of the relationships between these practices and local conditions.

In general, the use of soil and water conservation practices (SWCP) have demonstrated reductions in the total load of agricultural chemicals that leave cropped fields (Stewart et al., 1975; Beyerlein and Donigian, 1979; Shoemaker and Harris, 1979). However, SWCP are often not effective in controlling solution losses of contaminants compared to sediment losses (Baker et al., 1979; Beyerlein and Donigian, 1979; Smith et al., 1979; Dick and Daniel, 1987; Gilliam and Hoyt, 1987). Practices commonly used to control soil erosion and sediment delivery cannot be applied with the same confidence for water quality protection (Woolhiser, 1976; Johnson, 1979; Haith and Loehr, 1979; Wineman et al., 1979).

Most SWCP have been used primarily to control soil erosion; water resource protection has been viewed as an added benefit that is often not well documented. However, some soil management practices have been applied specifically to help protect water resources. Vegetative filter strips and riparian buffers have been used with different degrees of success to filter runoff water prior to reaching streams and lakes. Often the vegetative filters are most successful in removing sediment compared to the soluble phases of N or P (Dillaha, 1989; Parsons et al., 1990; Fabis et al., 1993). On the other hand, much evidence exists to show that riparian buffer zones have tremendous capacity to denitrify NO_3 carried into these locations (Hanson et al., 1994; Groffman et al., 1996; Bragan et al., 1997; Gold et al., 1998; Horwath et al., 1998; Jacinthe et al., 1998). Gilliam (1994) observed that the vast majority of NO_3 was removed from runoff water in narrow vegetated zones of wet soils before it reached areas traditionally considered as riparian buffers. Research has also shown that wetlands also maintain exceptionally high capacity to denitrify NO_3 (Buresh and Patrick, 1981; Patrick, 1982; Jorgensen, 1989). Significant reduction in the N load of tile drainage water originating from irrigated fields was observed after it passed through a cattail marsh near Oakes, North Dakota (J.T. Moraghan, personal communication).

Some soil management practices are counteractive with respect to surface water and groundwater protection (Hickman et al., 1994). Many studies have shown practices designed to improve surface water protection through reduced runoff have caused increased potential for groundwater contamination due to greater percolation of soil water through the root zone (Edwards and Amerman, 1984; Dick et al., 1986; Mielke et al., 1986; Baker, 1987; Edwards et al., 1988; Francis et al., 1988; Hatfield and Prueger, 1993). In particular, no-till systems and some types of terraces (Frere, 1976) are most likely to exhibit large contrasts between groundwater and surface water protection.

Many studies have shown that SWCP are usually not as effective when applied singularly as compared to combinations of two or more practices. For example, Burwell et al. (1977) showed that contour farming with terraces substantially reduced losses of water, sediment, and N compared to contour farming only. Similarly, Beyerlein and Donigian (1979) found that contour farming with terraces reduced the loss of sediment by 55% compared to a 37% reduction for contour farming only. However, Smith et al. (1979) found that cost-effectiveness of combinations of SWCP generally decrease compared to individual SWCP, particularly if certain practices were used. The least cost effective SWCP were intensive terrace systems, substituting sod-crops for row crops, and removing land from production. The most cost effective SWCP were conservation tillage, contouring, and strip-cropping.

Effective use of the following soil management recommendations to protect water resources from N contamination will require a site specific approach to adequately account for natural variation in environmental factors (Stewart and Woolhiser, 1976; Bailey and Waddell, 1979). The knowledge learned from water resource assessments described in the previous section can be used to determine the most appropriate combination of recommendations.

1) **The use of summer fallow should be eliminated or reduced to a minimum.** If fallow is used in the rotation, herbicides should be used for weed control as opposed to tillage. Tilled or "black" fallow is a potential threat to both surface water and groundwater resources; untilled "chem" fallow is mostly a potential threat to groundwater resources. More intensive cropping makes better use of water and nutrients stored in the soil.

2) **Crop rotations should include high residue crops for soil erosion protection, legumes for N fixation, and deep rooted crops to scavenge N that escapes shallow rootzones of other crops.** Rotations with these types of crops will help protect both surface water and groundwater.

3) **Cover crops should be planted when only minimal crop residue is left after harvest to protect soils during uncropped periods of the year.** Although cover crops may tie-up available soil NO_3 and help protect groundwater, the greatest benefit is related to reduced erosion and surface water protection.

4) **Conservation tillage should be used to provide greater protection from soil erosion and reduced runoff.** Conservation tillage is defined as any tillage and planting system that maintains at least 30 percent of the soil surface covered by residue after planting or, where soil erosion by wind is the primary concern, maintains at least 1,000 pounds per acre of flat small grain residue on the soil surface during critical erosion periods. On many soils in North Dakota, conservation tillage has agronomic benefits due to increased water storage. Soil types that receive low average annual rainfall, have high evaporation, are coarse textured, or on steep slopes tend to be the driest and would benefit the most from additional stored water. This practice is most likely to have its greatest effect on surface water resources by reducing the load of N associated with sediment. Under some circumstances this practice may contribute to increased NO_3 in groundwater resources.

5) **Farming operations should follow the local contour of the landscape as much as possible.** Usually this practice is most effective where slopes are uniform and other practices such as strip cropping, terracing, and grassed waterways are also applied. In areas where slopes are nonuniform and irregular, large fields should be divided into smaller units aligned in a transverse direction across the slope gradient. These practices are used to reduce runoff and erosion and are most beneficial to surface water protection.

6) **Grassed waterways should be maintained to reduce gully formation from concentrated flow of runoff water.** Although this practice is primarily meant to prevent gully erosion, it may also serve to trap sediment and in some cases reduce soluble nutrients if water infiltrates the soil surface. As mentioned before, grassed waterways are used in combination with other practices. Grassed waterways are not recommended for steep slopes where erosion control cannot be accomplished or on flat slopes where excessive sedimentation occurs. The total land area lost to waterway development has been determined to be essentially the same as the area that would be abandoned due to uncrossable gullies. However, maintenance of waterways in critical areas does provide for relatively unimpeded access for equipment compared to land with uncrossable gullies. This practice is most beneficial to surface water protection.

7) **Under circumstances where fields have long uniform regular slopes, terraces may be used for erosion control.** Depending on how water is diverted behind the terrace, some amount of deposition and infiltration will occur. Gradient

terraces divert water to grassed waterways and are most effective in areas of uniform slope of gentle to moderate steepness. Gradient terraces would have the greatest benefit to water quality by reducing sediment loads to surface water. Level terraces are designed to store water and depend on significant infiltration for water removal. Because level terraces allow for storage and significant infiltration, they remove both adsorbed and soluble nutrients from runoff. However, the benefit to surface water may become a detriment to groundwater with respect to NO_3. Storage terraces depend on subsurface diversion to remove stored surface water. Storage terraces will reduce sediment loading to local streams compared to unterraced slopes; however, the benefit is reduced when impounded water exceeds the terrace capacity and is diverted to local streams via subsurface tile outlets.

8) **Vegetative filter strips on the contour in cropland and at the lower edge of fields and riparian buffer strips adjacent to streams or lakes should be used to reduce the rate of runoff and increase sedimentation, infiltration, and denitrification.** The composition of vegetative filter strips varies depending on environmental conditions and the intended purpose. The one thing that all vegetative filter strips have in common is movement of runoff water in a direction transverse to the length or across the width. Filter strips should be used in locations where runoff occurs as uniform sheet flow. If runoff occurs as concentrated flow across filter strips, little deposition or infiltration will occur and the purpose is defeated. Concentrated flow that breaches vegetated filter strips becomes a problem as strips become filled with sediment. Utilizing other management practices upslope to control runoff and erosion can extend the life and function of a vegetated filter strip.

   Filter strips composed only of grasses are most effective in removing sediment from runoff compared to nutrients or pesticides. Cool season grasses that are sod forming provide the best filter. The width of grass filter strips separating contoured crops should expand as the slope of the land increases. For land slopes of < 1%, 1-10%, 10-20%, and 20-30%, minimum filter widths of 10 ft, 15 ft, 20 ft, and 25 ft, respectively, are recommended. These strips need to be wide enough to allow 30 minutes of contact time and nonerosive storm flow rates. For widths greater than 50 ft, 6 inch high dikes are required to provide uniform flow of runoff. Grass filter strips on the lower edges of cropped fields should have widths of at least 30 to 45 ft to accommodate turning of farm equipment.

   Effective widths for riparian buffer strips are generally much greater than field filter strips. Buffer strips may be several hundred feet wide. The ratio of field drainage area to filter area should be no greater than 50:1 and preferably within the range of 3:1 to 8:1. When upland soils have high clay contents, the width should be expanded to accommodate the greater distance needed for clay deposition. These types of vegetative strips have greater adsorptive capacity for soluble nutrients and pesticides and often include a woody vegetation component.

   The riparian buffer strip usually consists of 1) undisturbed forest, 2) managed forest, and 3) grass. The minimum recommended widths for these zones are 15 ft, 60 ft, and 20 ft, respectively. The width of the managed forest zone should include all soils in NRCS hydrologic group D and hydrologic group C that are frequently flooded. The managed forest zone should have a width that when combined with the undisturbed forest is at least 1/3 of the distance from the stream-bank or shoreline to the top of the adjacent upland slope. This combined width should also be a minimum of 75 ft for soils in NRCS capability classes I, IIe/s, and V; 100 ft for capability classes IIIe/s, and IVe/s; and 150 ft for capability classes VIe/s and VIIe/s.

   Maintenance of riparian buffer strips includes bank channel stabilization with rip-rap, revetments, weirs, and/or limiting livestock access to water by fencing or herding. The use of grade stabilization structures may help maintain stream-bank integrity by controlling gully erosion. Grazing in riparian areas needs to be closely managed. Dividing pastures into riparian areas and upland areas helps manage livestock to meet the needs of both types of vegetation. Livestock should never graze stream banks during periods of high vulnerability to soil erosion. Longer periods of rest added to the grazing cycle may be required for restoration of severely degraded riparian areas. Livestock should be excluded from the two forest zones and occasionally from the grass zone if no managed forest zone is present. Regular nutrient removal from the grassed zone through managed grazing or haying is required. Grazing and crop management for runoff and erosion control in upland areas should be used as secondary methods for riparian area maintenance.

9) **Sediment control basins may be used under some circumstances to trap sediment moving through minor drainage ways prior to entry into a stream or lake.** This practice will provide surface water protection particularly with respect to N adsorbed to sediment.

10) **Wetlands should be maintained on the landscape to the greatest extent possible due to their natural role as N sinks.** Wetlands function as settling basins and/or filters due to their depressional landscape position and active biologic processes. Through the processes of N uptake and denitrification, wetlands serve to protect surface water resources from excessive NO_3 concentrations. In landscapes that are not well integrated with streams and rivers, some types of wetlands may also serve to protect groundwater if their biologic function of cycling nutrients is maintained.

11) **Rangeland and pastures should be grazed at an intensity that maintains a minimum of 30% ground cover at all times.** Stocking rates that allow 50 to 60% utilization of above-ground biomass will allow sufficient plant growth for adequate cover. A common rule of thumb for proper grazing is "take half - leave half." Native rangeland in North Dakota should be grazed no earlier than June 1, with the exception of the southeastern corner of the state where grazing may begin about May 20. Pastures or rangeland dominated by the introduced grass species of **crested wheat grass** and **smooth bromegrass** should be grazed at the end of April or beginning of May, much earlier than native range. Proper grazing will help reduce soil erosion and protect surface water resources.

12) **Systematic use of rangeland improvements as outlined in NDSU Extension Service Publication R-1028, Water Quality: The Rangeland Component, are recommended to help distribute livestock more uniformly.** Water developments, fencing, burning, mowing, and salt and mineral placement are all practices that when used together make up a grazing system that improves forage efficiency and reduces soil erosion from over-used areas. Other range improvement practices that may be utilized within a given grazing system include 1) weed control, 2) proper fertilization, 3) grass interseeding, and 4) runoff control. These practices will help protect surface water resources from N contamination through erosion control.

Irrigation management

Irrigation has been demonstrated in some areas to have greater potential for groundwater contamination compared to dryland agriculture. Irrigation may increase the potential for groundwater contamination for several reasons. If not managed correctly, overapplication of water can result in substantial leaching through the root zone. Prevention of leaching and maintenance of adequate soil water levels requires a high level of management, particularly with shallow rooted crops on soils with low water-holding capacities (coarse textures). Greater inputs of nutrients are generally required for irrigated crops because of their increased yield potential compared to dryland crops. Montgomery et al. (1988) found that three times as much N was applied over the Oakes aquifer on irrigated fields compared to dryland fields. The concentration of NO_3 in the tile drainage beneath the irrigated fields was approximately three times higher compared to the tile drainage beneath dryland fields.

Irrigation management studies in North Dakota and other states demonstrate that managed inputs reduce the potential for groundwater contamination. Several studies indicate that water and N inputs can be reduced compared to conventional irrigation management without lowering yields (Montgomery et al., 1990; Ayars and Phene, 1993; Watts et al., 1993a; Derby et al., 1994; Knighton and Albus, 1992). The timing of water and nutrient inputs has been shown to be critical with respect to both yields and contaminant movement (Montgomery et al., 1990; Eisenhauer et al., 1993; Martin et al., 1993; Watts et al., 1993a,b).

Another potential source of groundwater contamination under irrigated management is backflow or spillage due to the practice of chemigation. A national survey of irrigators indicated that chemigation was used on 42%, 62%, and 3% of sprinkler, trickle, and furrow irrigated acreages, respectively (Lundstrom, 1988). Eighty percent of the chemigation was for fertilizer application. Irrigators who apply nitrogen through their irrigation systems in North Dakota must comply with Chapter 4-35.1 of the North Dakota Century Code and Article 7-09 of the North Dakota Administrative Code (ND Dept. of Agriculture). Chemigation rules require the installation of a functional check-valve in the water-line, a low pressure sensor, an inspection port, a low-pressure drain, an interlock between the water pump and chemical pump, a proper chemical injection pump, and a pressure operated check-valve in the chemical-injection-line.

The management practices recommended for water resource protection on dryland acreages also apply to irrigated fields. However, the following management recommendations reflect opportunities and needs that are unique to irrigation. Although these practices provide protection from N contamination for both surface water and groundwater, they are particularly important to groundwater.

1) **Schedule irrigation appropriately by monitoring soil water and crop water use.** Regular measurement of soil water is an accurate way of determining when to irrigate. An indirect method used to estimate soil-water balance, commonly called the "checkbook method," is based on knowledge of the soil water holding capacity, daily crop water use, and daily precipitation measurements. Soil water content determined using the checkbook method should be verified occasionally with field measurements.

It is critical that the water budget is determined systematically and accurately so that applications of water meet the needs of the crop but do not result in over-application.

2) **Time water applications to avoid water movement beyond the rooting zone.** Weather patterns should be assessed prior to each irrigation. Deficit irrigation techniques that leave room in the rooting zone for additional water from rainfall have been demonstrated to conserve water without yield reductions. Irrigation should not fill the soil to field capacity and the soil profile should never be used to store irrigation water through the winter. To the contrary, irrigation water should be managed so that stored soil water is at a minimum in the fall.

3) **Adjust water application amounts to meet varying crop demands at different growth stages.** Irrigation has the potential to meet these variable demands more readily than dryland agriculture, thus maintaining a stable environment for plant growth. Large amounts of unused available N are not likely to be left in the soil, if management results in maintenance of vigorous plant growth throughout the year. The potential for NO_3 leaching and groundwater contamination is diminished if this practice is followed.

4) **Irrigation water must be applied uniformly and accurately.** A functional flow meter and accurate pressure gauge, either at the pump or on the pipeline near the point of discharge, are essential for accurate application of irrigation water and N fertilizer. Uniform application rates can only be accomplished if irrigation equipment functions properly; therefore, sprinklers, nozzles, pipes, etc. must be checked regularly. Placing catch cans under the system to measure actual amounts of water delivered to the soil surface can check uniformity of application.

5) **When N is injected into an irrigation system, chemigation equipment that protects the water supply must be used.** State regulations regarding the proper chemigation equipment required to protect the water source from back-siphonage must be followed. Chemigation provides an opportunity to ensure that adequate N is supplied to the crop during critical growth stages. Application of nitrogen through the irrigation system can be accomplished at later growth stages when other methods of delivery are not possible. In addition, applying nitrogen through the irrigation system helps to split applications and avoids applying all the nitrogen in a single application, which has both economic and environmental benefits.

6) **The chemigation unit must be calibrated with each use to ensure accurate application of N.** An accurate way of measuring the amount of chemical being injected into the irrigation system is essential to good irrigation management. Accurate measurement of the amount of applied N not only optimizes chemical usage but also ensures a uniform application over the entire irrigated field if the system is designed and operating correctly.

7) **Secondary containment should be provided where N fertilizer is stored near the irrigation well when chemigation is practiced.** Secondary containment, made of impermeable material, reduces the risk of contamination in the case of a leak or spill.

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APPENDIX I

An Example of Groundwater Assessment for Potential Nitrogen Contamination

Introduction

The following example demonstrates how the factors that influence groundwater sensitivity to nitrogen contamination can be combined to arrive at an overall conclusion for a given area. Dickey County, ND was selected as the area of the following analysis. The selection was based on knowledge of the presence of glacial aquifers in the county (ND State Water Commission Staff, 1986) and availability of the Dickey County Soil Survey in digital format (NRCS Staff, 1999).

One of the critical requirements for each factor was availability or accessibility of data on a statewide basis, because this assessment was intended to be used on any area in North Dakota. All counties in North Dakota have a completed modern NRCS soil survey. Data can be extracted from these documents that determine **soil aeration, soil texture**, and the **hydrologic recharge area.** All counties in North Dakota have a completed State Water Commission groundwater studies report. Data extracted from these documents can determine the **depth to the top of the saturated zone in a given aquifer**. Satellite images of landuse and crops are available from the North Dakota Agricultural Statistics service. Data can be extracted from these images that discriminate between areas of **cultivation, various crops and summer fallow.**

The data required to do a groundwater assessment for sensitivity to nitrogen contamination is available for any area of North Dakota. However, manipulation of the data, particularly on extensive areas, is tremendously cumbersome and time-consuming if done without the aid of a computer. In the following example the GIS computer program ARC VIEW 3.1 (ESRI, 1998) was used to process the Dickey County data for assessment. A PC with 233 MHz micro processor, 64 MB RAM, and 2 GB harddrive was used to do the analysis.

The combination of factors listed in Tables 1 and 2 of chapter 2 can be accomplished in many ways. In other words, factors may be weighted to account for their importance or dominance with respect to the natural processes that affect water and solute flow. Assignment of factor weights should be done with extreme care and should be supported by a significant amount of evidence, preferably experimental results. In general, the following method does not assign weights to the factors due to the lack of scientific evidence that would validate such assignment. Instead, groundwater sensitivity is related to the intensity in which each factor manifests itself and the additive intensity of all the factors together.

Analysis of the Natural Factors (Vulnerability)

A project for analysis of Dickey County using the ARC VIEW program was opened. Soils data from the NRCS Soil Survey Geographic (SSURGO) data base were extracted for Dickey County. Tabular data is a component of the digital soil survey and occurs in several tables. To extract the data and assign it to specific delineations of soil in Dickey County, it was necessary to link several tables. When working with SSURGO data it is important to understand that many soil mapping units have more than one named component, some tables contain information for each component, and some contain information only for the soil mapping unit. It is important to understand this concept when linking soil data tables, or the data displayed may represent something other than what was intended. For the analysis of Dickey County, soil factors were assessed for all components of each mapping unit. In other words, soil components of less extent carry the same importance as those of greater extent with respect to groundwater sensitivity assessment.

After the appropriate tables were linked, ARC VIEW was used to query the tables for the data that represented vulnerable conditions. For the **soil aeration** factor, all soils with drainage classes of excessive, somewhat excessive, and well were selected for Dickey County (Fig.1). For the **soil texture** factor, all soils with single particle size classes of sandy, sandy skeletal, and fragmental and soils with dual particle size classes with the lower material being one of these three classes were selected for Dickey County (Fig. 2). For the **hydrologic recharge** factor, the depth to $CaCO_3$ was determined from the typical pedon description for each soil and added to the one of the tables. All soils with typical depths to $CaCO_3$ >30 inches were selected for Dickey County (Fig. 3).

The **depth to top of the saturated aquifer** was determined using data from the State Water Commission for Dickey County. Well location and depth to saturation were available in digital format. Depth to the top of the aquifer was extracted from the well logs in the Dickey County groundwater resources report and added to the digital data table. ARC VIEW was used to convert the digital data set with point locations to a grid and then to interpolate both sets of depth values. Both interpolated grids were reclassified into classes of 0-50 ft and >50 ft. A function in ARC VIEW called map calculator was used to then add the two grids which resulted in areas where both the aquifer depth and depth to saturation were 0-50 ft (Fig. 4).

The data files for **soil aeration, soil texture,** and **hydrologic recharge** were converted to grid files within the ARC VIEW project and added to the **depth to top of the saturated aquifer** using the map calculator. The result shows vulnerability of groundwater in Dickey County as five classes, **High , High Intermediate, Intermediate, Low Intermediate,** and **Low** (Fig. 5). **High** category represents areas where all four of the factors would have potential to contribute to nitrogen contamination. The **Low** category represents areas where none of the factors have potential to contribute to nitrogen contamination of groundwater. The categories **Low Intermediate, Intermediate,** and **High Intermediate** correspond to areas that would have one, two, and three factors, respectively, that would have potential to contribute to nitrogen contamination. ARC VIEW can then be used to produce information on any point within Dickey County that tells which of the four factors contribute to nitrogen vulnerability.

Analysis of the Anthropogenic Factors

ARC VIEW was used to import the digital boundary data as provided by the North Dakota Department of Transportation (NDDOT) for Dickey County. Farms and other cultural features occurred as point data that were converted to a grid format. Using the neighborhood statistics routine, areas of about 400 ft X 400 ft were delineated around these points (Fig. 6). The area was assumed to be an average area of influence from human activity around farms and other cultural points. Boundaries for urban areas were also imported from the NDDOT data set and converted to a grid format (Fig. 6).

ARC VIEW was use to import an image of landuse in North Dakota for 1998 produced by the Spatial Analysis Research Section, USDA, NASS, RD. The image was converted to a grid and clipped using the Dickey County outline provided by the NDDOT. Compared to the natural factors, many of the anthropogenic factors are subject to considerable change over time. This is particularly true for cropping and other landuse patterns. Consequently, landuse information from one season was modified using the ARC VIEW neighborhood statistics function to provide an estimate of landuse probability. Mean values were calculated for areas of 1/2 mile radius for summer fallow, cultivation, and potatoes plus corn. These values were then used to classify areas in Dickey County as low, interediate, or high probability for these types of landuse (Figs. 7, 8, and 9).

The probability of irrigation was determined by using the digital data set of irrigation diversion locations in Dickey County provided by the State Water Commission. Generally the areas of irrigation are within 1 mile of the point of diversion. Using the neighborhood statistics function in ARC VIEW, mean values were assigned to areas 1/2 mile in radius for points of diversion. These values were then used to classify areas in Dickey County as low, intermediate, or high probability for irrigation (Fig. 10). The ARC VIEW map calculator was used to sum the results of the anthropogenic factor analyses to determine an overall potential for nitrogen contamination of groundwater based on landuse for Dickey County (Fig. 11).

Analysis for Groundwater and Aquifer Sensitivity

The ARC VIEW map calculator was used to sum the results of the vulnerability analysis and the anthropogenic potential analysis to determine groundwater sensitivity to nitrogen contamination (Fig. 12). Finally, digital data outlining aquifer boundaries in Dickey County obtained from State Water Commission were used to clip those areas that overlay the aquifers to determine aquifer sensitivity (Fig. 13). It should be noted that greater confidence is placed on aquifer sensitivity as opposed to groundwater sensitivity, because well data from the State Water Commission is concentrated in areas with aquifers. Consequently, groundwater sensitivity for areas between aquifers is based on interpolation between points separated by large distances compared to aquifer sensitivity.

Studying the aquifer sensitivity map for Dickey County reveals that the areas of greatest concern are in the eastern part of the county. Using the ARC VIEW zoom function, aquifer sensitivity of smaller areas can be determined, such as near the town of Oakes, ND, where there is relatively high potential for aquifer contamination (Fig. 14). By zooming to an even smaller area, such as NW 1/4, Sec. 27, T.131 N., R. 59 W. about 1 mile southeast of Oakes, and using the ARC VIEW information, the factors that contribute to the sensitivity in specific fields can be determined. However, the information function only provides data about a specific point and does not reveal the areal extent of specific conditions that have practical management implications. To address management issues, the coverage for specific factors must be combined with the aquifer sensitivity coverage. For example, in the NW 1/4 of Sec. 27 the combined coverages of vulnerability factors and aquifer sensitivity (Fig. 15) reveal that the largest area has High Intermediate sensitivity and the contributing vulnerability factors are a shallow depth to water and groundwater recharge. When the coverage of anthropogenic factors is considered (Fig. 16), cultivation is shown to be the primary contributing factor to the main area of High Intermediate sensitivity.

If the NW 1/4 of Sec. 27 is managed as one unit, groundwater protection efforts should be based on the factors of greatest areal extent as outlined above. Reduction in aquifer contamination potential can be accomplished by using management methods listed in chapter 3 that are related to each of the cited factors. Soil testing and proper fertilizer management would be the primary recommendation for aquifer protection in the NW 1/4 of Sec. 27. If manure is used as a source of nitrogen, it should also be tested and applied according to the practices recommended in chapter 3. Of the SWCP listed in chapter 3 the primary recommendation for this area would be including nitrogen and water scavenging crops such as sunflower, safflower, or alfalfa in the rotation. The anthropogenic coverage (Fig. 16) shows that three possible locations of concentrated human activity occur in the NW 1/4 of Sec. 27. The extent of activity in these areas should be determined and appropriate farmstead / household management practices, such as sealing abandoned wells, can then be applied.

Review of the vulnerability and anthropogenic factor coverage (Figs. 15 and 16) shows the eastern area adjacent to the NW 1/4 of Sec. 27. In contrast to the NW 1/4, the aquifer sensitivity category of greatest extent is High in this area. Management for aquifer protection from nitrogen contamination in this area assumes greater importance compared to the NW 1/4. The factor coverages show that additional vulnerability factors, soil texture and aeration, plus greater probability of corn and potatoes contribute to higher contamination potential in the area east of the NW 1/4.

Compared to the NW 1/4, management priorities would be different in the area to the east. The primary recommendation would be strict avoidance of fertilizer applications in the fall. Split applications of nitrogen fertilizer would be recommended along with use of a nitrification inhibitor or a slow-release product. The use of nitrogen and water scavenging crops in the rotation also plays a more important role in this area compared to the NW 1/4 due to the likelihood of corn and/or potatoes in the rotation. Other recommendations as mentioned for the NW 1/4, such as soil and manure testing, would also be important in the area to the east.

The information and recommendations provided from the foregoing analysis must be tempered with reality. The recommendations are quite general; each site will require more precise recommendations based on agronomic and economic principles that are beyond the scope of this

It must also be recognized that categories represented in this analysis rely on a set of assumptions and probabilities that are subject to error. For instance, in the bottom part of Figures 15 and 16, a small finger-like area of high sensitivity occurs that has a high probability of irrigation. Due to its shape and size, the likelihood that this area does not exist is quite high. The area in question is most likely an aberration of the statistical method. It is also possible that corn and potatoes may be used regularly in the rotation on the NW 1/4 of Sec. 27, or this site may be irrigated. If either one or both of these are true, it would change the priority and type of management recommendations. At the very least, the presence of irrigation would require that the producer consider the use of irrigation practices outlined in chapter 3. Each analysis must be rectified with actual conditions at the site.

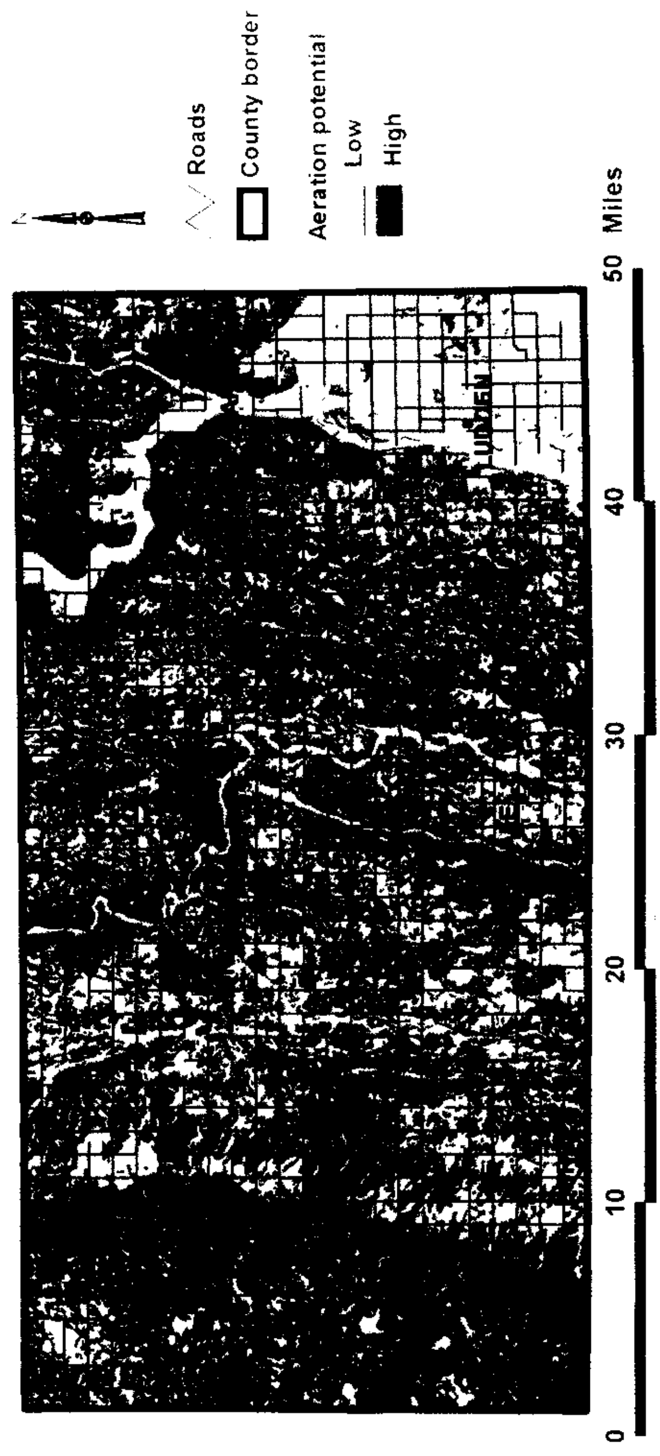


Figure 1. **Soil aeration potential to contribute to nitrogen contamination of groundwater in Dickey County.**

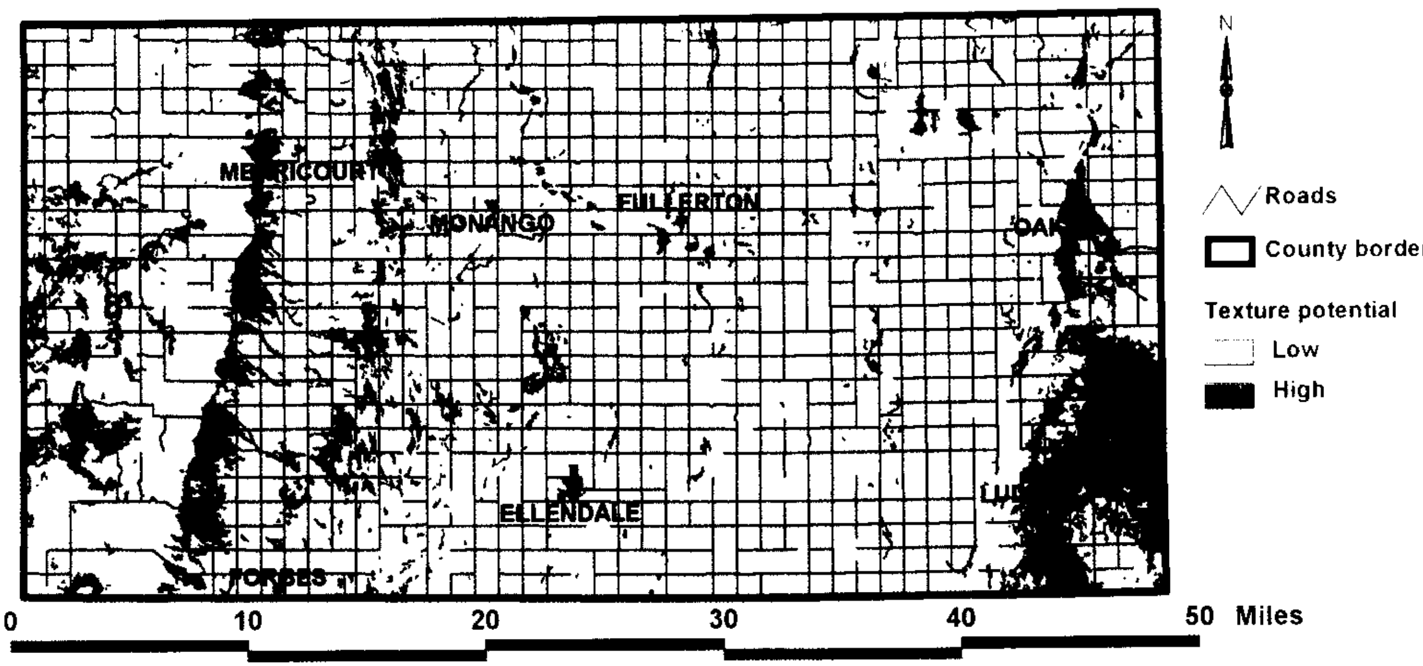


Figure 2. Soil texture potential to contribute to nitrogen contamination of groundwater in Dickey County.

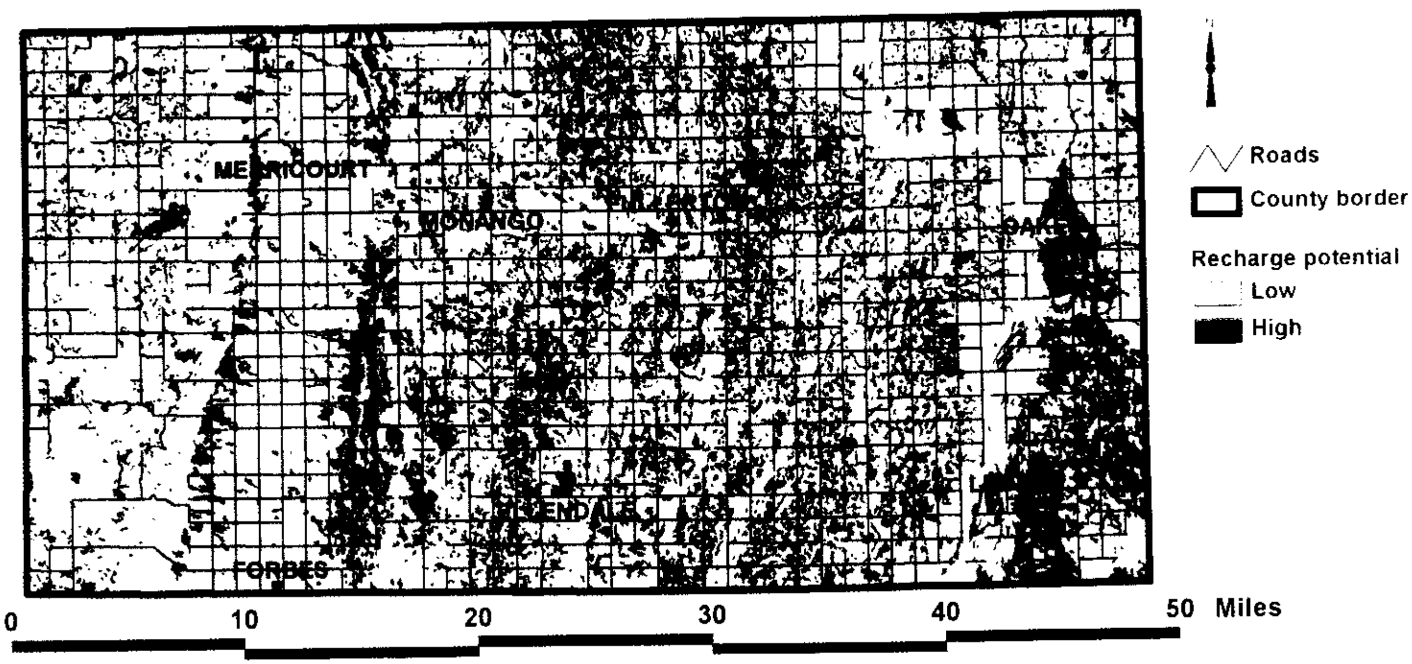


Figure 3. Groundwater recharge potential to contribute to nitrogen contamination of groundwater in Dickey County.

Figure 4. Water depth potential to contribute to nitrogen contamination of groundwater in Dickey County.

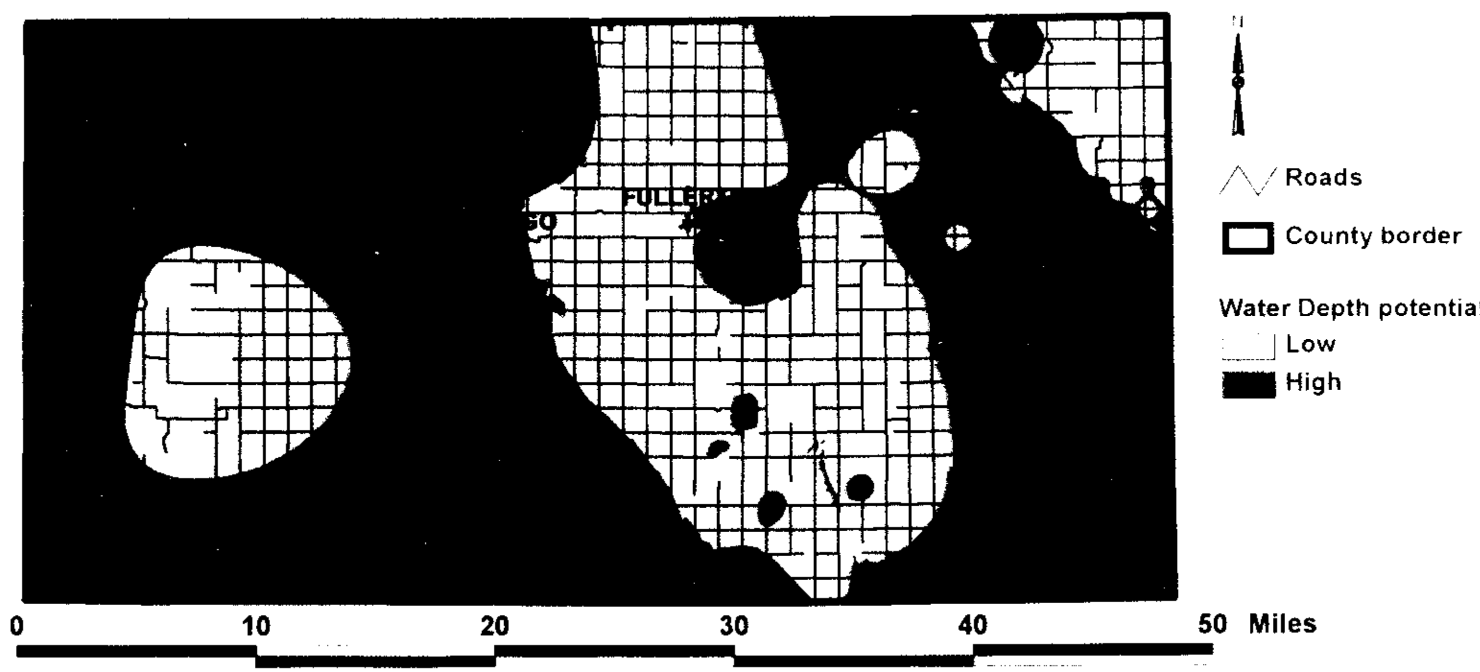
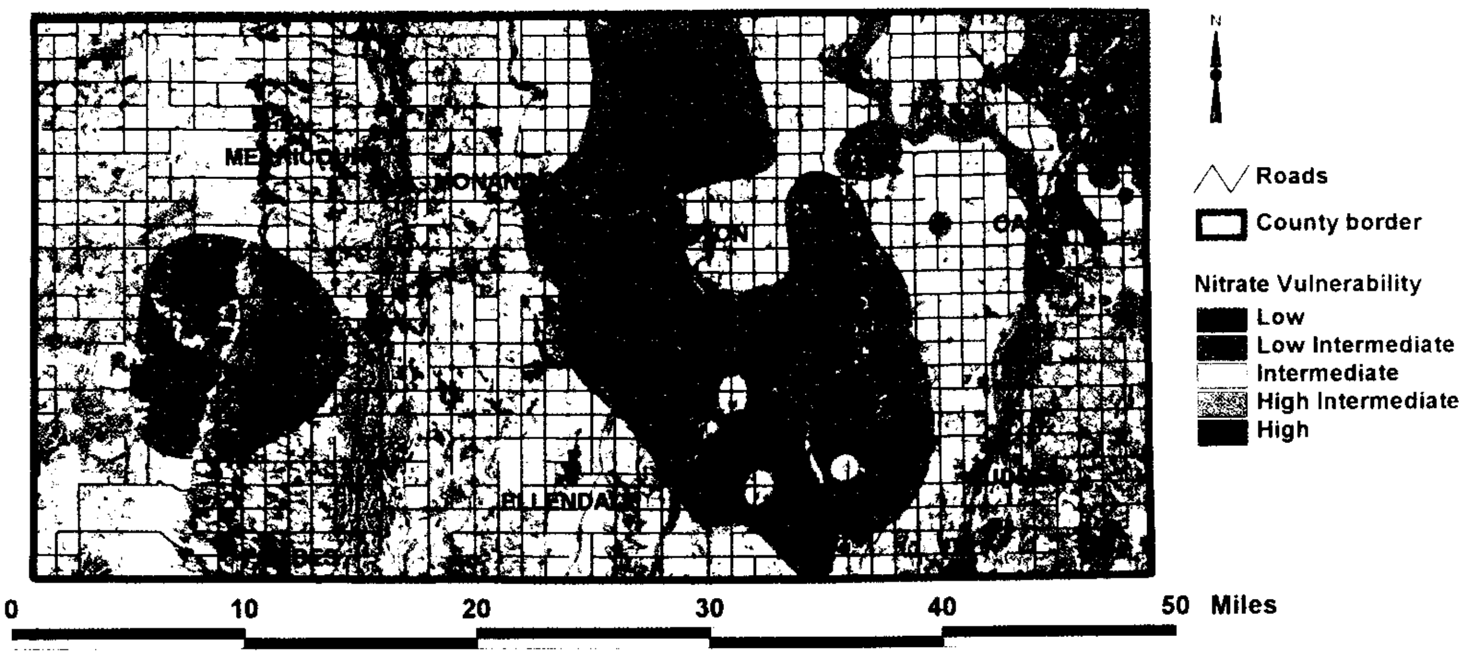


Figure 5. Groundwater vulnerability to nitrogen contamination in Dickey County.



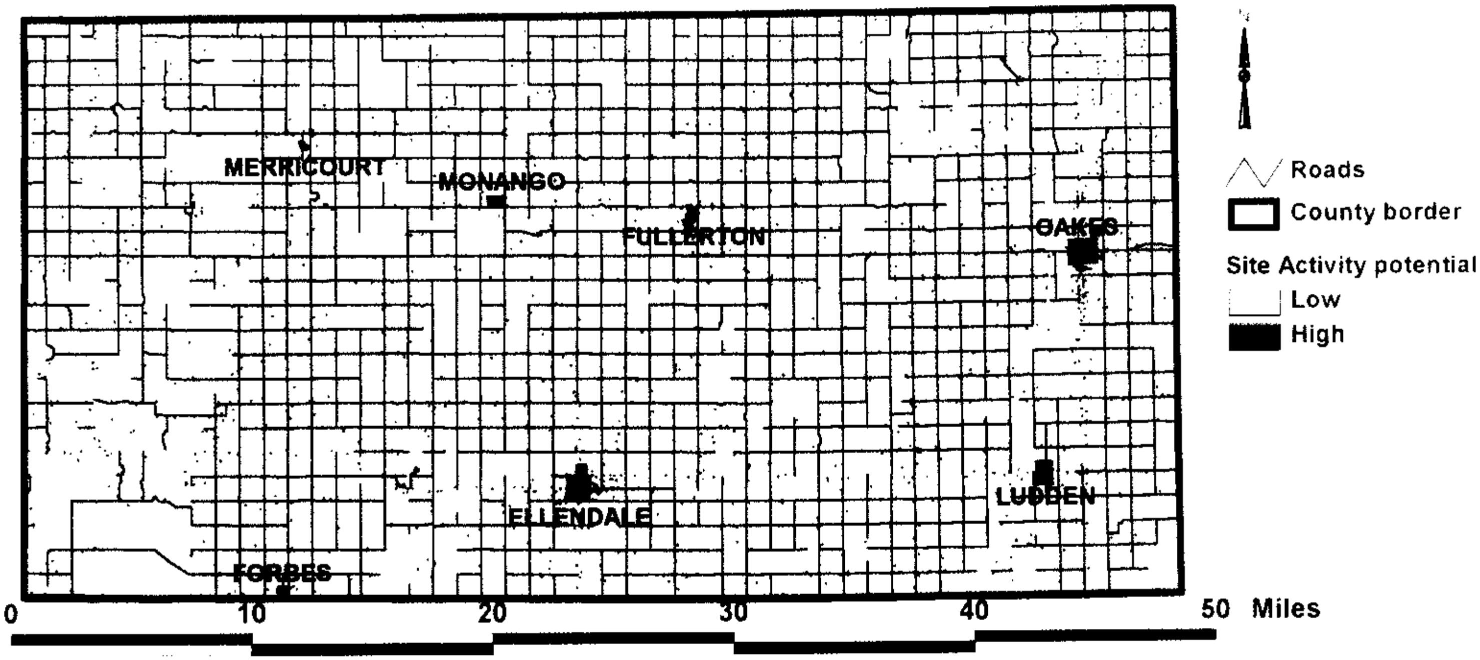


Figure 6. Urban and rural activity site potential to contribute to nitrogen contamination of groundwater in Dickey County.

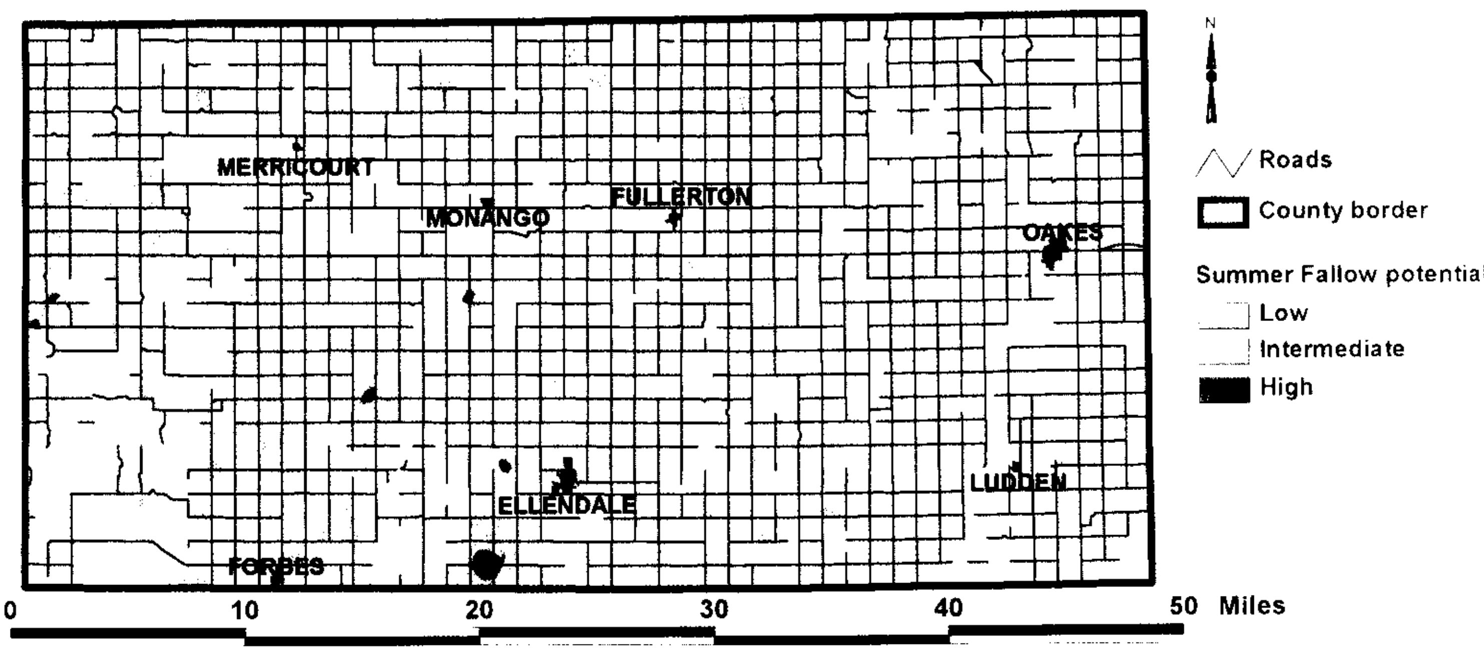


Figure 7. Summer fallow potential to contribute to nitrogen contamination of groundwater in Dickey County.

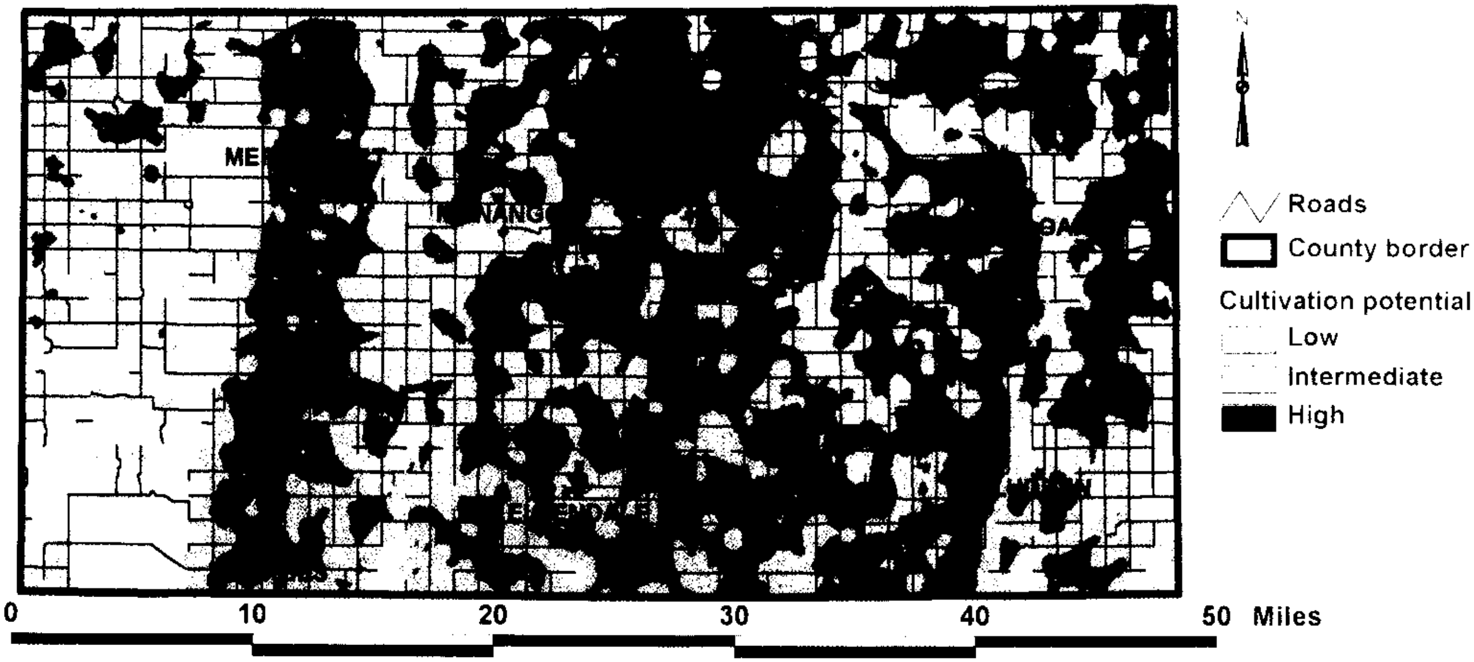


Figure 8. Cultivation potential to contribute to nitrogen contamination of groundwater in Dickey County.

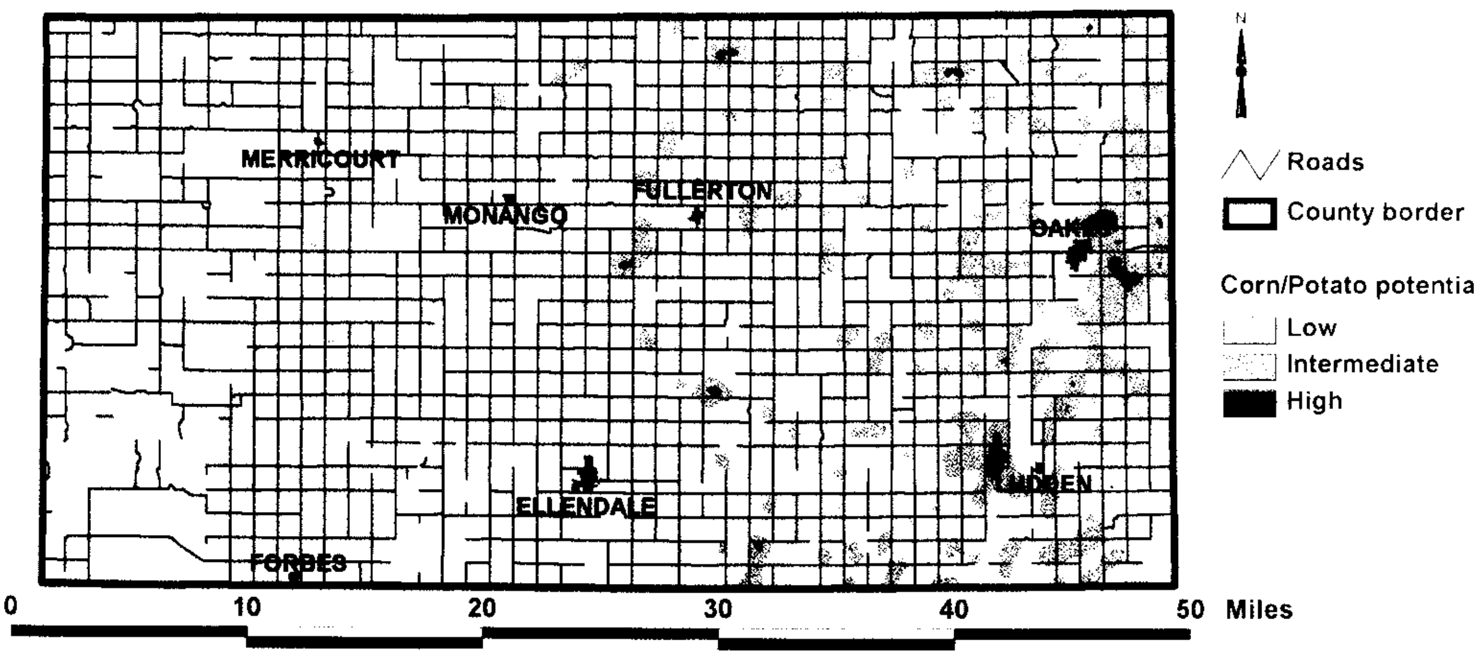


Figure 9. Corn/potato potential to contribute to nitrogen contamination of groundwater in Dickey County.

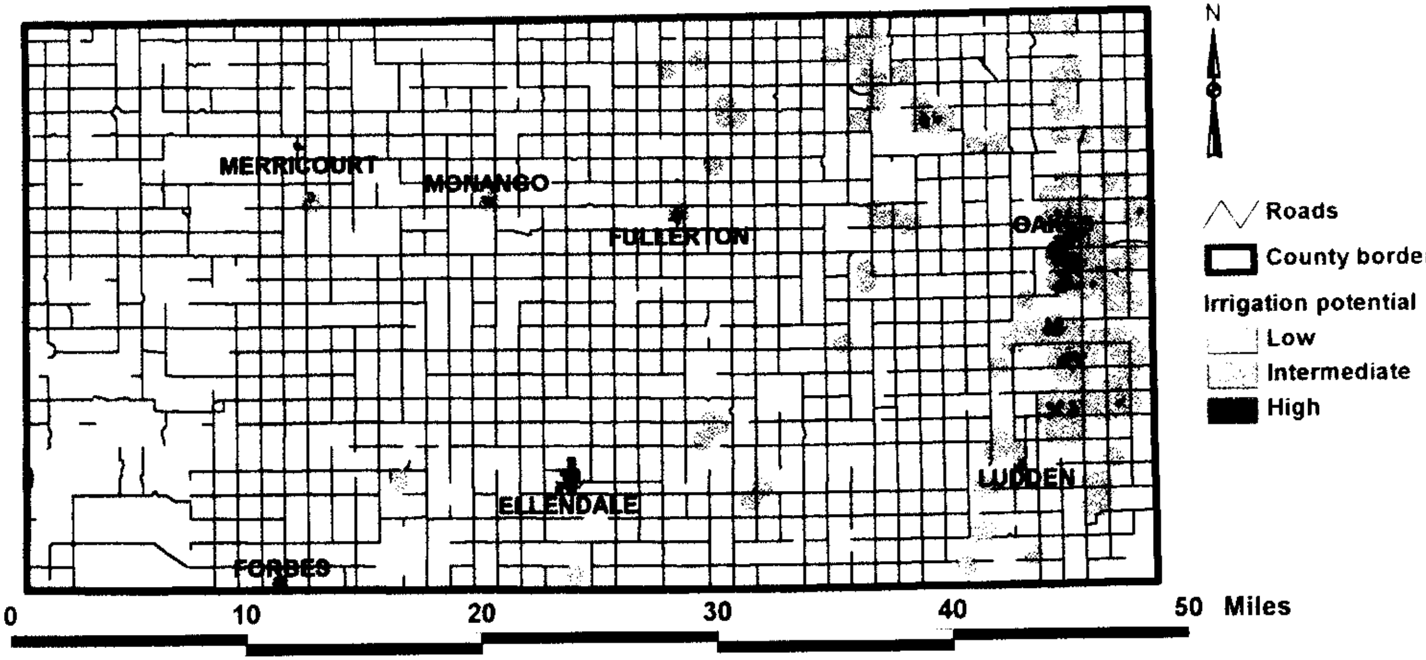


Figure 10. Irrigation potential to contribute to nitrogen contamination of groundwater in Dickey County.

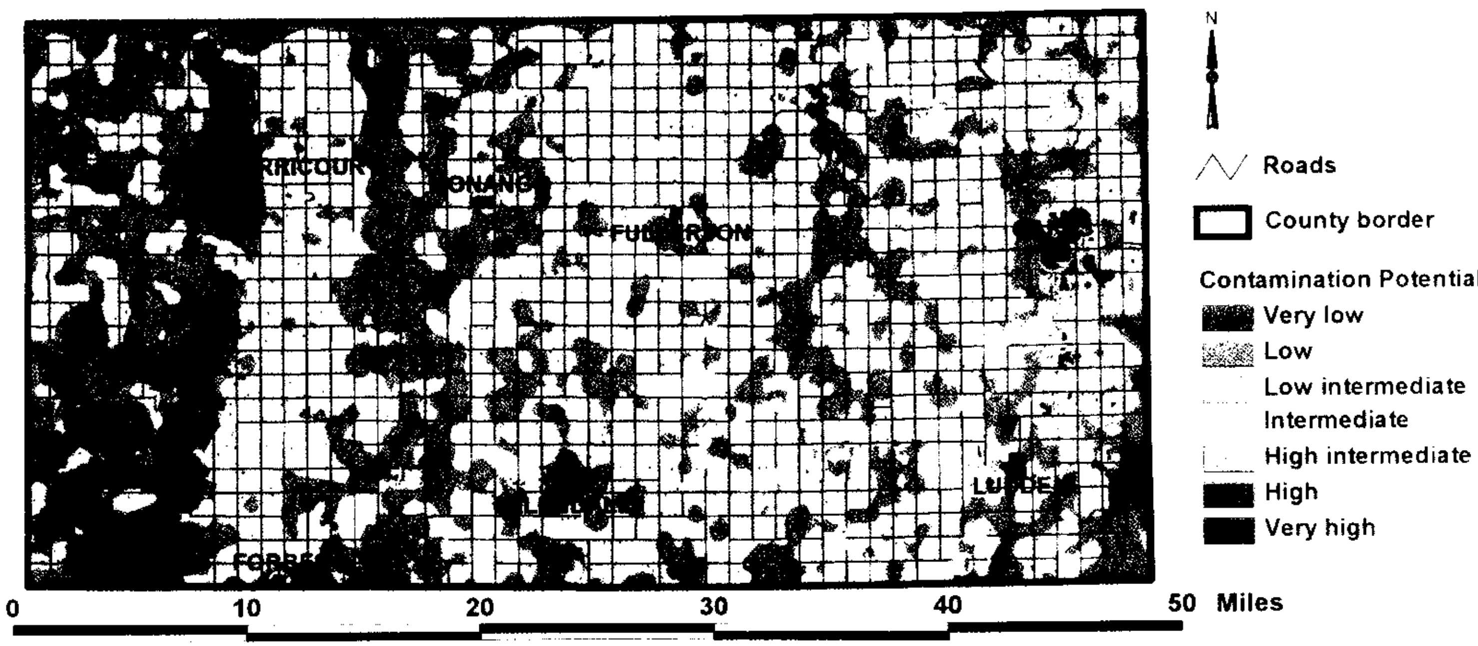


Figure 11. Potential nitrogen contamination of groundwater based on land use in Dickey County.

Figure 12. Groundwater sensitivity to nitrogen contamination in Dickey County.

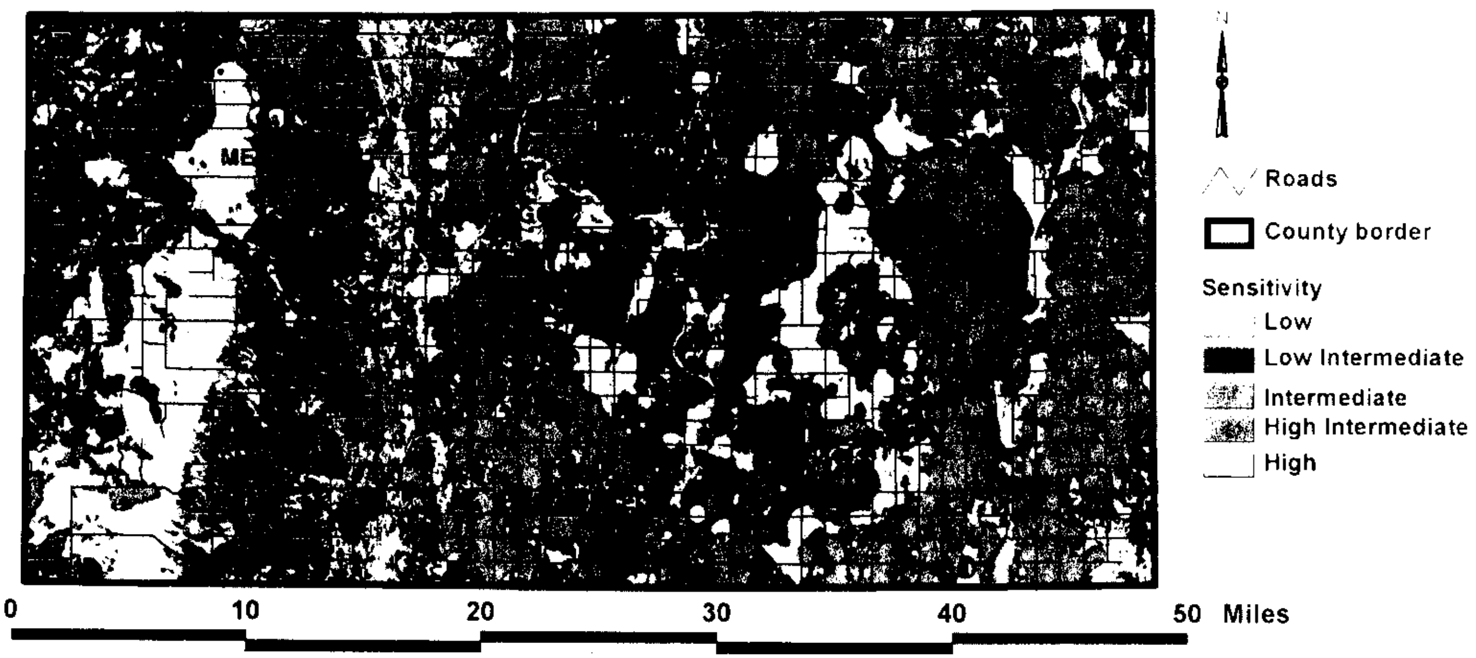


Figure 13. Aquifer sensitivity to nitrogen contamination in Dickey County.

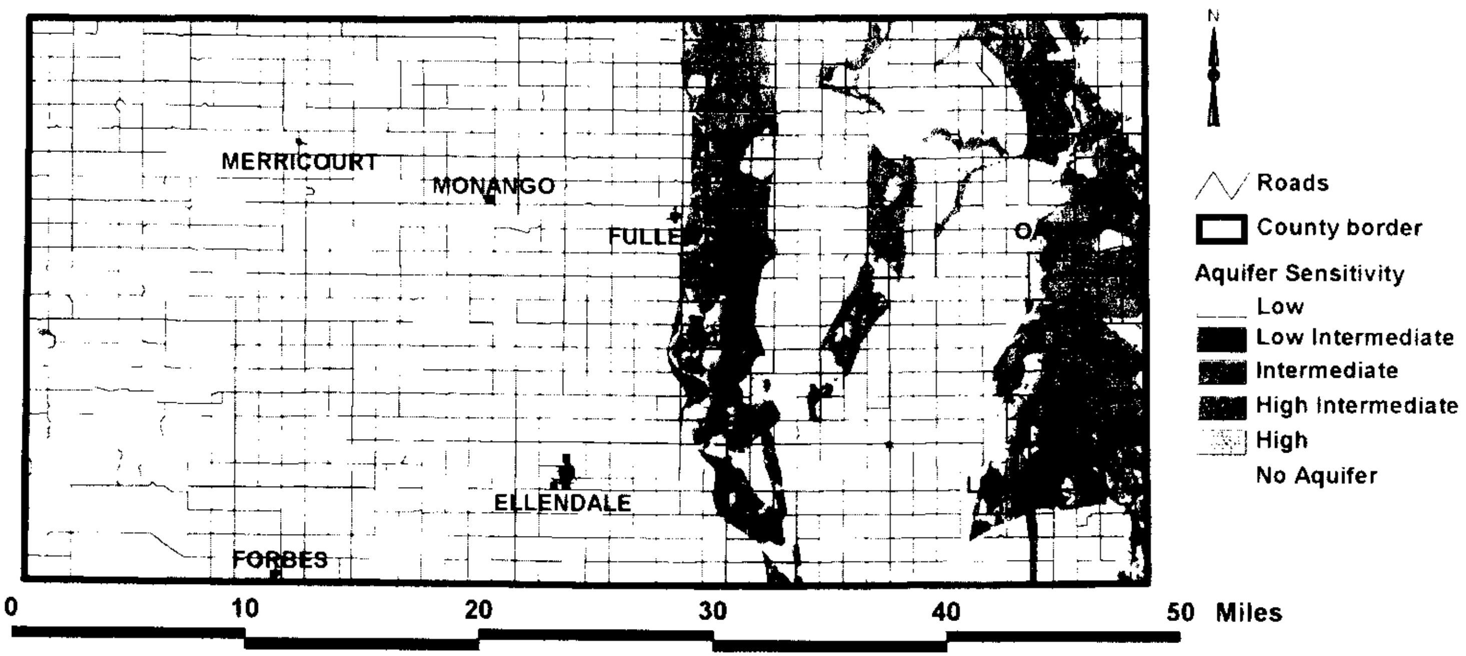


Figure 14. Aquifer sensitivity to nitrogen contamination near Oakes.

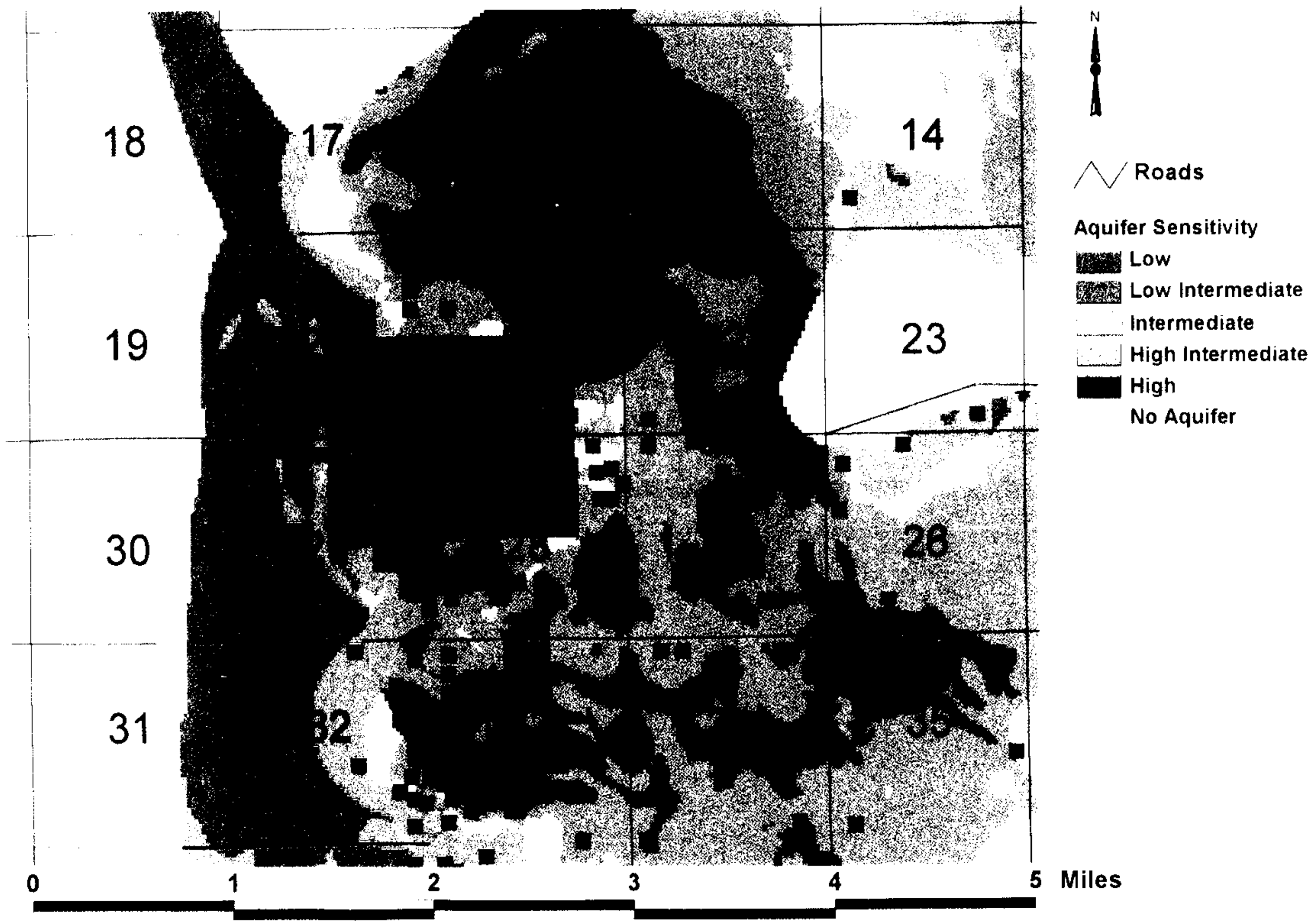


Figure 15. Vulnerability factors for the NW ¼, Sec. 27, T.131 N., R.59 W. near Oakes.

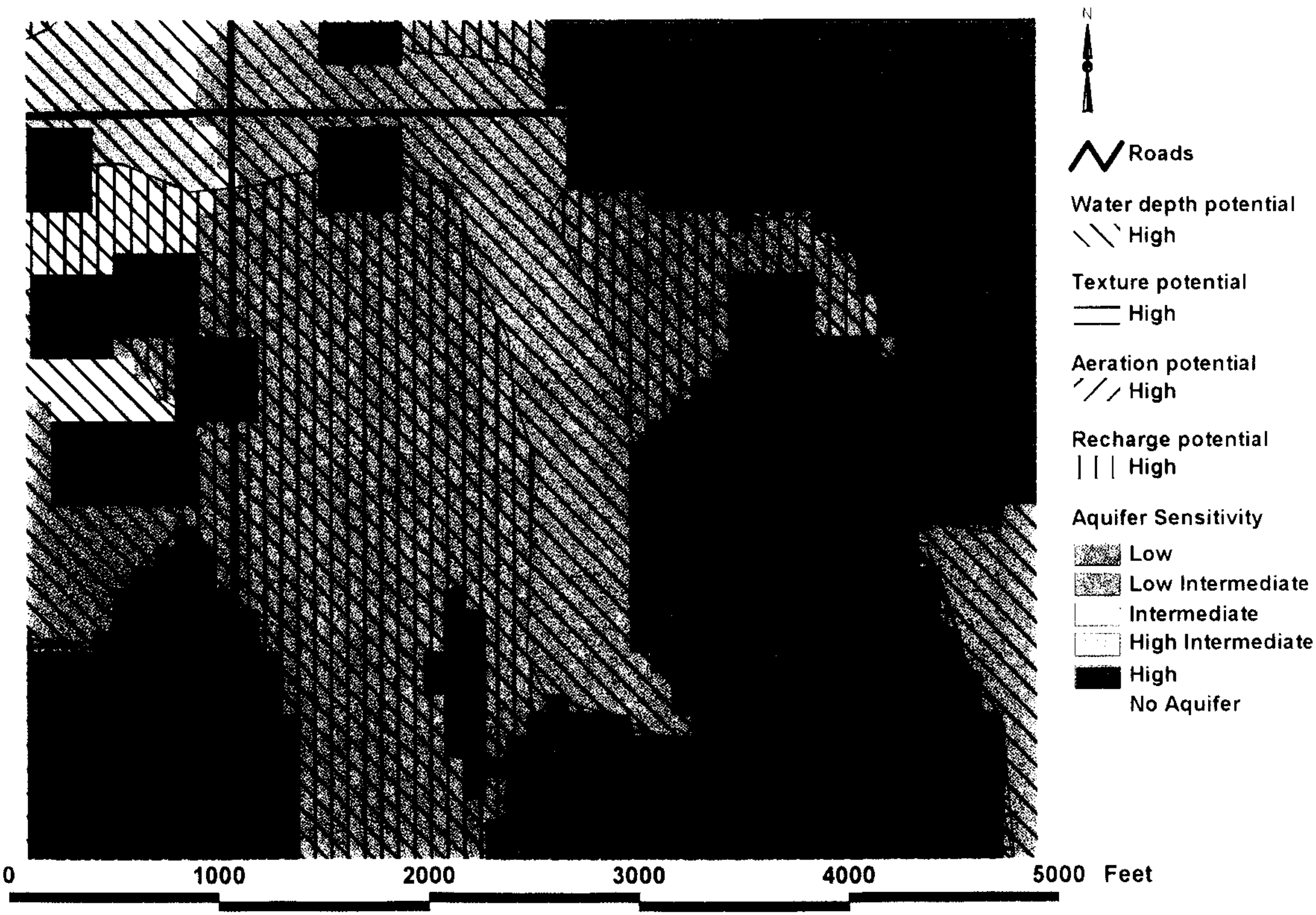
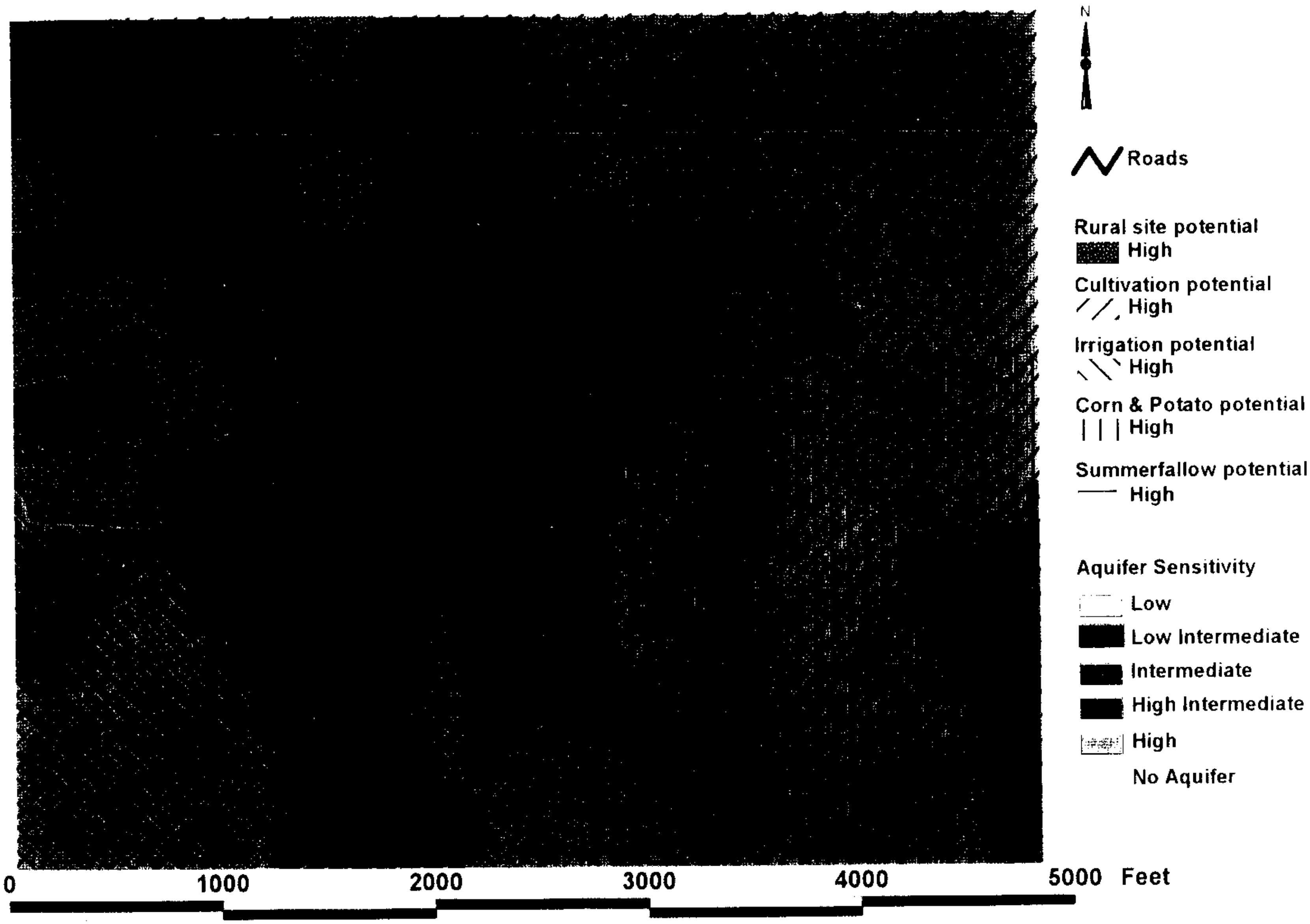


Figure 16. Anthropogenic factors for the NW ¼, Sec. 27, T.131 N., R.59 W. near Oakes.



APPENDIX II

An Example of Surface Water Assessment for Potential Nitrogen Contamination

Introduction

The following example demonstrates how the factors that influence surface water sensitivity to nitrogen contamination can be combined to arrive at an overall conclusion for a given area. The Monango NW 7.5 min. Quadrangle in Dickey County, ND was selected as the area of the following analysis. The selection was based on availability of the Dickey County Soil Survey in digital format (NRCS Staff, 1999) and the digital elevation model (DEM) data for the chosen quadrangle (USGS, 1999).

One of the critical requirements for each factor was availability or accessibility of data on a statewide basis, because this assessment was intended to be used on any area in North Dakota. Many counties in North Dakota have large areas with completed USGS topographic surveys that can help assess the **proximity** to surface water resources. All counties in North Dakota have a completed modern NRCS soil survey. Data can be extracted from these documents that determine **soil aeration, solution translocation**, **sediment translocation, land slope, and flooding frequency**. Satellite images of landuse and crops are available from the North Dakota Agricultural Statistics service. Data can be extracted from these images that discriminate between areas of **summer fallow, cultivation, and rangeland and forestland.**

The data required to do a surface water assessment for sensitivity to nitrogen contamination is available for most areas of North Dakota. However, manipulation of the data, particularly on extensive areas, is tremendously cumbersome and time-consuming if done without the aid of a computer. In the following example the GIS computer program, ARC VIEW 3.1 (ESRI, 1998) was used to process the Monango NW Quadrangle data for assessment. A PC with 233 MHz micro processor, 64 MB RAM, and 6 GB harddrive was used to do the analysis.

The combination of factors listed in Tables 3 and 4 of chapter 2 can be accomplished in many ways. Factors may be weighted to account for their importance or dominance with respect to the natural processes that affect translocation of contaminants to surface water resources. Assignment of factor weights should be done with extreme care and should be supported by a significant amount of evidence, preferably experimental results. In general, the following method does not assign weights to the factors due to the lack of scientific evidence that would validate such assignment. Instead, surface water sensitivity is related to the intensity in which each factor manifests itself and the additive intensity of all the factors together.

Analysis of the Natural Factors (Vulnerability)

A project for analysis of the Monango NW Quadrangle using the ARC VIEW program was opened. The topographic data (DEM) for this quadrangle was downloaded from the USGS GEODATA web site. Soils data from the NRCS Soil Survey Geographic (SSURGO) data base were extracted for Dickey County. Location of streams, lakes, and cultural features such as section lines in the Monango NW Quadrangle were extracted from ND Dept. of Transportation GIS database for Dickey County

The **proximity** factor was determined by first placing a 500 ft buffer around all streams and lakes using the ND DOT data. Then the elevation data from the Monango NW Quadrangle (Fig. 1) was used to determine probability of surface water movement from a given point on the landscape to given points along a stream. The location of the 500 ft buffer and the probability of surface water movement were combined as grid files to give three categories of potential nitrogen contamination (high, intermediate, and low) related to proximity (Fig. 2).

Tabular data is a component of the digital soil survey and occurs in several tables. It was necessary to link several tables to extract the data and assign it to specific delineations of soil in the Monango NW Quadrangle. After the appropriate tables were linked, ARC VIEW was used to query the tables for the data that represented vulnerable conditions. When working with SSURGO data many soil mapping units have more than one named component, some tables contain information for each component, and some contain information only for the soil mapping unit. It is important when linking soil data tables to understand this concept, or the data displayed may represent something other than what was intended. For the analysis of the Monango NW Quadrangle, soil factors were assessed for all components of each mapping unit. Soil components of less extent carry the same importance as those of greater extent with respect to the assessment of surface water sensitivity.

For the **soil aeration** factor, soils with drainage classes of excessive, somewhat excessive, and well were considered as high potential (Fig.3). Moderately well and somewhat poor drainage classes were considered as intermediate potential. Poor and very poor drainage classes were considered as low potential.

For the **translocation via solution** factor, soils in the hydrologic groups C and D were considered as high potential (Fig. 4). Hydrologic group B was considered as intermediate potential. Hydrologic group A was considered as low potential.

For the **translocation via sediment** factor both the hydrologic group and K factor are considered together to determine potential (Table 1). Figure 5 shows the sediment translocation potential in the Monango NW Quadrangle. It should be noted that for the soils of this area the sediment translocation (Fig. 5) and solution translocation (Fig. 4) potentials are exactly the same. This seems somewhat unrealistic since these are two very different translocation modes. The algorithm for sediment translocation (Table 1) for North Dakota soils probably requires some modification. However, this was beyond the scope of the present example and the algorithm as proposed by Goss (1992) was used despite the questionable results.

For the **land slope** factor, soils with slopes > 15% were considered as high potential (Fig. 6). Soils with slopes < 9% were considered as low potential. Soils with slopes > 9% and < 15% were considered as intermediate potential.

For the **flooding** factor, soils that are frequently flooded were considered as high potential (Fig. 7). Soils that are occasionally flooded were considered as intermediate potential. Soils that are never or rarely flooded were considered as low potential.

Table 1. Potential for N losses via the sediment phase of transport.

Potential	Criteria
high	Hydro. group C and K factor ≥ 0.21 *or* Hydro. Group D and K factor ≥ 0.10
intermediate	All soils that do not meet either the HIGH *or* LOW displacement criteria
low	Hydro. group A *or* Hydro. group B and K factor ≤ 0.10 *or* Hydro. group C and K factor ≤ 0.07 *or* Hydro. group D and K factor ≤ 0.05

The data files for **soil aeration, translocation via solution, translocation via sediment, landslope,** and **flooding recharge** were converted to grid files within the ARC VIEW project and added to the **proximity** grid using the map calculator. The result shows vulnerability of surface water in the Monango NW Quadrangle as five classes, **High, High Intermediate, Intermediate, Low Intermediate,** and **Low** (Fig. 5). The **High** category represents areas where most of the factors would have high potential to contribute to nitrogen contamination of surface water. The **Low** category represents areas where few of the factors have high potential to contribute to nitrogen contamination of surface water. ARC VIEW can then be used to produce information on any point within the Monango NW Quadrangle that tells which of the seven factors contribute to nitrogen vulnerability.

Analysis of the Anthropogenic Factors

ARC VIEW was used to import the digital boundary data as provided by the North Dakota Department of Transportation (NDDOT) for the Monango NW Quadrangle. Farms and other cultural features occurred as point data that were converted to a grid format. Using the neighborhood statistics routine, areas of about 400 ft X 400 ft were delineated around these points (Fig. 9). The area was assumed to be an average area of influence from human activity around farms and other cultural points. Boundaries for urban areas were also imported from the NDDOT data set, but none of these areas occurred in the Monango NW Quadrangle.

ARC VIEW was used to import an image of landuse in North Dakota for 1998 produced by the Spatial Analysis Research Section, USDA, NASS, RD. The image was converted to a grid and clipped using the Monango NW Quadrangle shape created in ARC VIEW. Compared to the natural factors, many of the anthropogenic factors are subject to considerable change over time. This is particularly true for cropping and other landuse patterns. Consequently, landuse information from one season was modified using the ARC VIEW neighborhood statistics function to provide an estimate of landuse probability. Mean values were calculated for areas of ½ mile radius for summer fallow and cultivation. These values were then used to classify areas in the Monango NW Quadrangle as high, intermediate, or low probability for these types of landuse (Figs. 10 and 11). The probability classification was equated to potential for surface water contamination with nitrogen.

The ARC VIEW map calculator was used to sum the results of the anthropogenic factor analyses to determine an overall potential for nitrogen contamination of surface water based on landuse for the Monango NW Quadrangle (Fig. 12).

Analysis for Surface Water Sensitivity

The ARC VIEW map calculator was used to sum the results of the vulnerability analysis and the anthropogenic potential analysis to determine surface water sensitivity to nitrogen contamination (Fig. 13). Using the ARC VIEW zoom function, surface water sensitivity of smaller areas can be determined, such as in Section 23, T.131 N., R. 64 W. near the town of Monango, North Dakota (not shown) where surface water sensitivity to nitrogen contamination ranges from Low Intermediate to High (Figs. 14 and 15). The factors that contribute to the sensitivity in specific fields can be determined in ARC VIEW by using the information function. However, this only provides data about a specific point and does not reveal the areal extent of specific conditions that have practical management implications. To address management issues, the coverage for specific factors must be combined with the aquifer sensitivity coverage. For example, in Sec. 23, the combined coverages of vulnerability factors and surface water sensitivity (Fig. 14) reveals that the largest area of High sensitivity was due to high potential of a combination of vulnerability factors, **transport** (solution and sediment combined because they were the same), **proximity**, and **aeration**. However, the largest area of high intermediate sensitivity was due to a high potential of a single vulnerability factor, **transport**. When the coverage of anthropogenic factors was considered with the surface water sensitivity in Section 23 (Fig. 15), only a relatively small area on the east and south edge appeared to be influenced by these factors. The rural site factor was a primary influence on surface water sensitivity in a few areas of minor extent, but of major interest due to an adjacent stream.

Reduction in surface water contamination potential can be accomplished by using management methods listed in chapter 3 that are related to each of the cited factors. Of primary concern in Section 23 is the stream in the NW ¼ surrounded by high sensitivity. One of the most effective SWCPs for stream protection would be maintenance of a functioning riparian buffer.

It appears that greatest area of Section 23 is rangeland or pasture, because it is not cultivated. Range management techniques that reduce soil erosion would help reduce nitrogen translocation to surface water. Control of soil erosion from rangeland and pasture would be beneficial for surface water protection throughout most of Section 23. Of the SWCP listed in chapter 3, the primary recommendation for this area would be grazing at an intensity that maintains a minimum of 30 percent ground cover at all times. Using methods of rangeland improvement, such as water pipelines, to help distribute livestock more uniformly would also be important to reducing translocation of nitrogen to surface water. The anthropogenic coverage (Fig. 15) shows that three possible locations of concentrated human activity occur in Section 23, and are critical to protection of the nearby stream. The extent of activity in these areas should be determined and the appropriate farmstead / household management practices, such as routing surface water around cattle yards, can then be applied.

Only along the east-south edge of Section 23, does it appear that the soils are cultivated (Fig. 15). The high potential for translocation of nitrogen in this area (Fig. 14) indicates that soil management techniques designed to reduce erosion should be a primary consideration. Of the SWCP listed in chapter 3, conservation tillage should be considered as an important option. Soil testing for nitrogen and application of fertilizer according to the test results would also serve to reduce nitrogen translocation to surface water. Manure used as a source of nitrogen should also be tested and applied according to the practices recommended in chapter 3. It also appears that summer fallow is used as a management tool to some extent in this area. A reduction in the use of summer fallow in the rotation would also have benefits to surface water, but probably not to the extent of the previously mentioned management options

The information and recommendations provided from the foregoing analysis must be tempered with reality. The recommendations are quite general; each site will require more precise recommendations based on agronomic and economic principles that are beyond the scope of this report. It must also be recognized that categories represented in this analysis rely on a set of assumptions and probabilities that are subject to error. In addition, results may appear nonsensical when all the conditions used to display information are not fully considered. For instance, in the NE corner of Figure 15, an area of intermediate summer fallow potential exists that has no indication of cultivation. The reason for this apparent anomaly is because only the areas of high cultivation probability (potential) were selected for representation in Figure 15. However, areas of intermediate summer fallow probability were selected for representation, because no areas of high probability existed in Section 23. Therefore, the areas of cultivation and summer fallow probability will not coincide. Each analysis must be rectified with actual conditions at the site.

Figure 1. Monango NW 7.5 min. quadrangle in Dickey County.

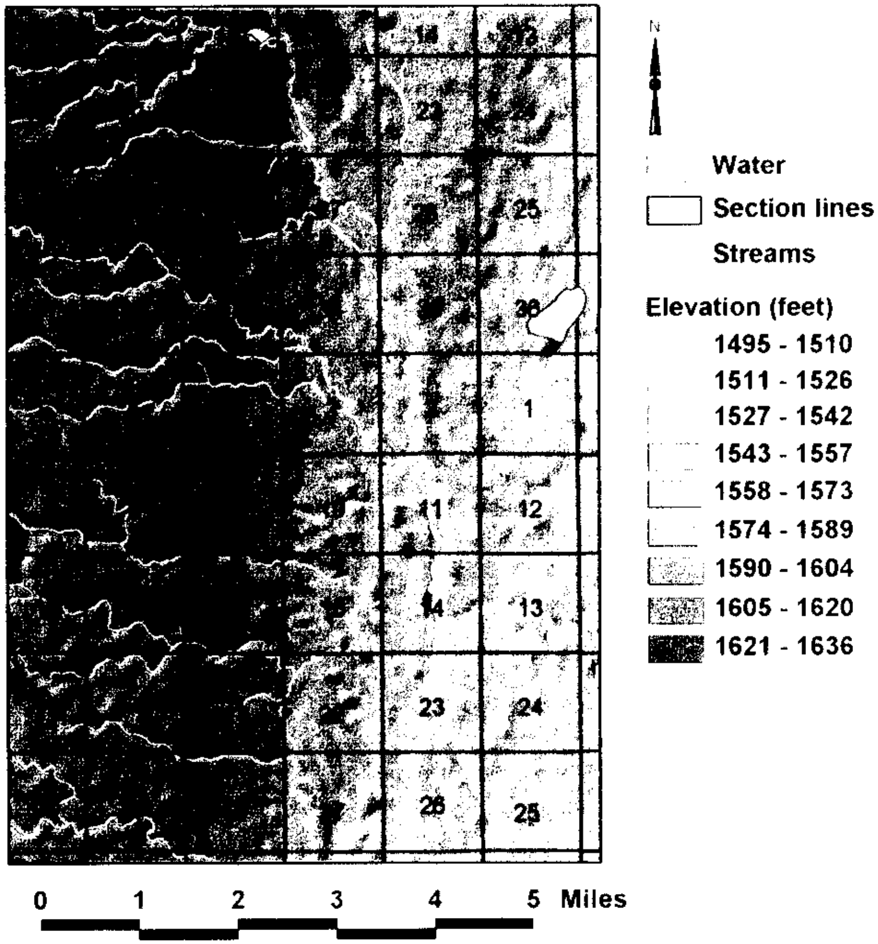


Figure 2. Proximity potential to contribute to nitrogen contamination of surface water in the Monango NW quadrangle in Dickey County.

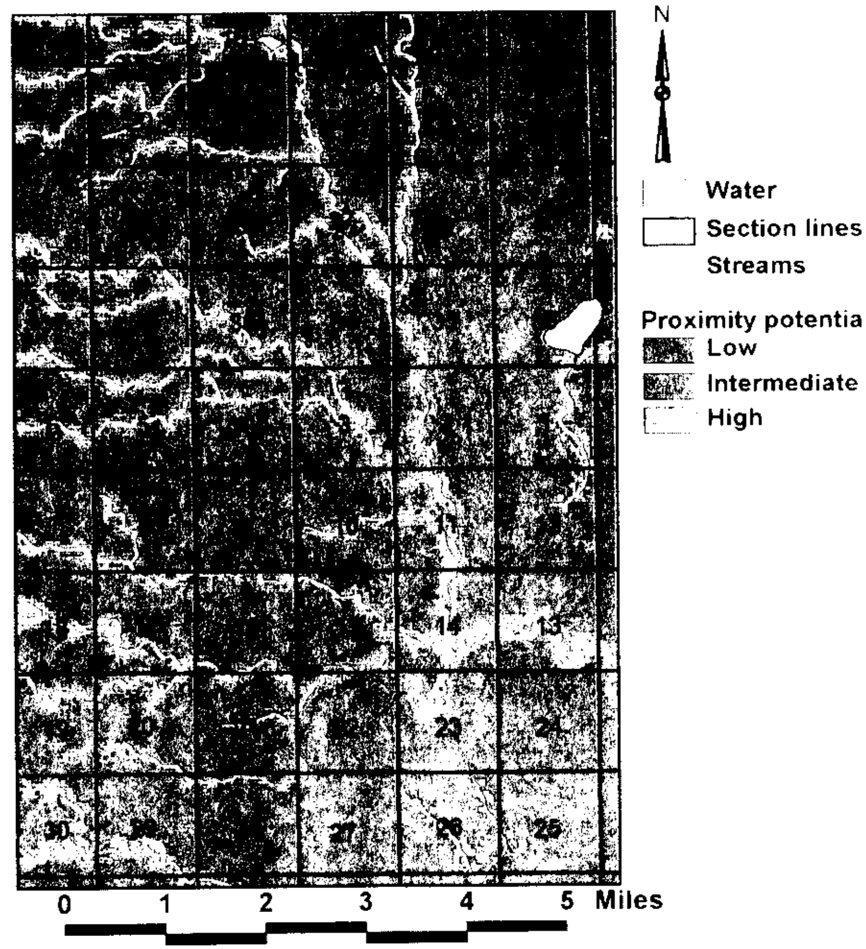


Figure 3. Soil aeration potential to contribute to nitrogen contamination of surface water in the Monango NW quadrangle in Dickey County.

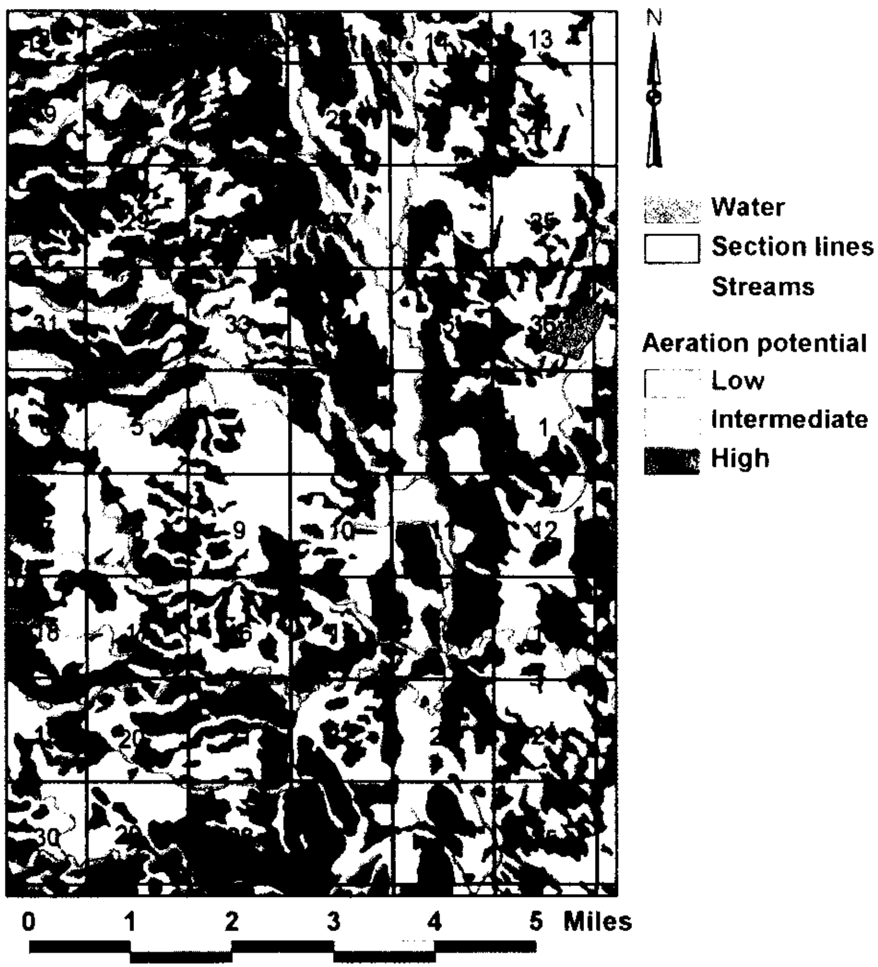


Figure 4. Solution transport potential to contribute to nitrogen contamination of surface water in the Monango NW quadrangle in Dickey County.

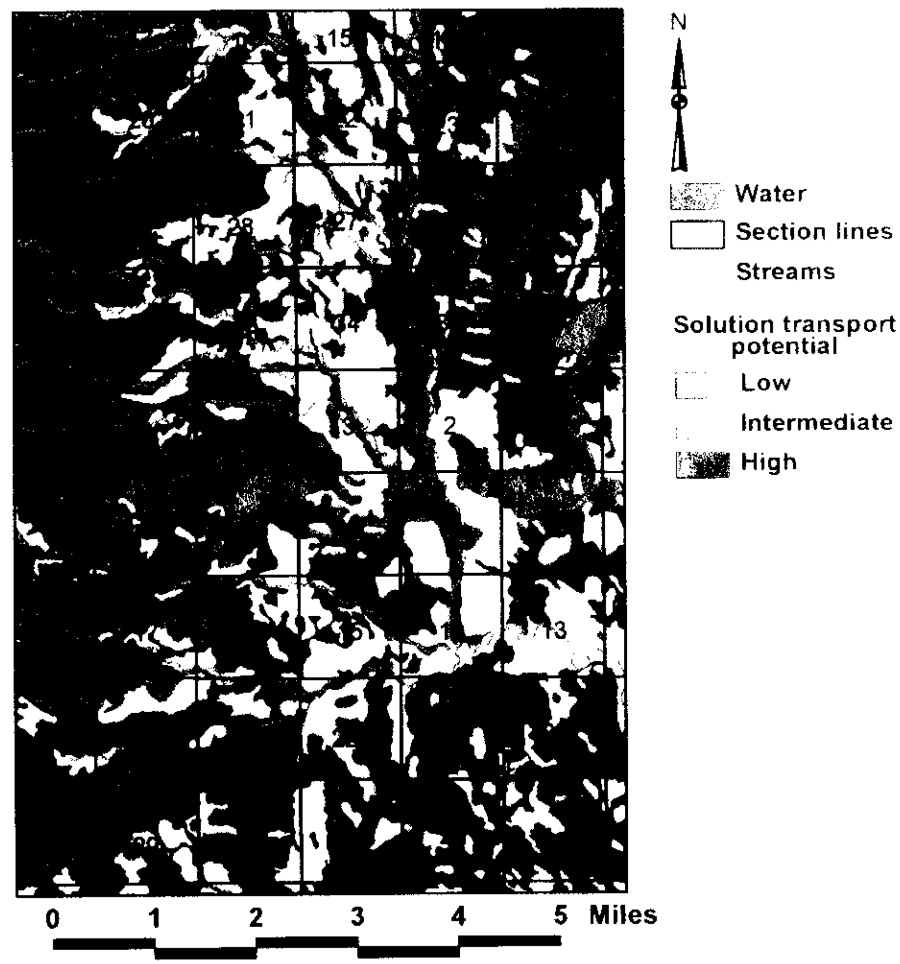


Figure 5. Sediment transport potential to contribute to nitrogen contamination of surface water in the Monango NW quadrangle in Dickey County.

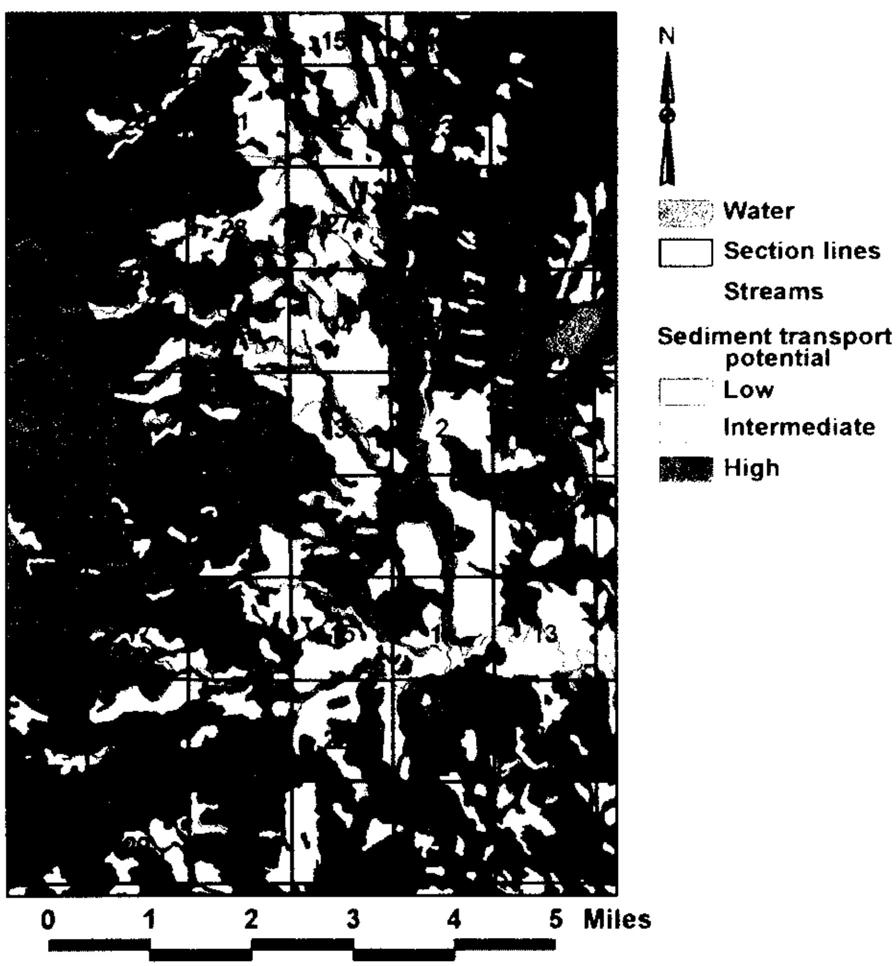


Figure 6. Slope potential to contribute to nitrogen contamination of surface water in the Monango NW quadrangle in Dickey County.

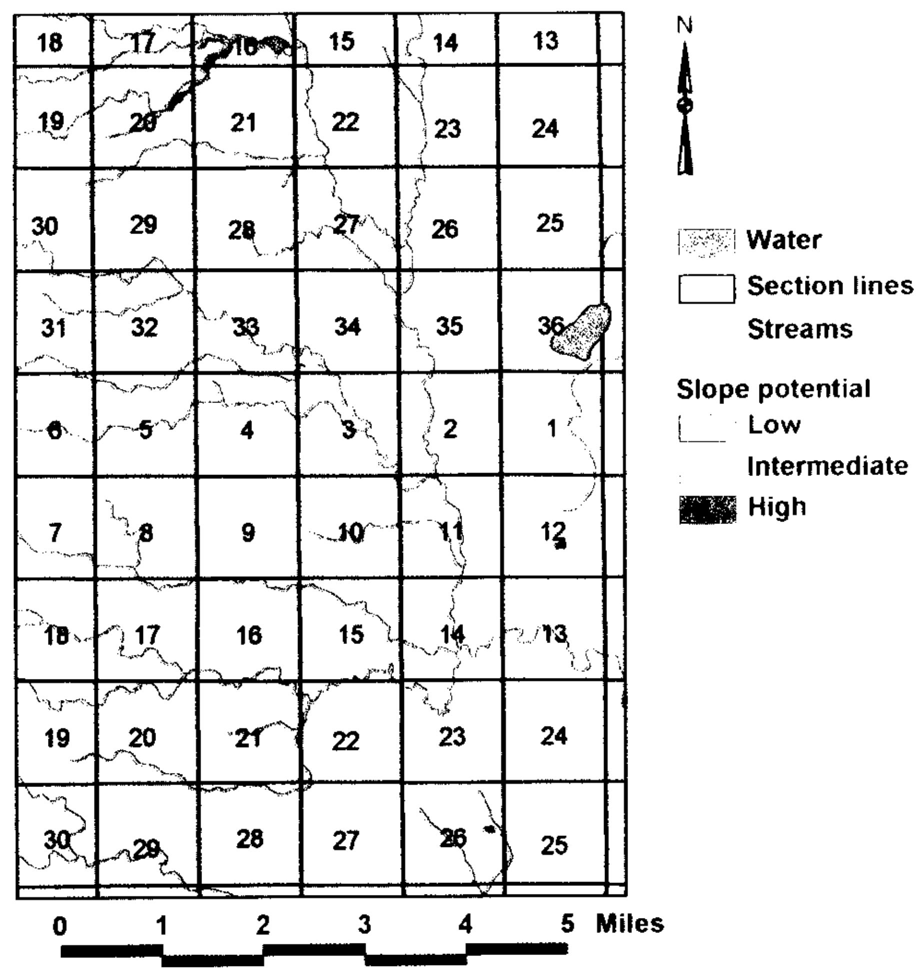


Figure 7. Flooding potential to contribute to nitrogen contamination of surface water in the Monango NW quadrangle in Dickey County.

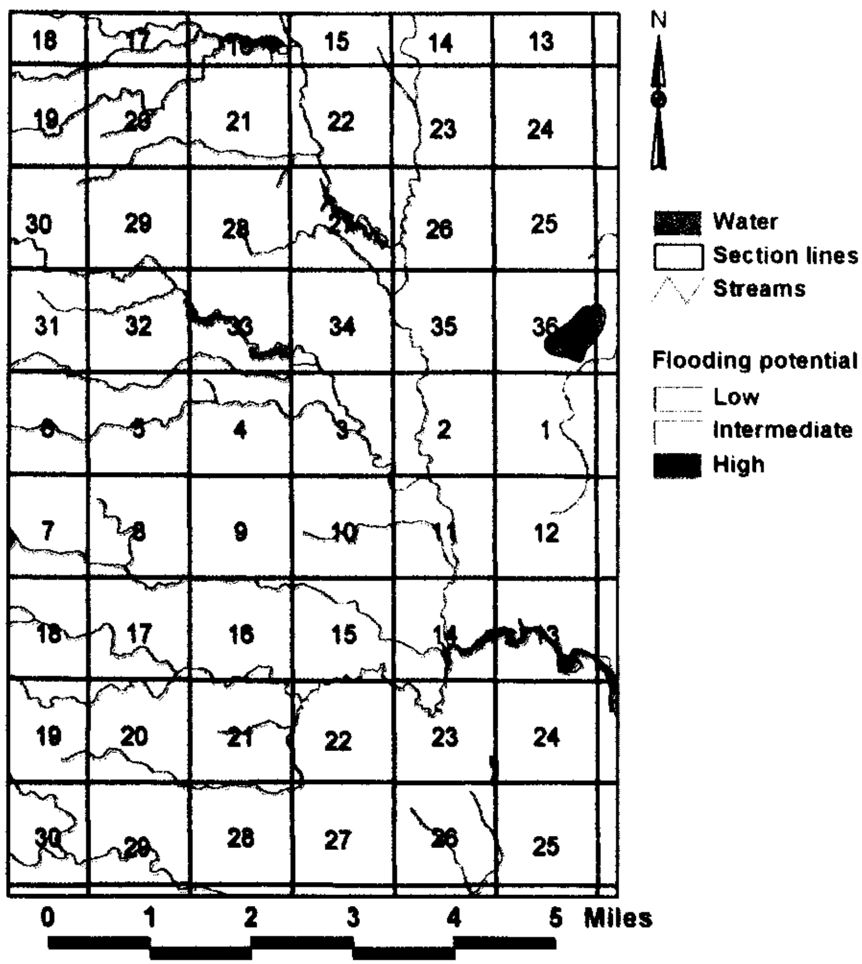


Figure 8. Vulnerability areas of surface water to nitrogen contamination in the Monango NW Quadrangle in Dickey County.

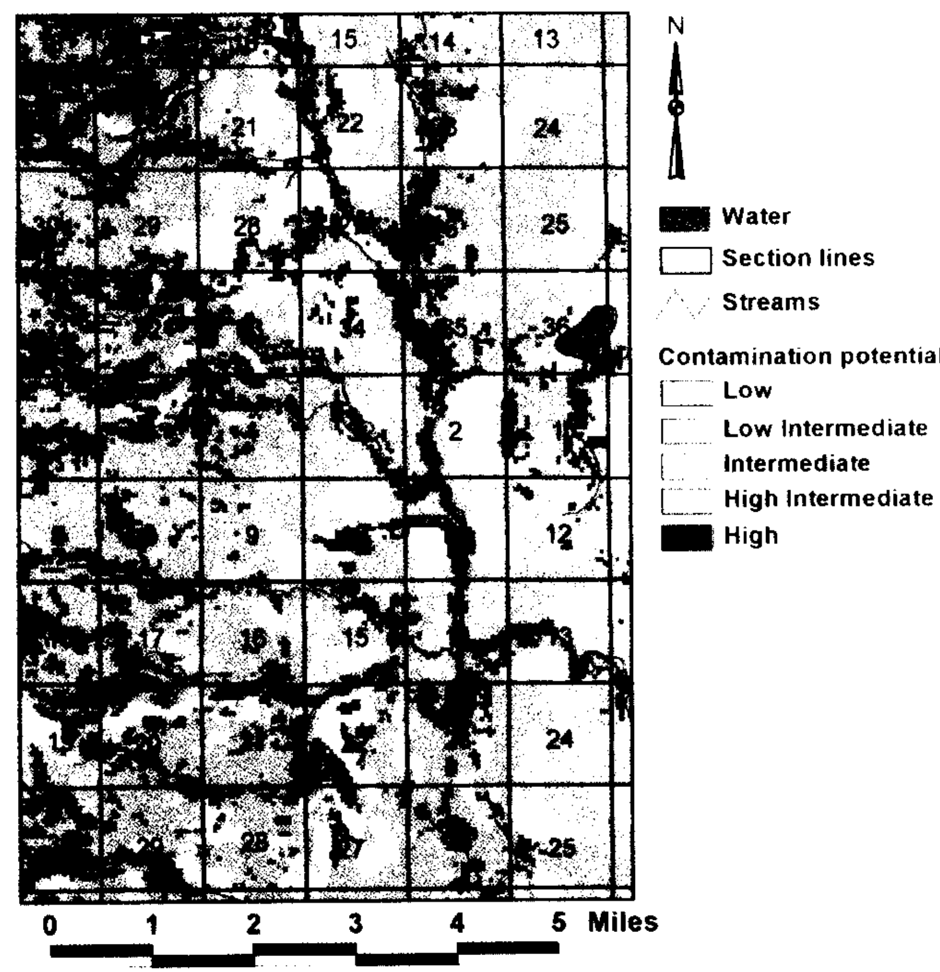


Figure 9. Rural activity site potential to contribute to nitrogen contamination of surface water in the Monango NW quadrangle in Dickey County.

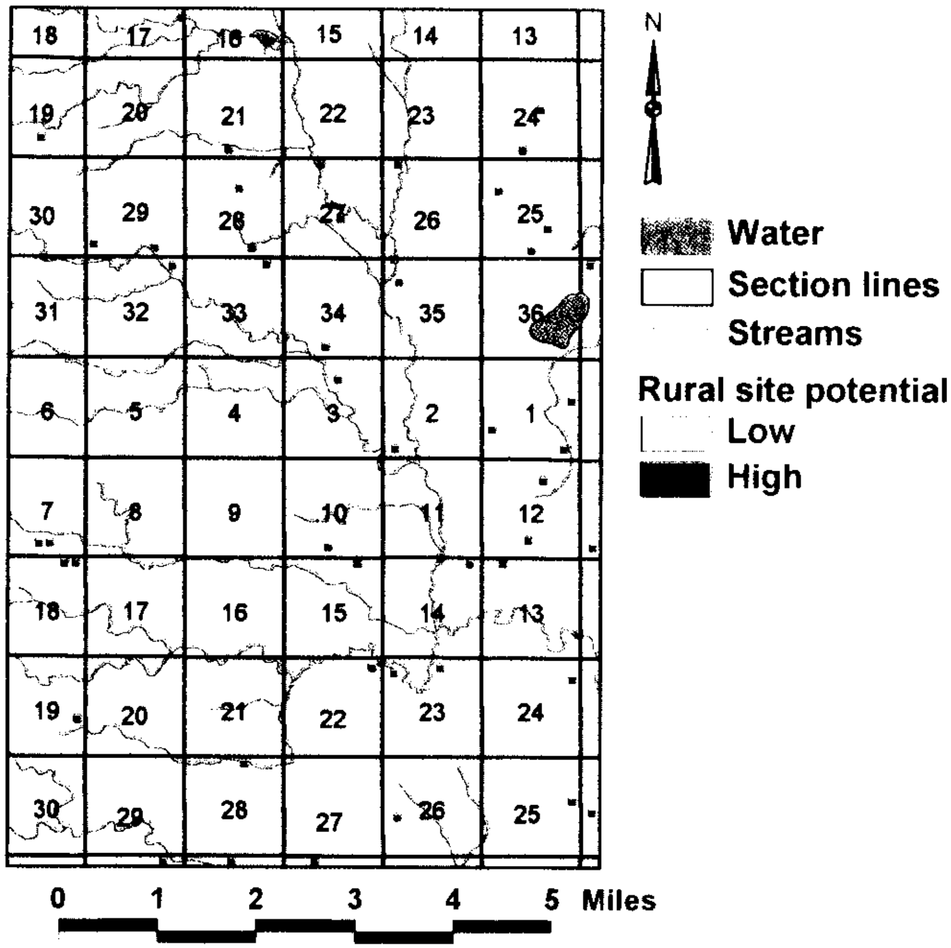


Figure 10. Summer fallow potential to contribute to nitrogen contamination of surface water in the Monango NW quadrangle in Dickey County.

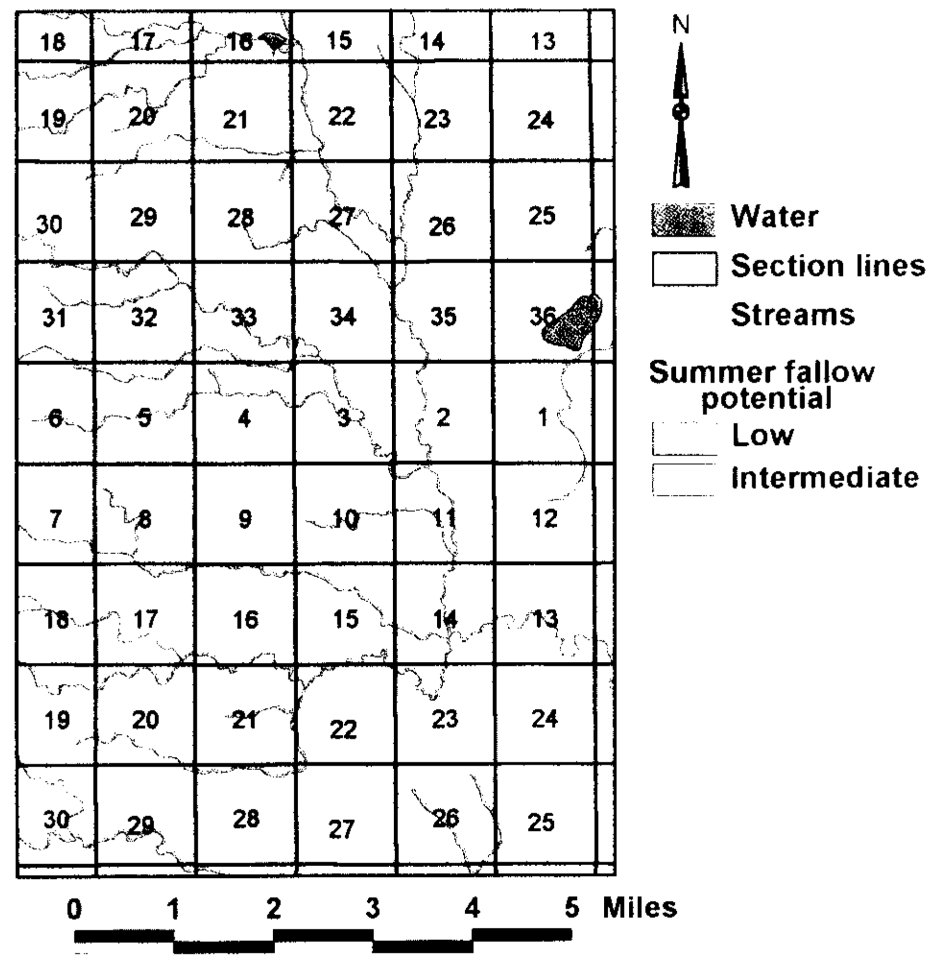


Figure 11. Cultivation potential to contribute to nitrogen contamination of surface water in the Monango NW quadrangle in Dickey County.

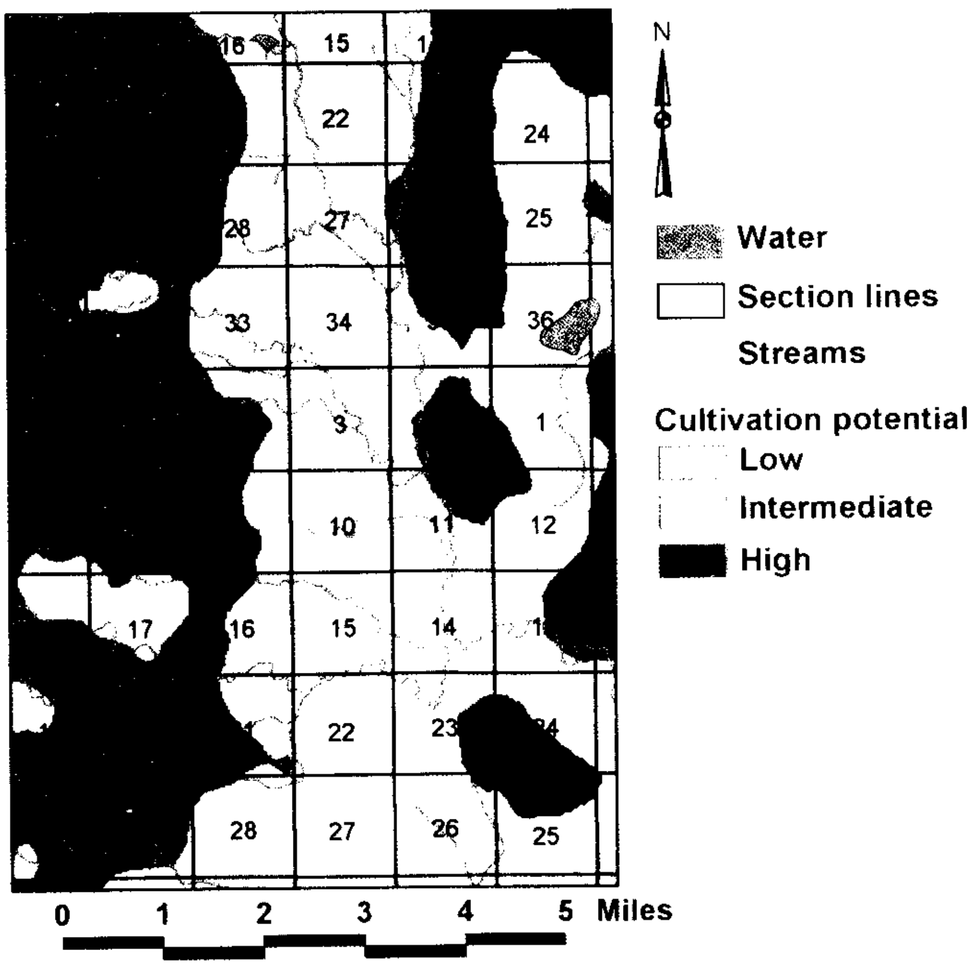


Figure 12. Areas of potential nitrogen contamination of surface water based on landuse in the Monango NW Quadrangle in Dickey County.

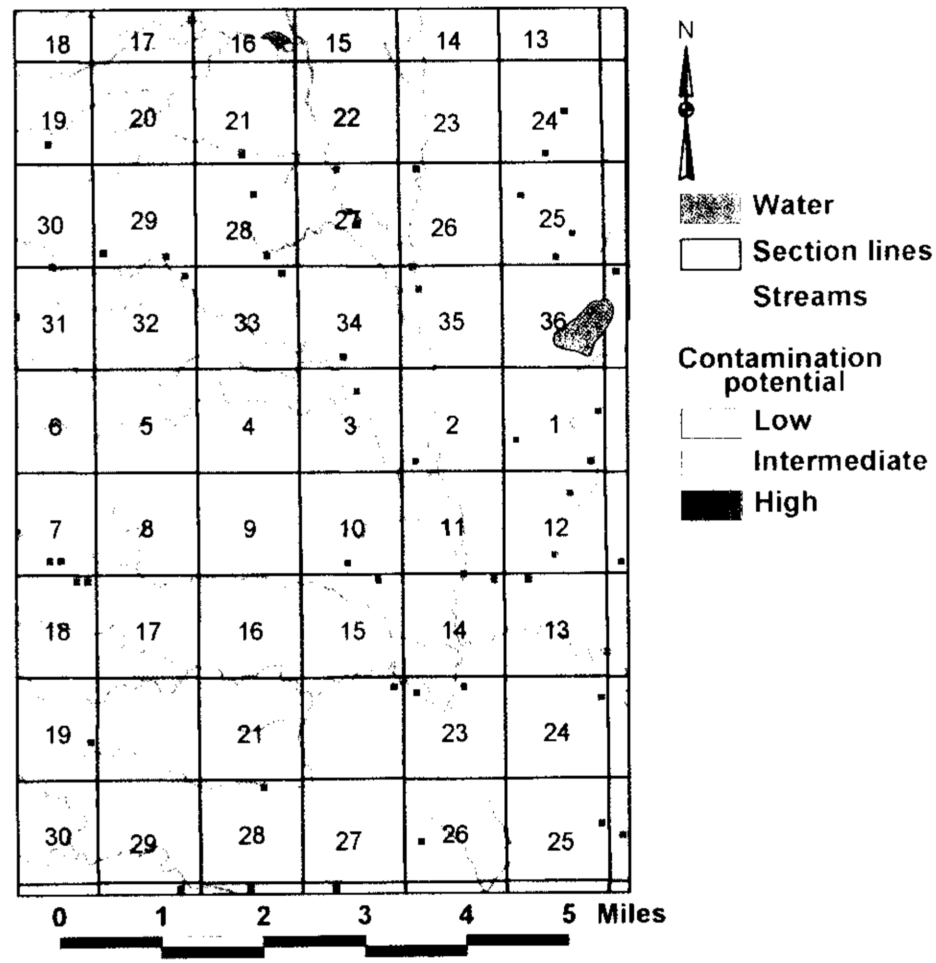


Figure 13. Sensitivity areas of surface water to nitrogen contamination in the Monango NW Quadrangle in Dickey County.

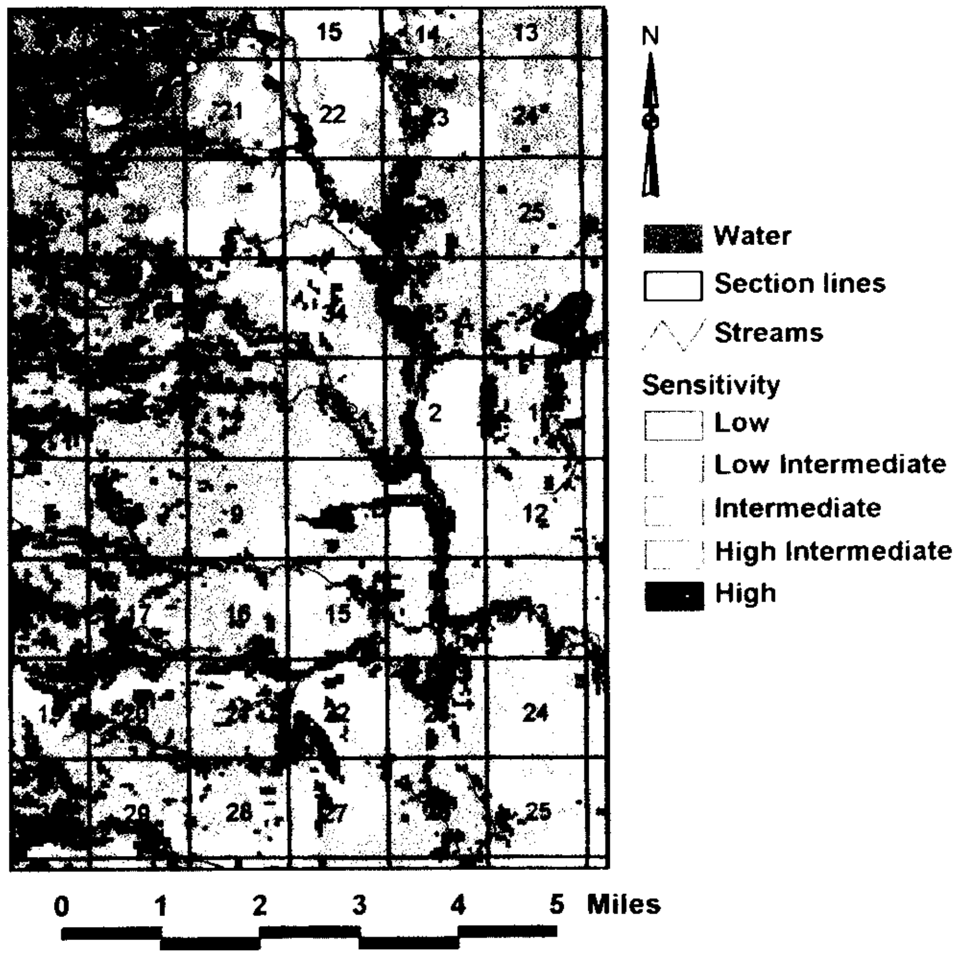


Figure 14. Vulnerability factors for Sec. 23, T.131 N., R.64W near Monango.

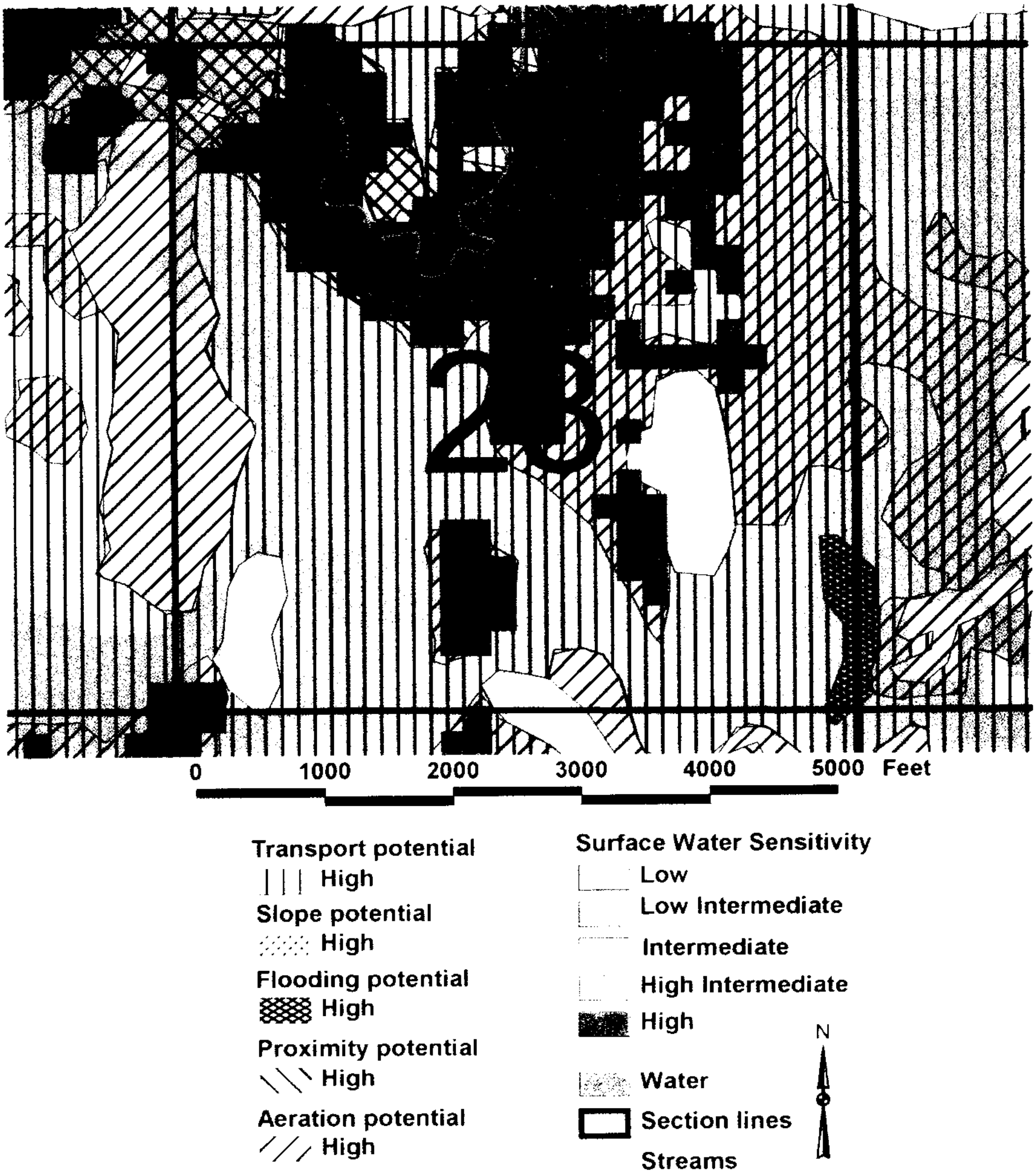


Figure 15. Anthropogenic factors for Sec. 23, T.131 N., R.64W near Monango.