Nitrogen in Minnesota Ground Water

Prepared for the Legislative Water Commission

Minnesota Pollution Control Agency

Minnesota Department of Agriculture

December 1991

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NITROGEN IN MINNESOTA GROUND WATER

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2. Mr. Montgomery was principal author of the following chapters: Chapter F (Nitrogen Source Comparison); Chapter G (Crop Production); Chapter K (Turfgrass); and Chapter L (Nitrogen Contributions from Forest, Prairie, and Miscellaneous Sources).

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ABBREVIATIONS

A	Acre
BMP	Best Management Practice
bu/A	Bushels per acre
BWSR	Board of Water and Soil Resources
CWA	Clean Water Act
CWI	County Well Index
CWP	Minnesota Clean Water Partnership Program
EPA	United States Environmental Protection Agency
Hb	Hemoglobin
lb/A	Pounds per acre
mg/l	Milligrams per liter
MetHb	Methemoglobin
MDA	Minnesota Department of Agriculture
MDH	Minnesota Department of Health
MGS	Minnesota Geological Survey
MPCA	Minnesota Pollution Control Agency
N	Nitrogen
NADP	National Atmospheric Deposition Program
NFS	National Forest Service
NH_4-N	Ammonium as Nitrogen
NI	Nitrification inhibitor
NO_2-N	Nitrite as nitrogen
NO_3	Nitrate
NO_3-N	Nitrate as nitrogen
NPS	MPCA Nonpoint Source Pollution Program
PSNT	Presidedress nitrate test
RI	Rapid infiltration wastewater treatment method
SE MN	Southeastern Minnesota Regional laboratory
SCS	Soil Conservation Service
STORET	National computerized water quality data base
SWCD	Soil and Water Conservation Districts
USDA	United States Department of Agriculture
USGS	United States Geological Survey
WHP	Wellhead Protection

Nitrogen in Minnesota Ground Water
Executive Summary

INTRODUCTION

The Minnesota Pollution Control Agency and the Minnesota Department of Agriculture have completed a comprehensive study examining existing data and literature related to nitrogen in the state's ground water. The report was required by the Minnesota Legislature as part of the Ground Water Protection Act of 1989. Other collaborators in the study included the Minnesota Board of Water and Soil Resources and the University of Minnesota Agricultural Experiment Station. Recommendations made by local governments in comprehensive local water plans were also considered for this report.

Nitrogen is one of the most widely distributed elements in nature and is present virtually everywhere on the earth's crust in one or more of its many chemical forms. Nitrate (NO_3), a dominant and mobile form of nitrogen, is commonly found in ground and surface waters throughout the country. Its source can be natural or the result of human activities.

There are increasing concerns over nitrate concentrations and, to a lesser degree, other forms of nitrogen, found in the state's water resources. The decline in ground water quality in many areas of the United States has paralleled increased nitrogen usage in plant nutrition and/or increased discharge from other human activities. There are no substitutes for nitrogen in plant nutrition: this fact makes the problem unique. The most practical way to approach nitrogen issues is through careful management.

Most of the report fits into two main sections. The first part examines current Minnesota ground water nitrogen conditions and trends. The second section investigates nitrogen contributors, their importance and effect on ground water, and the effectiveness of related management practices for minimizing nitrogen contamination. Other parts of the report review human and animal health and the environmental consequences of elevated nitrogen in ground water, options for communities and homeowners with wells contaminated by nitrates, and local, state and federal programs. This information will help policy makers, researchers, and local water planners who work with nitrogen-related issues.

The Executive Summary gives an overview of the report's contents and conclusions. For additional details, explanation, or references, the reader is encouraged to refer to the full report. The highest priority recommendations follow the Executive Summary.

Many of the findings of the Nitrogen Fertilizer Task Force (NFTF) were incorporated throughout pertinent sections of this report. The task force was established by the Legislature as part of the Ground Water Protection Act of 1989. Its purpose was to design a program to reduce nitrate loading into Minnesota water resources resulting from agricultural activities. The NFTF's membership included a diverse group of representatives from the agricultural community, environmental groups, local and state government. Voluntary Best Management Practices (BMPs), specific for unique combinations of soils, climatic conditions, and cropping systems, were designated as the cornerstone of the NFTF's educational campaign. A regulatory response was also developed by the NFTF.

The Minnesota Department of Agriculture (MDA) is developing and implementing many of the NFTF's recommendations. The BMPs proposed by the group have been adopted by the MDA and promoted by MDA, the University of Minnesota, local, state and federal agencies and private organizations and companies. Other measures, including responses to local nitrate contamination problems, BMP demonstrations, and periodic reviews of new research findings and technologies, are under development. The report to the Commissioner of Agriculture by the NFTF is included as an appendix to this report.

ENVIRONMENTAL AND HUMAN HEALTH EFFECTS

Nitrate is the dominant form of nitrogen in water. The only verified human health concern associated with exposure to nitrate is methemoglobinemia, commonly known as "blue baby syndrome." This disease, which generally affects only infants, affects the ability of the blood to carry oxygen. The Minnesota Recommended Allowable Limit (RAL) and the federal Maximum Contaminant Level (MCL) for nitrate in water are both set at 10 mg/L nitrate-N. Most documented methemoglobinemia cases in Minnesota occurred prior to 1950. Three cases, one fatal, have been documented in South Dakota, Minnesota and Iowa since 1979. The number of reported cases may underestimate actual events since most states, including Minnesota, do not have a methemoglobinemia registry established.

Two additional health effects have been postulated to be associated with exposure to nitrate in drinking water: a) esophageal and gastric cancer and b) central nervous system birth defects. Neither of these health effects have been adequately substantiated by experimental evidence.

Animals are also susceptible to methemoglobinemia. Ruminants (cows, sheep and goats) are potentially more susceptible than other animals. In determining a safe nitrate-N drinking water concentration for animals, the nitrate contribution from feed must also be considered. In general, the literature indicates that water containing less than 100 mg/L nitrate-N can be considered safe for livestock and poultry.

Nitrate can also contribute to increased algae and weed growth in surface waters. The ammonia and ammonium hydroxide forms of nitrogen are directly toxic to fish.

NITROGEN IN GROUND WATER-- EXISTING CONDITIONS AND TRENDS

Ground water monitoring results of four nitrogen compounds were examined for this report: nitrate (NO_3), nitrite (NO_2), ammonium (NH_4) and organic nitrogen. Nitrate was the compound most frequently found at elevated concentrations in ground water and is the focus of much of the discussion in this report.

Nitrite, ammonium and organic nitrogen concentrations, measured in over 350 wells throughout the state by four agencies, were generally quite low. Average nitrite-N concentrations in 367 wells were 0.02 mg/L. Elevated ammonium is occasionally found in ground water, most often in association with mismanagement of human, animal or industrial waste.

The nitrate concentration in any given sample of well water is the result of numerous factors, including surrounding land use and management, ground water flow hydraulics, ground water residence time, climatic conditions, ground water chemistry, well depth in relation to geologic stratigraphy and water table elevation, type of well sampled and well construction.

Nitrate data sets

Minnesota does not have a statewide ground water monitoring program that is designed specifically to assess the extent and trends of nitrate concentrations. Nitrate data have been collected in Minnesota through various federal, state and local programs, with most of the information generated since the late 1970s. For this report, 16 data sets were examined to better understand the degree of the nitrate problem in Minnesota and trends related to nitrate differences across the state. There are major differences between existing data sets in sampling purpose, field and laboratory methodologies, areas sampled, years and frequency of sampling, data management, and documented well location and construction information. EPA-approved methods were used to produce data in 14 data sets representing a total of 26,340 wells. Computerized data from seven of these data sets were obtained, evaluated and described in this report. A brief description and summary statistics were included for nine other data sets. Three data sets also provided limited information regarding nitrate differences between aquifers and changes in nitrate concentration with time.

Degree of problem

The data summarized in this report clearly illustrate that nitrate contamination of ground water resources is a problem in many areas of Minnesota. Major differences in groundwater nitrate conditions are found when comparing results from the 16 data sets. Data sets created by targeting mostly shallow wells in geologically sensitive areas under agricultural production showed a relatively high percentage (27 to 44%) of wells exceeding 10 mg/L nitrate-N. Sampling programs targeting newly constructed wells or municipal wells showed a much lower percentage (1 to 4%) of wells with nitrate-N exceeding 10 mg/L. MPCA ambient monitoring program results from 484 wells in different aquifers throughout the state showed nitrate-N exceeding 10 mg/L in 7% of the wells sampled. Results from certain county sampling programs showed more than 20% of wells exceeding 10 mg/L nitrate-N, yet other counties had less than 6% of wells exceeding 10 mg/L nitrate-N.

The degree of contamination is variable across the state. In many areas, there is very little information to assess the situation. A majority of the nitrate data has been collected in the southern half of the state, particularly southeastern Minnesota (including the Twin Cities). Limited data in northeastern and northwestern Minnesota show a low percentage of wells with elevated nitrate. Central Minnesota appears to have a wide range of ground water nitrate conditions. Numerous wells in southeast and southwest Minnesota have elevated nitrate levels; however, there is great variability in the degree of nitrate contamination within these regions. South central and west central Minnesota show less evidence of nitrate problems than southeast and southwest Minnesota, but both of these regions have high nitrate wells in certain areas. Nitrate concentrations are variable in the seven-county metropolitan area, but are generally higher towards the southeast.

Difference among aquifers

Three data sets had sufficient nitrate data collected from different aquifers to allow limited comparison of nitrate among aquifers. In all three data sets, unconfined surficial sand aquifer wells generally had higher nitrate levels than buried drift wells. Nitrate concentrations were consistently low in older bedrock formation aquifers of the southeastern quarter of the state (St. Lawrence, Franconia, Ironton, Galesville, Mt. Simon and Hinkley formations). Varying degrees of nitrate contamination are evident in the other major bedrock aquifers in the southeastern quarter of the state, including the Cedar Valley-Maquoketa-Dubuque-Galena, Decorah-Platteville-Glenwood, St. Peter and Prairie du Chien-Jordan.

Change with time

There are very few wells in Minnesota that have continuous nitrate sampling records sufficient for time-trend analysis. Twenty-two monitoring wells have been sampled quarterly since 1986 by MDA. Results showed some wells with increasing nitrate levels and other wells with decreasing nitrate levels. In addition to the MDA well data analysis, 29 Minnesota Department of Health (MDH) municipal well records were visually examined for this report. Criteria for selecting these 29 wells included 1) elevated (>5 mg/L) nitrate-N during recent tests, and 2) at least five measurements taken over a 12- to 40-year period. The relatively small number of wells analyzed, inconsistency in trends, and uncertainty of data integrity limits the usefulness of this data set in drawing regional or statewide conclusions regarding long-term nitrate trends.

Relationship between age of ground water and nitrate

For this report, tritium and nitrate data were obtained for 302 ground water samples collected during 1990 by several different groups in many areas of the state. Tritium is a radioactive isotope that can be used to help understand the age of ground water. Atmospheric concentrations of tritium increased considerably during the mid to late 1950s due to nuclear testing. All wells (34) with nitrate-N above 10 mg/L withdrew water that had entered the ground since 1953. These results suggest that the current nitrate problem is due to land use activities since 1953. From 77 different well water samples dated as pre-1953 water, only one had nitrate-N in excess of 1 mg/L. The low nitrate levels in pre-1953 water suggest either: 1) very little nitrate was entering ground water before 1953; 2) nitrate entering ground water prior to the mid-1950s was lost through denitrification; or 3) a combination of the two.

Nitrate losses within aquifers

With the exception of plant uptake of nitrogen from areas of very high water tables and discharge to surface water, the only known ground water nitrate loss mechanism is through denitrification (conversion of nitrate to gaseous nitrogen). Studies conducted in the United States and other countries have shown denitrification to occur within aquifers when the chemical and biological conditions are suitable (low dissolved oxygen, low redox potential, denitrifying bacteria present, and most importantly, a source of organic carbon). While there is a potential for denitrification to occur in Minnesota ground water, this issue has been examined in very few areas of the state and is extremely difficult to assess.

Surface water nitrogen

Streams routinely monitored by the MPCA at 110 sites across the state from 1981 to 1990 generally had nitrate-N levels below 3 mg/L. Nine sites had nitrate-N levels exceeding 10 mg/L 10 percent of the time. The same stream sites monitored for ammonium-N generally showed concentrations less than 1 mg/L. Ammonia (NH_3), which is toxic to fish, exceeded standards in eight of the 110 stream sites 10 percent of the time. In lakes, nitrate-N is usually found at concentrations less than 0.1 mg/L and ammonium-N is typically between 0.4 and 2 mg/L. Since some lakes in southwest Minnesota are reported to be nitrogen-limited, existing nitrogen in these areas may be controlling the amount of algae produced.

NITROGEN CONTRIBUTORS AND BEST MANAGEMENT PRACTICES

Cropland contributors

Yields have increased dramatically in the past 30 years as a result of a better understanding of plant nutrition and improved plant varieties. One dilemma facing agriculture is that the application of nitrogen, a critical component in increased crop production and profitability, will often have detrimental effects on water quality if not optimally managed. Fortunately, when optimum nitrogen management is used, adverse impacts on water quality can be minimized or eliminated. A variety of management practices are currently available for crop production which are compatible with minimizing nitrogen movement into water resources. Contributions and characteristics of all Minnesota's nitrogen sources must be considered when formulating an overall nitrogen strategy.

Total estimated annual inputs across the state's cropland from agricultural activities contributed approximately 773,000 tons of plant-available nitrogen (1987 estimate). Commercial fertilizers, legumes, and manures contributed 75, 12 and 13% of this total and would equate to 53, 9, and 9 lb/cropland acre, respectively, for these "applied" sources. Relative importance of each source varied significantly across the state.

In addition, soil organic matter contributes a variable amount depending upon soil conditions. Estimates are approximately 10 to 100 lb N/A/year. The reviewed literature clearly identified the need to account for all sources of nitrogen in a management plan. Ground water nitrogen contributions from agricultural activities can be dramatically reduced by accounting for these sources and matching the inputs, both in terms of amounts and timing, to the physiological needs of the crop.

Commercial Fertilizer

Dependence on commercial fertilizer has grown tremendously in Minnesota agricultural production since the early 1960s. Fertilizer sales in the past five years have now stabilized and have ranged from 550,000 to 650,000 tons nitrogen/year. In 1990, Minnesota ranked fourth nationally in nitrogen fertilizer sales. Approximately 69% of the nitrogen was applied to corn (grain, silage, and sweet), 26% applied to small grains, 2% to sugar beets, and the remaining to miscellaneous crops. The following conclusions can be made about nitrogen fertilizers:

- The amount of available nitrogen (nitrate and ammonium), regardless of its source (commercial fertilizer, manure, legume, or residual soil nitrate), is clearly the single most important factor affecting leaching losses.

- Leaching losses can be greatly minimized by not exceeding the crop's physiological need for nitrogen; losses are commonly linear or curvilinear after the "threshold value" for a given crop is exceeded. In corn production, it appears that the balance between nitrogen use efficiency and yield falls somewhere between 90 and 95% of the maximum obtainable yield.

- Leaching losses are highly dependent upon the amount of nitrogen left in the soil profile at the end of the cropping season. Under most Minnesota cropping/climatic conditions, the majority of leaching losses take place during the non-cropping season.

- With specialty crops such as potatoes, it is not clear within the current literature what level of yield reduction would be required to keep leaching losses at an acceptable level.

- The practice of fall nitrogen applications on fine-textured soils does not necessarily pose a significant threat to ground water. Soils in southeast Minnesota are an exception. Fall application is not recommended in this region due to high geological sensitivity. The Nitrogen Fertilizer Task Force report clearly identifies regions where fall applications are feasible.

- Sidedress applications in fine-textured soils can result in nitrogen which is "positionally unavailable" resulting in reduced nitrogen use efficiency. Preplant or early sidedress applications are highly recommended.

- Timing of nitrogen applications in coarse-textured soils is critical. Sidedressing, multiple applications, and fertigation are instrumental management tools in reducing nitrate leaching losses.

Manure

Manure is a vital and valuable nitrogen resource. In 1987, approximately 98,000 tons of plant-available nitrogen was supplied by manure, which is equivalent to 9 lb/A when spread uniformly across all of the state's cropland. The significance of manure is tremendously variable with location; the highest application rates (per area of cropland) are located in the central and southeast portion of the state. On a county level, manures can account for 2 to 25% of the "applied" amounts (credits from legumes, manures, and fertilizers) and averaged 12% across the state based on the 1987 data. Based on county level assessments of the relationships between crop nitrogen needs and manure-nitrogen, it appears that sufficient cropland exists to adequately accommodate manure from existing animal populations.

A number of complications arise when attempting to use manure as a nitrogen source. Vague estimates of manure application rates, extreme nitrogen variability of the manure, variable gaseous losses during storage and application, and uncertainties associated with the proportion available for plant uptake are some of the most salient problems. This may explain why most state and federal studies come to the same conclusion: traditionally, farmers fail to take the proper credit for manure. Like any nitrogen source, the organic nitrogen in manure is eventually transformed to inorganic forms. Accordingly, over-application of manure poses an environmental threat. The following conclusions were made about manures and manure management.

- Information is sparse regarding how Minnesota farmers store, credit, and apply their manure.

- The heaviest loadings of manure are occurring in the states' most sensitive hydrogeologic regions.

- Ground water contamination will likely occur if rates, regardless of the source, exceed crop needs. Due to the slow nitrogen release from manure, the amount available for leaching at any point in time is limited. Yet continual mineralization occurs after the crop needs are satisfied, and there is the potential for "off-season" leaching losses. Few studies have examined these long-term effects.

- Storage and handling have a profound effect on the amount of nitrogen in manure by the time it is distributed onto soil. Volatilization losses do not pose an immediate water quality concern, but the lack of understanding of these losses makes it very difficult to properly credit the portion of the nitrogen that eventually is applied to the field. For this reason, the Nitrogen Fertilizer Task Force highly recommends periodic manure analysis.

- The general consensus among researchers is that farmers need to be better educated about manure management. An overall effort to educate farmers must take into account manure and legume credits, as well as commercial fertilizers, if a ground water protection program is to be successful.

Legumes

In 1987 legume crops supplied approximately 96,000 tons, or 12%, of the "applied" plant-available nitrogen. This is equivalent to applying 9 lb/A across all of Minnesota's cropland acres. Soybeans, alfalfa, and clover are the biggest contributors and the University of Minnesota recommends crediting 20-40, 75-150 and 75 lb N/A for these respective crops. Existing literature suggests that alfalfa and clovers are excellent scavengers for nitrate and beneficial to ground water quality. However, there are some concerns once these crops are "plowed down" since elevated nitrate conditions can occur as a result of the tremendous mineralization which follows. Conclusions and recommendations about legumes follow.

- Existing literature strongly suggests that plowing down or other methods of killing alfalfa increases the potential for nitrate leaching losses during the next one to two cropping seasons. Additional research is needed on the long-term effects of other legumes on water quality.

- High nitrogen use crops must be selected and other sources of nitrogen must be minimized following legume crops.

- The practice of applying manure before plowing down alfalfa or clover results in an oversupply of nitrogen and a high potential for leaching loss.

Cropland management

The ultimate goal in nitrogen management is to maximize nitrogen use efficiency. The more efficiently the producer can get nitrogen into the crop, the less that will be available for leaching through the root zone and eventually into the ground water system. The effectiveness of a number of nitrogen management strategies have been reviewed. Effects of yield goal selection, tillage, nitrification inhibitors, timing strategies, irrigation management and other practices under Minnesota conditions have been evaluated. Where necessary, related data from the state's contiguous neighbors were also used. The resulting conclusions, along with the recommendations from the Nitrogen Fertilizer Task Force, provide a solid platform of management practices specific for Minnesota's diverse agricultural conditions.

Yield Goal Selection

- Selection of yield goal and the subsequent nitrogen application rate has a profound effect on ground water quality. Limited research has indicated that growers tend to set unrealistic goals, commonly missing them by 10 to 30%, and as a result, application rates are higher than necessary to maximize yields and maximize economic returns.

- The Nitrogen Fertilizer Task Force is strongly recommending the "running average" concept for yield goal selection. Yield goals are based on the past five-year average, excluding the worst year. This approach will provide a sound basis for a field-specific nitrogen rate that is environmentally and agronomically sound.

- Tools such as soil testing, and to a lesser degree plant tissue sampling, play a valuable role in determining application rates once a yield goal is established. Soil testing, in the appropriate portions of the state, is highly recommended.

- Technology for farming soil types rather than fields is quickly becoming a reality and may play an important role in the future of agriculture.

Irrigated Agriculture

- A number of state and national studies strongly indicate a direct correlation between irrigation development and nitrate concentrations in ground water. There are a number of contributing factors including: higher nitrogen rates; low soil moisture-holding capacities; and increased leaching due to the additional water inputs. Yet the "cause and effect" relationship is often poorly understood under grower-operated conditions.

- Irrigation, even on some of the coarse-textured soils of Minnesota, does not necessarily mean a significant increase in subsurface drainage. Irrigation is good insurance that a healthy, uniform stand of plants capable of high nitrogen uptake will be developed. Under careful nitrogen management, the bulk of the leaching losses will occur during Minnesota's off-season recharge period, not during the irrigation season.

- Keeping losses of nitrogen to an acceptable level may be extremely difficult, requiring very precise management in some of Minnesota's very coarse-textured soils.

- Irrigators need to be well educated in all facets of irrigation/nitrogen management. Efforts must be made to keep irrigation an asset rather than an environmental liability. The potential for environmental degradation under poor management is extremely high.

- Fertigation is a valuable tool for minimizing the amount of available nitrogen in the soil profile at any one time during the cropping season. Environmental and economic benefits of fertigation generally outweigh the risks when proper safety equipment is used.

Tillage

- Percolation is higher under conservation tillage than conventional tillage due to: wetter soil profiles caused by mulching effect of crop residue; more macropores; and possible reduction of surface runoff.

- Nitrate concentrations are commonly lower under conservation tillage, but because of the increased percolation losses, the net leaching loads are commonly the same as conventional tillage practices.

- The volume of surface runoff can be reduced as much as 20 to 50% in comparison to conventional tillage practices. However, nitrogen losses from surface runoff under any type of tillage are generally minor in comparison to other avenues of loss.

- Nitrogen management decisions such as rates and timings will generally have a much larger impact on water quality than method of tillage in cropping systems reliant upon commercial fertilizers. Tillage methods on soils following legumes or manure application can have a major effect on mineralization rates.

Inhibitors

- Effects of nitrification inhibitors have been highly variable under Minnesota's diverse soils and climatic conditions.

- Under irrigated, coarse-textured soils, researchers have found that inhibitors can reduce the potential for leaching. Factors such as selection of proper rates and efficient irrigation management can commonly overshadow the differences that inhibitors make.

- Inhibitor effects are most likely to be observed in yield performances when nitrogen is limiting.

- Under conditions where high percolation of soil water (generally limited to coarse-textured soils) or soils prone to extended saturated conditions (generally fine-textured soils), the use of nitrification inhibitors should be encouraged.

- Nitrification inhibitors can, under specific conditions, increase leaching losses by keeping the nitrogen "positionally unavailable" during the nitrogen uptake period.

- Specific recommendations for the use of inhibitors are given for each region of the state in the "Recommendations of the Nitrogen Fertilizer Task Force."

Feedlots

A rough estimate of the number of feedlots in Minnesota is 45,000 to 60,000. Manure nitrogen can move into ground water below outdoor animal holding areas, manure storage areas, fields with applied manure and abandoned feedlots.

A soil seal will usually develop under animal-holding areas that are continually used, preventing much movement of water through the soil surface. Saturated conditions in the feedlot surface, coupled with high amounts of organic carbon, makes a feedlot surface conducive for denitrification. This seal can be broken and a number of investigators have found nitrate and ammonium moving through the soil profile and into ground water below inactive feedlots. This is a common problem with abandoned feedlots. In abandoned feedlot situations, manure scraping and removal followed by planting alfalfa or other high nitrogen use crops will reduce the potential for nitrate leaching to ground water. Runoff from active feedlots and subsequent infiltration has also been shown to be an important nitrogen contributor to ground water.

Earthen manure storage basins installed in medium and coarse-textured soils without added liners have been found to leak in the northern United States, resulting in nitrogen movement to ground water. A number of other studies have shown earthen basins to effectively seal themselves, with minimal ground water impacts. While the results regarding self-sealing are conflicting, it is generally believed that earthen basin site, design and construction are important considerations in minimizing nitrogen impacts to ground water.

The Minnesota Pollution Control Agency first developed rules to control pollution from animal waste facilities in 1971. In 1979, the rules were changed to allow counties to process feedlot permits, and since that time 25 counties have volunteered to participate in the program. Over 16,000 feedlot permit applications have been reviewed in the last 20 years. Until a few years ago, the primary focus of the feedlot program was on surface water protection. Ground water protection has received greater attention from the feedlot program in recent years.

Septic Systems

Approximately 400,000 Minnesota households dispose of wastewater into septic systems. In many cases, there is no other economical, environmentally acceptable alternative for treating these wastes. Unfortunately, these systems are not designed to remove nitrogen. Nitrate is usually the contaminant of greatest concern below a properly designed and constructed septic system. On the average, about 45 gallons/person/day of wastewater with a total nitrogen concentration of about 50 mg/L is released into soil from septic systems (7 lb of N/person/year).

Impacts of septic systems on ground water will primarily depend on the nitrogen loading to the aquifer, diluting capacity of the aquifer, and the potential for denitrification in the soil below the system. The diluting capacity of an aquifer is reduced when the density of systems increases.

From 66 individual septic systems monitored for ground water impacts in numerous studies in northern U.S. and Canada, the following generalizations can be made about the nature of nitrogen contamination from individual on-site wastewater treatment systems:

- Nitrate-N concentrations are often between 10 and 40 mg/L at the surface of the water table directly below septic absorption systems in coarse-textured soils.

- Nitrate concentrations are highest at the top of the water table near the points of effluent release and decrease substantially with depth.

- Dilution and dispersion result in decreasing ground water nitrate concentrations down gradient so that nitrate-N is usually below 10 mg/L within 50 to 100 feet from the absorption field when background nitrate is low. In aquifers with a low potential for dispersivity, long narrow plumes can result with sharp lateral and vertical boundaries.

- Highly elevated ammonium can be found in ground water below septic systems, where systems do not conform with current siting and construction standards (e.g., high water table).

While individual septic systems may not create obvious increases in well water nitrate concentrations, the cumulative impact of multiple drainfields in a housing development are more noticeable. Aquifer nitrate-N concentrations between 5 and 15 mg/L have often been found to exist in wells on the downgradient side of high density developments. One of the critical factors affecting nitrate concentrations is average lot size. When average lot sizes are less than 1 to 2 acres in a development with numerous homes and coarse-textured soils, there is a great potential for shallow wells to have elevated nitrate.

Several different types of systems have been developed which promote denitrification, resulting in nitrogen losses of between 50 and 95 percent. Further testing and evaluation of these systems is needed.

The amount of nitrogen in septage (liquid and solid material pumped during cleaning) generated in the state is estimated to be about 360,000 pounds. While the total contribution of nitrogen statewide from septage is very small, localized ground water nitrogen problems can result when improperly applied.

Turf

Information on applied nitrogen rates to Minnesota's turf (lawns, golf courses and landscaped areas) and its subsequent effect on ground water is very limited. However, research from other northern states indicate that turf, when fertilized with reasonable rates, can be satisfactorily maintained yet present little risk to ground water quality. Turf specialists from a number of universities are advocating turf as an environmental benefit and fertilization may be more beneficial in protecting ground water than contaminating it. The following generalizations can be made about the nitrogen contamination from turf:

- Potential environmental risks associated with nitrogen applied to turf appear to be minimal if application rates do not exceed the turf's physiological needs. Leaching losses are commonly minimal because of prolific root development, increased moisture-holding capacity directly below the thatch, and turf's ability to utilize high rates of nitrogen. Risks of leaching losses rapidly increase as application rate exceeds plant nutrient needs.

- Maximum amounts of required nitrogen were found to vary with management, residual soil nitrogen, and varieties but in general, most research suggests that applied rates should not exceed 160 to 175 lb/A/year (3.6-4.0 lb/1000 ft^2). Adequate nitrogen nutrition insures good vigorous top growth, extensive root development serving as an effective filter, and a porous protective covering capable of minimizing runoff.

- Runoff volumes under turf tend to be minimal during the growing season. Existing studies have concluded that nitrogen runoff losses are small in comparison to other avenues of nitrogen loss. Losses could be a potential problem if runoff occurred immediately after fertilization.

- Timing of nitrogen application, nitrogen source selection and proper rates are of critical importance under very coarse-textured soils, particularly overlying shallow aquifers.

Municipal and Industrial Waste

Land application of wastewater through spray irrigation or rapid infiltration basins is permitted for eight private domestic complexes, 46 municipalities and 30 industrial facilities. From very limited data, applied municipal wastewater appears to have relatively low nitrogen concentrations. Industrial wastewater can have very high nitrogen concentrations, sometimes exceeding 100 mg/L. Little is known about industrial effluent nitrogen concentrations or the ground water nitrogen levels below fields receiving industrial wastewater in Minnesota.

One hundred fifty-two communities regularly apply sewage sludge in Minnesota on a total of about 9000 acres of cropland. Research has shown that excessive sludge application can result in elevated ground water nitrate concentrations. Municipal sewage sludge application is regulated by the MPCA and application rates are usually based on nitrogen needs of the crop.

Excessively leaking wastewater treatment ponds have been shown to cause elevated nitrogen levels in ground water. Criteria for pond design has become more stringent in recent years, thereby decreasing the amount of leakage from newly constructed ponds.

During 1991, 575 municipal and 317 industrial facilities were permitted for discharge of treated wastewater into surface water. Total nitrogen concentrations in this wastewater are often over 25 mg/L. MPCA sampling of 52 municipal wastewater treatment facilities with mechanical treatment indicated a mean effluent total nitrogen concentration of 17 mg/L from all facilities.

Municipal and industrial waste is not a large part of nitrogen input to ground water statewide. However, improperly designed, constructed or managed treatment systems do represent potential localized ground water threats. Nitrogen from these sources should always be accounted for when managing cropping systems.

Natural Sources

Nitrogen concentrations under natural forest and prairie conditions are generally very low, seldom exceeding 3 mg/L. Natural fires and forest clear cutting can increase leaching losses, but these alterations are generally small.

Atmospheric deposition contributions throughout the state, as quantified from several monitoring programs during the past 20 years, have determined that inorganic nitrogen amounts typically range from 5-12 lb/A/year. Depositional amounts in areas burning fossil fuels or immediately adjacent to ammonium sources, such as feedlots, could be significantly higher.

Contribution of nitrogen from the decomposition of soil organic matter is extremely important, yet little is known in terms of estimating the portion which will undergo mineralization. Tile drainage, tillage, previous crop residue, and climatic conditions have a profound effect on mineralization rates. Although the relative contribution from soil organic matter is less important in agricultural soils, organic matter is estimated to supply 40 to 50% of the state's inorganic nitrogen supply.

Various biological and chemical methods for estimating mineralizable nitrogen are available. However, none of these tests currently appear to be universally accepted and reliable enough to warrant routine use in soil testing laboratories.

RESPONSE OPTIONS FOR COMMUNITIES AND HOMEOWNERS

Options for communities with unacceptable nitrate levels include drilling a new well, blending high and low nitrate water, installing a treatment system, or connecting to a rural water system. The latter two options are often cost prohibitive and it is not always possible to drill to a deeper aquifer. The preferable long term solution is pollution prevention. Implementation of wellhead protection is advised.

Nitrate testing of public and domestic water supplies is necessary to promote public health protection. Homeowners with high nitrate may have the following options: drilling a new well, installing treatment systems that remove nitrate, purchasing bottled water or continuing to drink high nitrate water. There are disadvantages with each of these options. The option most often recommended by Minnesota Department of Health is drilling a new well.

STATE AND FEDERAL PROGRAMS

In response to a growing national concern about the ecological and health impacts of nitrogenous compounds, the U.S. Environmental Protection Agency is developing a nitrogen action plan. The nitrogen action plan work group has drafted recommendations that are organized into five categories, including 1) develop a state nutrient management programs, 2) improve on-farm nitrogen management to protect water quality, 3) improve public and private drinking water quality, 4) increase control of point sources through current regulatory authority, and 5) research areas of uncertainty. The draft federal nitrogen action plan would be implemented in two phases. Phase I emphasizes the use of current regulatory authorities, pollution prevention techniques, and research. Activities under Phase II would begin if voluntary efforts and current legal authorities were insufficient.

Minnesota has a number of existing and developing programs that affect, or have the potential to reduce nitrogen movement to ground water. The effect of these programs on ground water nitrogen levels will not be known for many years. The only statewide effort that focuses specifically on nitrogen pollution prevention is the Minnesota Nitrogen Fertilizer Management Plan, which was developed and recommended by the Nitrogen Fertilizer Task Force. Several programs exist that each deal with a variety of contaminants from specific sources, such as feedlots, septic systems and municipal and industrial waste. Other programs deal with multiple pollution sources, including the Minnesota Clean Water Partnership Program, Wellhead Protection Program, and Comprehensive Local Water Planning. Several other regional and local efforts are underway. These existing programs show promise for minimizing nitrogen movement to ground water.

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Nitrogen in Minnesota Ground Water High Priority Recommendations

The main objective of this report is to supply the Legislative Water Commission and other concerned policy makers with information and recommendations about how to minimize nitrogen in ground water. The conclusions and comments presented in the report are based on the review of existing data and research. Nitrogen sources, relationships of those sources to ground water contamination problems, and the effectiveness of best management practices are reviewed and summarized.

This study represents a significant effort to consolidate and evaluate the existing information on this important economic and environmental issue. Despite this major undertaking, definitive solutions to nitrogen management and water quality protection are not completely clear. As additional information is developed, improved nitrogen management systems should be evaluated and implemented.

The report's recommendations are designed to improve statewide assessment of water quality status, provide technical support for improved nitrogen management, and focus research and education efforts. Implementation of the recommendations will be beneficial in reduction of nitrogen into ground water and in further defining the nature of the nitrate problem. However, the time required to observe the resulting effect on ground water quality is unknown because of the complex nature of this issue. This report should be updated periodically to incorporate new information from the many ongoing programs and research activities related to nitrogen.

The authors viewed the following recommendations as most important from a statewide perspective. Issues and priorities may be quite different at the local level.

MONITORING AND DATA NEEDS

Long-term monitoring is needed to assess nitrate trends over time in the principal aquifers throughout the state. In addition, long-term monitoring should be focused in high priority or problem areas to evaluate the effectiveness of implemented Best Management Practices. This information is needed to gauge progress in meeting the goals of the Ground Water Protection Act and to prioritize ground water protection efforts. Long-term monitoring efforts should be incorporated into existing monitoring programs, such as the MPCA Ground Water Monitoring and Assessment Program and MDA Pesticide and Nutrient Monitoring Assessment Program. Strict sampling, analysis, and data management standards are needed to produce reliable results.

Minnesota needs reliable, statewide information to identify areas and aquifers where nitrate concentrations exceed or approach water standards. This information could be used to: 1) help set county priorities regarding nitrogen management, 2) target programs such as the Nitrogen Fertilizer Management Plan and Clean Water Partnership; and 3) provide an increased level of drinking water protection for domestic water supply users.

To increase the usefulness of nitrate data, the following recommendations are made:

- Establish statewide standards for collection and analysis of nitrate data.
- Maintain a statewide nitrate data registry for water quality results meeting established data standards.
- Enhance existing state programs to provide technical assistance to local and regional monitoring efforts.

CROPLAND PRODUCTION AND FEEDLOTS

Commercial fertilizer, legumes, and manures supplied approximately 75, 12 and 13% of the applied plant-available nitrogen statewide in 1987. Nitrogen contributions from each of these sources are extremely diverse across the state. It is impossible to separate the direct effect on ground water from these individual sources under normal farming practices. Yet the net effect of poor nitrogen management is the same regardless of the source: nitrogen application in excess of crop needs poses a potential ground water problem.

Sound nitrogen management could alleviate much of the nitrate contamination problem resulting from current agricultural production levels and at the same time result in economically viable crop production. The outcome of the extensive literature review within this report suggests that corn, the crop that utilizes about 70% of the state's commercial nitrogen, can be grown at economically viable yields with minimal impacts on the state's water resources. Environmental nitrogen problems associated with agriculture will not be solved by focusing in on one specific nitrogen source or management practice; it is absolutely essential that an overall nitrogen management plan is tailored to specific characteristics associated with individual farming systems. Specific recommendations are given below.

Yield goal selection

From a statewide perspective, acceptance and selection of a realistic yield goal may be the most important single recommendation. Based on data from other Midwestern states, this could translate into reductions of 10 to 30 lb/A of nitrogen input under corn production with little effect on yield.

- An economic risk assessment of the "running yield goal" approach for Minnesota corn production should be conducted. With this approach, yield goals are based on the past five-year average, excluding the worst year. Similar efforts need to be made for small grains where other factors, such as stored soil moisture and soil test values, confound the decision making process.
- Localized information demonstrating the economic and environmental benefits resulting from realistic yield goal selection using the "running yield goal" concept should be promoted.
- Nutrient management specific to soil types and conditions within a field will become widely available through grid soil sampling, on-the-go yield measurement, soil-specific fertilizer

application and other new application technologies. The "running yield goal" concept should be promoted as a component of nutrient management specific to soil types and conditions.

- Research shows that dealers have the biggest influence on farmers in terms of nitrogen management. Dealers, crop consultants, and others who advise farmers need to be better educated on the balance between farm economics and the environment.

Manure management and feedlots

Contributions from manure are the heaviest in southeast and central Minnesota which are generally the state's most sensitive hydrogeologic regions. Even in these areas, nitrogen from manure is still only a moderate contributor of plant-available nitrogen at a county level; however, critical overloading is more than likely occurring in site specific situations. Clearly, the critical issue is not the amount of manure generated in these and other areas of the state but how manure is managed. Manure management has received less research and educational attention than commercial fertilizers. Specific recommendations include:

- Clearly the biggest need in manure management is to educate farmers to take the proper credits for this valuable resource. Manure testing, ability of farmers to estimate application rates, and keeping records are critical components of this process.

- Additional information is needed regarding nitrogen leaching and surface losses under recommended manure application rates. Effects of timing of applications, application methods, interactions with tillage, and other management factors need to be better understood. This type of information will help refine best management practices.

- Techniques for predicting nitrogen availability from manure in the application year and subsequent years need to be improved.

- Improved application equipment that enables more precise application rates and more uniform field distribution is clearly needed. This would greatly improve farmer's confidence in the use of manure as a source of nutrients.

- Research is needed to obtain a better understanding of how Minnesota farmers store, credit and apply their manure resources. Basic inventory information (such as types and numbers of holding facilities, quantities of actual manure, etc.) is limited. This will help target educational efforts.

- The feasibility of manure exchange or transport programs for concentrated animal operation areas must be explored.

- Nontraditional manure handling methods, such as composting, need to be assessed from both economic and environmental perspectives.

- Crediting relationships between manure and legumes and subsequent addition of fertilizer nitrogen must receive greater attention. Research and government programs should address the water quality aspects of *total* nitrogen management.

- State funded incentive programs should be developed, training should be provided and technical assistance should be increased to encourage counties to adopt and actively administer the MPCA's Feedlot Rules (Chapter 7020). Counties should submit annual reports describing program status.

- Local government, through local water planning, could make a valuable contribution by collecting data pertaining to storage, crediting, and application of manure. Information on abandoned feedlots should also be part of the data collection effort. Technical and financial assistance will be necessary to design and evaluate inventories.

- Cost sharing for the construction of adequate manure storage facilities should be increased.

- Technical support from local, state and federal levels needs to be expanded in the areas of facility construction/maintenance and manure management assistance.

Irrigated agriculture

Irrigation management, coupled with nitrogen BMPs, can minimize nitrate leaching. However, the nature of most of Minnesota's irrigated crop production is directly related to sensitive hydrogeologic conditions. Problems result from the low moisture-holding capacity, increased nitrogen inputs in response to higher yield goals, and specialty crops which require intense management. Intense rainfall may cause some uncontrollable leaching despite optimum irrigation and nitrogen management.

- Localized crop coefficient curves should be developed so the irrigator can make accurate estimates of crop water use and minimize percolation losses.

- Best Management Practices, specific for Minnesota's outwash sand regions, need to be developed for irrigated potatoes and other high nitrogen use crops. The BMPs must incorporate the nitrogen/irrigation interactions.

- Effects of irrigation scheduling schemes, such as deficit scheduling based on plant phenology, need to be better understood in terms of nitrate leaching losses.

- Information is limited in terms of how farmers actually perform their irrigation scheduling, particularly in the eastern portion of the state. This type of information will help focus educational efforts.

- Irrigators need to be educated in all facets of irrigation/nitrogen management. A voluntary certification program, ideally administered through the irrigators themselves, should be developed.

SEPTIC SYSTEMS

Most septic systems currently in use do not effectively remove nitrogen. While the amount of nitrogen generated is small in terms of statewide loading, septic systems can affect well water on the local level.

- The MPCA and University of Minnesota (working with the Individual Sewage Treatment Advisory Committee and the State of Wisconsin), should further evaluate, test, develop and promote denitrification systems. Each system should be evaluated for costs of installation and maintenance, nitrogen reduction, other pollutant reduction, and overall system performance. Based on the results of this work and on alternative system testing in other states, recommendations should be made regarding the feasibility of using these systems on a widespread basis in Minnesota.

- Until septic systems that treat nitrogen are proven, feasible and are commercially available, minimum lot sizes for new housing developments should be set by each county so that ground water impacts are minimized.

THE NITROGEN FERTILIZER TASK FORCE RECOMMENDATIONS

The Nitrogen Fertilizer Management Plan created by the Nitrogen Fertilizer Task Force in 1990 describes Best Management Practices (BMPs) which are appropriate for virtually every possible condition across the state's diverse soils and climate. The plan emphasizes education and, if required, a regulatory response. Educational efforts through the University of Minnesota Agricultural Experiment Station, Extension Service, and the Minnesota Department of Agriculture have already begun. Extension agents, Soil and Water Conservation Districts, associated state agencies, and most importantly—farmers—are currently being educated about the Nitrogen Fertilizer Management Plan. Associated extension bulletins will be available in late 1991. Plans for training dealers and consultants and others within the industry are also being prepared. The MDA, in association with organizations and agencies, should continue to promote and coordinate the implementation of the plan.

Research is continually providing new information and technology applicable to nitrogen management. Research is needed to address the problems identified throughout this report. Procedures and policies must be developed to incorporate these new findings into the existing body of accepted BMPs. One possible mechanism for achieving this would be a technical advisory group to the MDA. This group would be responsible for the review of ongoing BMP development and would formulate recommendations to the MDA regarding the technical aspects of BMP development.

In the review and development of the BMP recommendations, the task force identified aspects of nitrogen management that required additional research. This research is necessary to further refine BMPs and enable the nitrogen users to more precisely apply the optimum environmental and agronomic nitrogen practices. Funds should be allocated for basic and field research on total nitrogen management, especially that which incorporates water quality concerns. The following is a list of some of the needs identified by the task force. This list is not meant to be inclusive, but rather serves only to highlight some immediate needs.

- Nitrogen interactions and credits from non-fertilizer sources such as organic matter, legumes and manure need to be more thoroughly understood. Attention should be directed to initial and subsequent release of nitrogen and the impact on water quality.

- Development and verification of soil testing techniques to predict plant-available nitrogen in humid conditions needs to be accelerated. Efforts to develop a "quick test" that meets Minnesota needs and conditions should be supported.

- Manure management research needs to be increased and accelerated because of the lack of information available to guide sound environmental and agronomic decisions related to manure management.

In areas where significant nitrate contamination from agricultural practices exists or levels are increasing, and voluntary BMP adoption is unacceptable, the NFTF recommends a regulatory response. The MDA is responsible for implementing the regulatory requirements. Many details of this response still need to be addressed. The MDA is currently assessing the number of sites which may require a response. The assessment and response to areas with intense nitrate contamination should proceed and the MDA should continue to develop and implement as quickly as resources allow.

PUBLIC HEALTH PROTECTION

All parents of infants should know the nitrate concentrations in water used to mix formula for their babies and be made aware of health concerns associated with drinking high-nitrate water. An assessment should be made to ascertain the level of awareness that families expecting babies have regarding their well water nitrate levels and methemoglobinemia. Adequate protection of infants, the most vulnerable group of society to high nitrate levels, must be a high priority.

DEGRADATION PREVENTION GOAL

The Ground Water Protection Act of 1989 included a goal of degradation prevention for the state of Minnesota. The language in the Act states:

> "It is the goal of the state that ground water be maintained in its natural condition, free from any degradation caused by human activities. It is recognized that for some human activities this degradation prevention goal cannot be practicably achieved. However, where prevention is practicable it is intended that it be achieved. Where it is not currently practicable, the development of methods and technology that will make prevention practicable is encouraged."

Programs, policies, and activities to achieve this goal as it relates to nitrogen need to be developed and implemented. The progress toward achievement of this goal also needs to be assessed continually.

PURPOSE AND SCOPE

This study was conducted in response to the 1989 Groundwater Bill (Chapter 103H, Article 1, Section 12), which directed the Minnesota Pollution Control Agency (MPCA) and the Minnesota Department of Agriculture (MDA) to prepare a report on nitrate (NO_3) and related nitrogen compounds in ground water. The report was prepared in consultation with the Board of Water and Soil Resources and Minnesota Experiment Station. Other agencies were also consulted during the report writing and review process.

One of the primary objectives of this study was to examine and summarize existing data and literature in order to provide legislators, federal, state, and local water planners, and other policy makers the information necessary to most appropriately respond to the issue of nitrogen (N) in ground water. This comprehensive report was written with the intent of providing enough detailed information and related references to satisfy those readers interested in studying specific issues, yet focusing on the most pertinent and relevant information needed to understand the situation in Minnesota.

An overview of N characteristics and health and environmental concerns associated with N are provided in the background section. The body of the report consists of two parts which are divided into twelve chapters. Part 1 (chapters A through E) provides information about: 1) background on health related issues in humans and animals; 2) the current state of knowledge regarding existing NO_3 conditions in Minnesota ground water, 3) changes in NO_3 concentration with time, 4) fate of NO_3 in ground water, 5) concentrations of nitrate, ammonium, and organic N in groundwater, and 5) surface water N conditions. A section on monitoring needs is also included within Part 1.

Part 2 (Chapter F, G, H, I, J, K, M) describes N inputs, causative factors of NO_3 contamination and best management practices and policy associated with the primary nitrogen sources. A comparison was made to assess the relative contributions from major N sources. Six chapters in Part 2 describe N management and potential ground water impacts associated with crop production, feedlots, septic systems, municipal and industrial waste, and turf. A discussion of N from soil organic matter, forests, and prairies is presented in Chapter L. Nitrogen impacts from landfills and N fertilizer manufacture and handling is also included in this chapter.

Another objective of this study was to review federal, state and local response to the issue of N in ground water and make feasible recommendations for improvement in state and local response (Chapter M). Recommendations are made at the end of many chapters throughout the report with the highest priority recommendations listed following the executive summary.

The Nitrogen Fertilizer Task Force report, completed in August 1990, is included as an addendum to this report.

NITROGEN CYCLE AND SOURCES[1]

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THE NITROGEN CYCLE

Nitrogen (N) in the environment is governed by a complex of interrelated chemical and biological transformations. These reactions are summarized in what is termed the "Nitrogen Cycle" which describes the pathways, sinks and transformations of N. Nitrogen can take on many different chemical valences and the characteristics of the various forms vary tremendously. It is important that the audience of this publication understands some of the basic processes involved with this cycle. Sound N management decisions can then be made based upon this understanding.

Processes summarized are biological, chemical or physical in nature. Figure A-1 depicts the major reactions of the cycle. Although there are numerous species of N, the species of particular interest in the soil are nitrate (NO_3^-), ammonium (NH_4^+), and organic forms of N. Nitrate and ammonium are of particular interest since these two inorganic species are the only forms which higher plants can utilize for their nutritional needs. Characteristics of these species are summarized:

> **Organic nitrogen:** This is the predominant species within the soil system and a common constituent of sewage and manure. Amounts found in soil vary with the amount of organic matter present. For Minnesota soils, amounts of organic N present can range from 1,000 to 7,000 lb/A. Soils, as a general rule, contain approximately 1000 lb/N per acre for each percent of soil organic matter. Most of this is associated with the upper 6 to 12" of the profile. Organic N is tightly bound and must be transformed to inorganic forms by microbial activity before plant uptake can occur. Organic N may be a significant source of N in surface runoff but rarely contributes to ground water contamination.
>
> **Nitrate (NO_3):** Nitrate is extremely soluble and its negative chemical charge excludes it from adsorption to soil colloid exchange sites (which also tend to be negatively charged). Due to these characteristics, the NO_3 is highly mobile in the soil profile. **It is important to note that throughout the publication, the authors express ground water concentrations in the following form: NO_3-N. This is the concentration, or mass per volume of water, of N which is in the nitrate form. Utilizing this terminology, the U.S. drinking water standard is <u>10 mg/L of NO_3-N</u>.** Some authors,particularly in Europe, express the concentration to reflect the total mass of both the N and the three associated oxygen molecules. This type of expression is written simply as NO_3. The U.S. drinking water standard would then be 4.4 times greater and written as **44 mg/L of NO_3.** It is important that the reader understands the difference between these two forms of expressions.

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1. Taken and modified from MDA's "Recommendations of the Nitrogen Task Force", 1990.

Ammonium (NH_4): Ammonium, due to its positive charge, is tightly bound to soil colloids surfaces and clay interlayers. Unlike NO_3, NH_4 seldom moves through the soil profile. The most serious environmental threat is to surface water though soil erosion processes. Ammonia (NH_3) is one of the gaseous forms of N and is quickly converted to NH_4 within the soil system. Ammonia and NH_4 based N fertilizers are the most commonly used forms in United States.

Another form (although of much less importance in plant nutrition and ground water contamination) which needs to be defined is nitrite.

Nitrite (NO_2): Nitrite is an intermediate product in the conversion of NH_4 to NO_3 in soil. It is of toxicological concern in the human system. Although highly soluble, nitrite is also very unstable and is rarely detected in ground water except at very low levels.

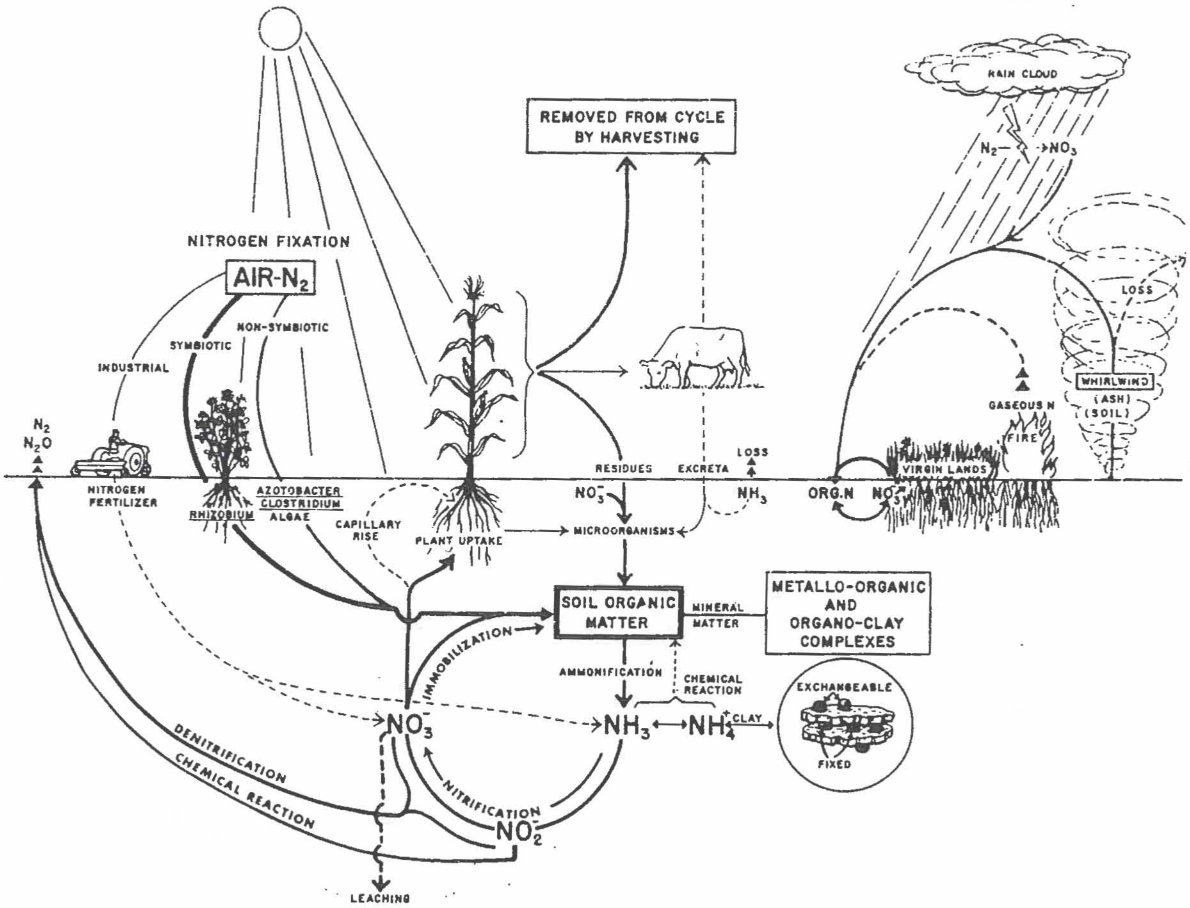

Figure A-1. The nitrogen cycle in soil. Taken from Stevenson (1982).

The primary chemical and biological processes of the N cycle include:

Mineralization: This is defined as the microbial degradation of organic N to produce the inorganic forms of NH_4 (intermediate step is called ammonification) and NO_3.

Nitrification: The oxidation process of NH_4 to NO_2 to NO_3. Very specific bacteria are responsible for these two conversion reactions.

Immobilization: The utilization of NO_3 by plants and microbes producing various organic N species. A related term is **assimilation** which is the utilization of NH_4.

Denitrification: The biochemical reduction of NO_3 and NO_2 to gaseous molecular N or an oxide of N. This can be a significant loss mechanism in saturated soils and in some aquifer conditions.

Volatilization: The loss of NH_3 to the atmosphere. This occurs primarily when NH_4-based or urea-based fertilizer or manure are surface-applied without incorporation.

Leaching: The process of transport of solutes in water percolating through the soil profile. Nitrate is the principal N species susceptible to leaching. Leaching of NO_3 is the primary avenue of N movement into ground water systems.

NITROGEN SOURCES

Nitrogen is commonly termed as being ubiquitous meaning that it is seemingly present everywhere at the same time. It is interesting to note that there is 3 tons of N per square foot of the earth's surface associated with the atmosphere. Yet the most dominate source of the earth's N supply is in the lithosphere (97%); most occurs in association with igneous rocks of the earth's crust and NH_4 held within the lattices of such primary minerals as mica and feldspar (Stevenson, 1982). Most of the remaining 2% is found in the atmosphere. Amounts found in the hydrosphere (oceans, etc.) and biosphere (living matter) account for only 0.01% of the earth's supply. Ironically, it is these seemingly insignificant amounts of N associated with plant and animal activities that are extremely important in terms of ground water quality.

The following sources of N have been identified as those having an impact on ground water (MDA, 1990). Included is where within the document the reader can obtain more information.

Agronomic Inputs:	**Chapter**
1. Atmospheric sources.	
a) Biological fixation by legumes	G
b) Atmospheric fixation	L
c) Precipitation	L
2. Commercial fertilizers for crops	G
3. Soil organic matter	L
4. Crop residues	G
5. Manure	G
Other External Sources:	
1. Septic systems	I
2. Feedlots	H
3. Golf courses and other non-lawn green space	K
4. Lawn fertilizer applications	K
5. Municipal and industrial wastes	J
6. Landfills, Spills	L

POTENTIAL HEALTH AND ENVIRONMENTAL EFFECTS OF NITROGEN CONTAMINATED GROUND WATER

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Several potential adverse effects of N compounds on human and animal health and the environment have been identified, although many of these have not been verified. Methemoglobinemia is the only verified human health concern associated with NO_3. Other toxic effects postulated to be associated with NO_3 include central nervous system birth defects and carcinogenic effects. Animal health effects associated with N include methemoglobinemia and general malaise. Potential environmental effects of N include eutrophication of aquatic ecosystems, aquatic toxicity (primarily associated with NH_4), contribution to acid deposition, and partial depletion of stratospheric ozone by nitrous oxide (NO_2).

HUMAN HEALTH EFFECTS

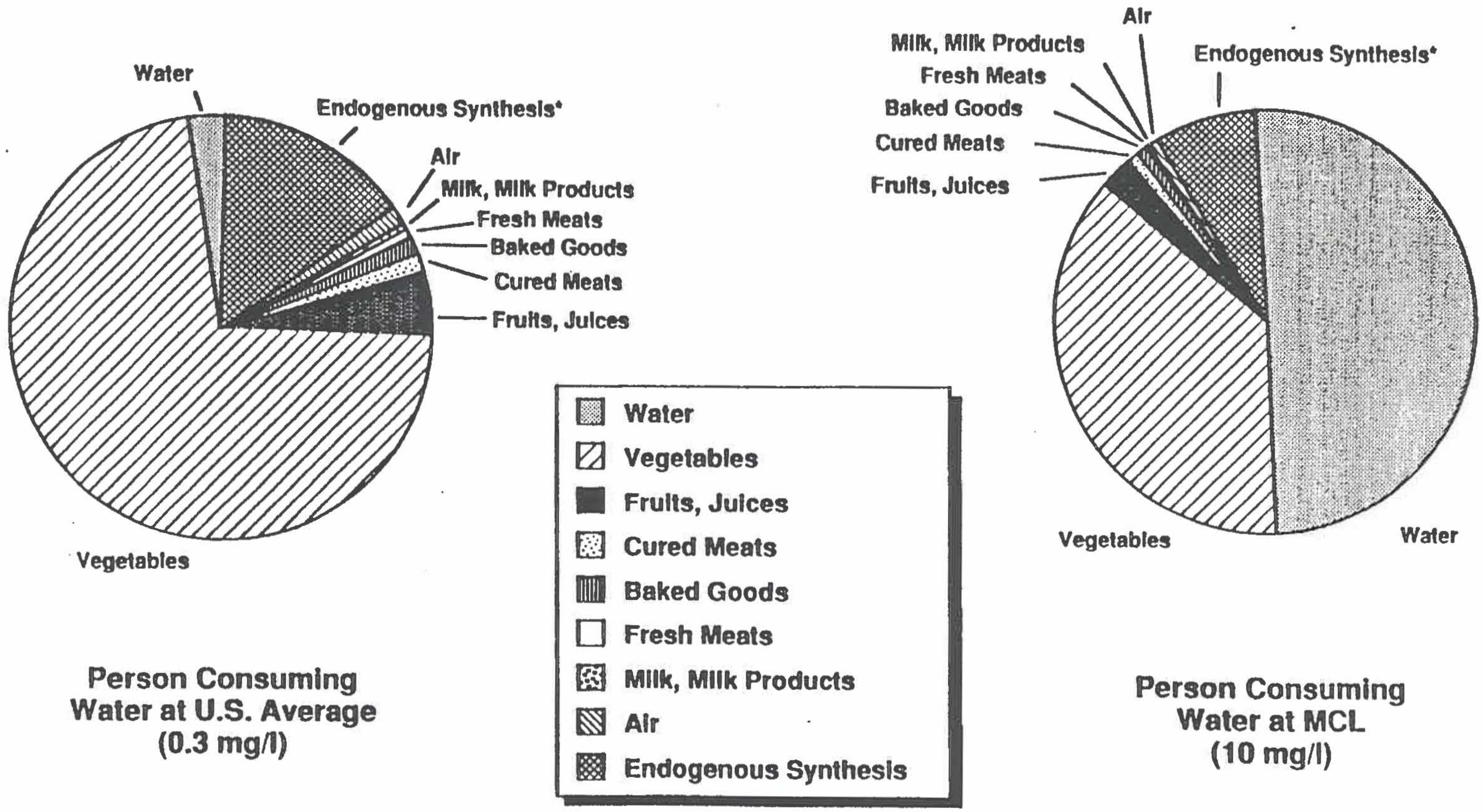
Contamination of ground or surface water with NO_3 presents a potential health threat to any human population which relies on that water resource as a source of drinking water. However, drinking water is just one of several possible NO_3 exposure routes for humans. Fruits, vegetables, cured meats, baked goods, fresh meats, milk products and air can be potential NO_3 sources. Drinking water becomes a significant component of total nitrate exposure when NO_3-N approaches or exceeds the 10 mg/L drinking water standard. Figure 1 displays the estimated contributions of the various NO_3 sources to the average adult total daily NO_3 exposure under two drinking water NO_3-N scenarios (EPA, 1991).

Methemoglobinemia

The primary health concern associated with exposure to NO_3 is methemoglobinemia, commonly known as the "blue baby disease". Infants of less than three months of age are most susceptible to this toxic effect (Craun et al., 1981), although individual adults may display increased susceptibility due to various factors (EPA, 1991). This condition occurs when NO_3 is reduced to nitrite (NO_2) in the stomach or oral cavity. Nitrite is absorbed into the bloodstream from the gastrointestinal (GI) tract. In the blood stream, NO_2 oxidizes ferrous iron (Fe^{+2}) of the hemoglobin heme group to ferric iron (Fe^{+3}), converting hemoglobin (Hb) to methemoglobin (metHb). Methemoglobin interferes with oxygen transport by irreversibly binding oxygen so that it is not released to deoxygenated tissues (Walton, 1951; Follet and Walker, 1989). If conditions are not conducive to the reduction of nitrate to nitrite during digestion, nitrate is metabolized and excreted without apparent adverse effect (Craun et al., 1981).

Increased susceptibility of infants to methemoglobinemia has several potential contributing factors. The pH of the infant GI tract (pH 4.6 to 6.5) is higher than that of adults (pH 2.0 to 5.0). Decreased stomach acidity (higher pH) allows NO_3-reducing bacteria to flourish in the stomach and upper intestine (Adam, 1980). In addition, fetal Hb is more susceptible to oxidation by NO_2. Fetal Hb comprises 60 - 80% of the Hb at birth and decreases to 20 - 30% by three months of age (Adam, 1980). Adult red blood cells contain enzymes which reduce metHb to Hb and thus maintain low blood metHb levels; these enzymes are

Figure A-2
Human Exposure to Nitrate Under Two Different Scenarios
From EPA (1991)



Assumptions: Average water consumption equals 2 liters per day
Average air intake equals 20 m^3 per day
Milligrams are expressed as nitrate-nitrogen

Source: NAS, 1981, with modifications
*Packer, 1990

reduced or lacking in infant red blood cells (Adam, 1980). Lastly, infants consume most or all of their nourishment in fluid form, consequently, they consume up to three times more fluid per body mass than adults. This high fluid intake rate insures that an infant consuming water from a contaminated source will experience a higher NO_3 exposure than an adult drinking water from the same source (Craun et al., 1980).

The affliction has dramatic symptoms in its acute stages. Freshly removed blood is chocolate brown in color. Afflicted infants develop a bluish to lavender color starting around the lips and extremities. Other symptoms are those related to oxygen deprivation and cyanosis including breathing difficulties, central nervous symptom effects (from mild dizziness and lethargy to coma and convulsions), cardiac disrythmias and circulatory failure (Walton, 1951).

Subacute effects of methemoglobinemia are not known. It has been postulated that nervous system damage could result from a chronic oxygen-depleted condition (Anonymous, 1988). However, adults with hereditary methemoglobinemia (with blood metHb concentrations of 10 to 25%) display no apparent adverse health effects, have uncomplicated pregnancies, and experience normal lifespans (EPA, 1991).

Reported methemoglobinemia incidence is fairly low. Between 1945 and 1974, approximately 2,000 cases of infant methemoglobinemia were reported in world literature (Shuval and Gruener, 1972); however, methemoglobinemia may often go unreported or may be misdiagnosed (Johnson and Kross, 1990). Rosenfield and Huston (1950) reviewed infant methemoglobinemia cases associated with increased NO_3-N in drinking water from private rural wells in Minnesota between 1947 and 1949. All affected infants had been fed either infant formula prepared with NO_3-contaminated water or cows' milk diluted with contaminated water. Over this three year period, 146 cases were documented including 16 deaths. None of the cases of infant methemoglobinemia occurred when the suspected drinking water source contained less than 30 mg/L NO_3-N. At the time of the study, awareness of the affliction was increasing among the medical community and the population at large. The authors witnessed a decrease in the number of cases during the course of the study; for the last six months of 1949, no cases were recorded (Rosenfield and Huston, 1950).

More recently, a single case of non-fatal infant methemoglobinemia occurred in Iowa in 1979, a non-fatal case occurred in Minnesota in 1979, and a fatal case occurred in South Dakota in 1986. In the Iowa case, the afflicted five-week old infant had been fed formula prepared with water containing 285 mg/L NO_3-N (Rajogopal and Tobin, 1989). The Minnesota case involved a 1 month old boy who had been fed water containing 90 mg/L NO_3-N.[2] In the South Dakota case, a six week old infant had been fed infant formula prepared with water containing 150 mg/L NO_3-N; the family physician failed to diagnose the problem (Johnson et al., 1987).

In Minnesota, no registry is maintained for methemoglobinemia cases. Iowa has maintained a methemoglobinemia registry since 1989; no cases had been reported in the first two years of its maintenance.[3]

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2. Personal communication with Jim Feddema, Minnesota Department of Health, St. Cloud, MN.

3. Personal communication with Russell Currier, Iowa Department of Health, Des Moines, IA)

Carcinogenic Effects

Nitrate and NO_2 have not been classified by the EPA as to human carcinogenicity (EPA, 1991). A number of studies have suggested an association between NO_3 intake and gastric and esophageal cancer. Most of the studies have correlated stomach cancer mortality or incidence against either national daily average NO_3 intake data or against average NO_3 concentration in regional drinking water sources. Many factors which are not accounted for in these studies may be important in determining the role of NO_3 in cancer etiology. These factors include age, smoking, medicinal use, dietary deficiencies of vitamins, antioxidants, and trace elements, dietary excesses and acidity of the GI tract. Other studies have actually reported a negative correlation between NO_3 intake and stomach cancer incidence (Follet and Walker, 1989).

It has been proposed that N-nitroso compounds may form in the acidic environment of the stomach by reaction between NO_2 and secondary or tertiary amines. N-nitroso compounds (nitrosamines and nitrosamides) have been classified by the EPA as probable human carcinogens (EPA, 1991). Nitrosamines and nitrosamides have possible human mutagenic, carcinogenic and teratogenic properties, although these has never been verified (EPA, 1991). Several N-nitroso compounds are proven animal carcinogens. Studies have also proven 77 of 100 N-nitrosamines mutagenic (Rajagopal and Tobin, 1989).

The proposed mechanism of N-nitroso compound formation and subsequent carcinogenic response in an exposed individual is a multi-step process. Nitrate is ingested and reduced to NO_2 in the stomach or oral cavity. Nitrite reacts with secondary or tertiary amines in the stomach to form N-nitroso compounds. Finally, the N-nitroso compound acts as a carcinogen at the target organ. These several steps make it difficult to establish a definitive cause-and-effect relationship between NO_3 in drinking water and cancer incidence (Follet and Walker, 1989). Animal feeding studies have shown tumor formation when animals are fed NO_2 and secondary amines, but not when they are fed NO_3 and secondary amines (Adam, 1980).

In general, the available information on NO_3, N-nitroso compounds and stomach and esophageal cancer is inconclusive. The World Health Organization and the National Academy of Sciences have both concluded that the evidence implicating NO_3, NO_2 and nitrosamines in the development of gastric cancer in humans is circumstantial (Black, 1989). In summarizing the current state of knowledge on carcinogenicity and NO_3 contaminated ground water, Black (1989) stated that "at this time, one can say that a set of mechanisms is known by which nitrate and nitrite may react in the body to cause stomach cancer. Whether the reactions are of significance at the levels involved in practice remains to be determined."

Central Nervous System Birth Defects

It has been proposed that NO_3-N in the drinking water of pregnant women is associated with central nervous system (CNS) birth defects. This relationship is based primarily upon the results of two recent epidemiological studies.

Dorsch (1984b), in a South Australia study, compared pregnant women receiving drinking water from three sources. The drinking water sources were defined, according to estimated NO_3-N as "low" (rainwater source, less than 1.1 mg/L NO_3-N), "medium" (surface water/groundwater source, between 1.1 and 3.4 mg/L NO_3-N) and "high" (> 3.4 mg/L NO_3-N). Dorsch found a statistically significant three fold increase in risk of CNS and musculoskeletal malformations of the fetus for women of the medium drinking water group and a statistically significant four fold increase in the high drinking water group. Several problems exist with this study. Perhaps the most important is that the observed correlation was actually between birth defect incidence and drinking water source, not NO_3-N concentration since these were only estimated. It is very possible that other differences existed between the water sources than nitrate concentrations.

Dorsch, in a later paper, concluded that NO_3 contribution of the drinking water was too small relative to the other dietary NO_3 sources to cause the observed difference in birth defects between the two groups of women. Dorsch also suggested that some other factor, correlated with the water source, may have been responsible (Dorsch, 1984a as cited in Black, 1989). Dorsch stated in a letter to Dr. Dennis Keeney[4] that "given the reservations raised in the two subsequent publications enclosed herewith, and the absence of substantiating findings from other studies, I now believe the evidence for a causal association (with nitrate) is tenuous at best" (Black, 1989).

Arbuckle et al. (1988) looked at clinical records of 130 CNS birth defect cases in New Brunswick, Canada from an 11 year period. The authors noted a moderate, but not statistically significant, increase in risk of CNS birth defect incidence for women drinking 5.9 mg/L NO_3-N water as compared to women drinking 0.02 mg/L NO_3-N water. For municipal drinking water sources and private spring drinking water sources, an increase in NO_3 exposure was associated with a decrease in risk of a CNS birth defect incidence; the association was not statistically significant. As in the Dorsch study, any association observed was actually a correlation between drinking water source and CNS birth defect incidence since the NO_3 concentrations were based upon analysis of a single water sample collected from the mother's prenatal address. Water samples were collected at the time that the study was conducted (approximately 1986) while the birth defect records came from between 1973 and 1983.

Since these two studies are the primary source of concern for the correlation between NO_3 exposure via drinking water and central nervous system birth defects, it seems that no significant evidence exists for support of this theory. At best, this is an area which warrants further research.

Derivation of Health Based Drinking Water Standard

The Environmental Protection Agency (EPA) has set the Maximum Contaminant Level Goal (MCLG) and the Maximum Contaminant Level (MCL) for drinking water at 10 mg/L NO_3-N. The MCLG, which is unenforceable, is established at the concentration at which no known or anticipated adverse human health effects occur and which allows for an adequate margin of safety. The MCL is established for public water supplies; it is based upon the MCLG, but also takes cost into account (EPA, 1991). The current Minnesota NO_3-N Recommended Allowable Limit (RAL), an unenforceable, health-based standard, is also 10 mg/L.

The 10 mg/L standard is based on methemoglobinemia incidence in infants as determined by a study conducted by the American Public Health Association

4. As cited in Black, 1989.

(AMPHA). The AMPHA study compiled data from 49 states for the period of pre-1945 to 1950. A total of 278 cases, including 39 fatal cases, was recorded. No cases were associated with NO_3-N concentrations below 10 mg/L; 2.3% of the cases were associated with concentrations between 10 mg/L and 20 mg/L. As a result of this study, the AMPHA recommended a 10 mg/L NO_3-N standard but noted that most methemoglobinemia cases were associated with concentrations greater than 40 mg/L (Walton, 1951). In 1962, the U.S. Public Health Service set a limit of 10 mg/L NO_3-N for domestic water supplies; this is the standard adopted as the national Primary Drinking Water standard, later adopted as the MCL (Rajagopal and Tobin, 1989). In summary, the 10 mg/L drinking water standard was determined based upon observations that no cases of infant methemoglobinemia have been observed below 10 mg/L.

ANIMAL (LIVESTOCK) HEALTH EFFECTS

Acute NO_3 toxicity in animals also takes the form of methemoglobinemia. Ruminants, such as cows, sheep and goats, are more susceptible to this affliction than non-ruminants. In these animals, the rumen, which is capable of digesting roughage, harbors NO_3-reducing bacteria which convert NO_3 to NO_2. In a healthy ruminant with an adequate diet, NO_2 is converted to NH_4 which may be used to build protein. If ruminants ingest large quantities of NO_3 quickly, it is possible for NO_2 to accumulate; this NO_2 may be absorbed through the oral and GI tracts into the blood where it will reduce hemoglobin (Hb) to methemoglobin (metHb). Studies have indicated that the rate of metHb formation is quicker for ruminants than for man, horse, or pig (Follet and Walker, 1989).

In single stomach animals, such as swine, poultry and horses, the reduction of NO_3 to NO_2 is not as rapid or as efficient as it is in the rumen, thus making non-ruminants less susceptible than ruminants to methemoglobinemia. However, in single stomach animals, NO_2-reducing microbes are also less prevalent, so that if NO_2 is produced, its conversion to NH_4 is less efficient than in ruminants. Horses are more susceptible than swine or poultry to methemoglobinemia due to their large cecum which acts similarly to a rumen, digesting roughage and converting NO_3 to NO_2 (Anderson et al., 1989).

Methemoglobinemia symptoms in animals include asphyxiation and labored breathing, rapid pulse, frothing at the mouth, lack of coordination, labored breathing, rapid heartbeat, abdominal pain and vomiting, convulsions, blue tint to the mucous membrane, muzzle and eyes, and chocolate brown colored blood (Anderson et al., 1989; Jackson et al., 1983). In pregnant cows, abortion may result (Anderson et al., 1989).

Chronic NO_3 poisoning in animals is difficult to diagnose because the clinical symptoms are those related to impaired animal health in general. Clinical symptoms include breathing difficulties, uneasiness, lowered blood pressure, reduction of milk secretion, avitaminosis A, thyroid dysfunction, and abortion. Other symptoms may include reduced rate of gain, poor growth, diarrhea, digestive disturbances, loss of young animals, arthritic or related conditions, abortions or still births (Ridder et al., 1974).

Just as for humans, water represents only one source of nitrate for animals. In some situations, feed (hays, forages, and silage) may contribute greater amounts of nitrate than drinking water (Bergsrud and Linn, 1990). For example, animals ingesting silage or pastured forage containing NO_3-N in excess of 0.2% will contract methemoglobinemia regardless of drinking water NO_3-N concentrations (Olson and Kurtz, 1982).

Several factors must be considered in determining safe drinking water levels of NO_3-N for stock animals. Both dietary and drinking water NO_3 sources must be considered; other important factors include animal species, quantity of water and feed ingested, and type of feed (Bergsrud and Linn, 1990). The National Academy of Science recommends that, in general, drinking water may be considered safe for livestock and poultry if it contains less than 100 mg/L NO_3-N and less than 10 mg/L NO_2-N (Anderson et al., 1989; Bergsrud and Linn, 1990). Table 1 provides recommendations on the use of water of varying NO_3-N concentrations for stock animals.

Table A-1
Use of Water with Known Nitrate Content
(adapted from Bergsrud and Linn, 1990)

NO3-N concentration (mg/L)	Recommendation
Less than 100	Experimental evidence indicates that water should not harm livestock or poultry.
100 to 300	This water should not by itself harm livestock or poultry. If hays, forages or silage contain high levels of nitrate, this water may contribute significantly to a nitrate problem in cattle, sheep or horses.
Over 300	This water should not be used; it could cause methemoglobinemia in cattle, sheep or horses. Because this level of nitrate contributes to the salts content in a significant amount, the use of this water for swine or poultry should be avoided.

Information on non-domesticated animals and NO_3 concentrations in drinking water is not widely available. However, the information presented above for non-ruminants may be considered as a point of reference for comparison.

ENVIRONMENTAL EFFECTS

Environmental effects of excess nitrogen to the environment are diverse. Nitrate can contribute to the eutrophication of water bodies. The ammonia (NH_3) and ammonium hydroxide (NH_4OH) forms of nitrogen are directly toxic to fish. Nitrogen oxides may contribute to stratospheric ozone depletion and, in the form of acid deposition, cause general ecosystem and material damage (Anderson et al., 1989; EPA, 1991).

Eutrophication is the increased rate of productivity in lakes, bays and slow moving streams due to excess nutrient loadings. Symptoms of eutrophication include algal blooms, algal mats, luxuriant development of selected aquatic macrophytes, and depletion of oxygen on lake bottoms (Anderson et al., 1989). In general, for freshwater bodies, phosphorus is the limiting growth factor rather than nitrogen. However, excessive loadings of N may, in certain cases, stimulate the growth of algae and contribute to eutrophication. Waters affected by urban activities tend to be N-limited; but overall, N-limited lakes are in the minority (EPA, 1991).

The total nitrogen to total phosphorus (TN:TP) ratio can be more important in determining eutrophication than the absolute nutrient loading. When the ratio is less than 10:1 (when P is high relative to N), N tends to be the limiting factor; in this case, any increased loading of N will enhance eutrophication. However, the optimal N:P ratio varies among the algal species contributing to eutrophication and ratios within a water body can vacillate naturally between seasons (EPA, 1991). Additionally, several basin, water, and limnological factors in addition to nutrient loadings may influence lake productivity (Anderson et al., 1989). These factors make it difficult to declare a standard concentration of NO_3-N below which no eutrophication will occur.

Ammonia gas dissolves in water to form ammonium hydroxide which dissociates to ammonium ion (NH_4) and hydroxide ion (OH). The distribution of these three species (NH_3, NH_4OH, and NH_4) for any given total concentration of NH_4-N depends upon temperature, pH, dissolved oxygen concentration and salinity. Ammonium is generally the predominant species in lakes and streams at normal physicochemical condition; it is usually rapidly taken up by aquatic plants and is almost harmless to aquatic animals. However, NH_3 and NH_4OH (referred to as un-ionized NH_3) are toxic to aquatic animals. Un-ionized NH_3, as a fraction of total NH_4-N, increases directly with temperature and pH. The degree of toxicity of un-ionized NH_3 is both species and age dependent. Salmonids (trout and salmon species) are particularly susceptible; young rainbow trout fry are killed if the total NH_4-N content is 0.3 mg/L, even under normal physicochemical conditions (pH 6-7, temperature 5-10 degrees Centigrade). Non-salmonids can generally survive concentrations ten times greater than those which are fatal to salmonids (Goldman and Horne, 1983). In general, in levels in excess of 1 mg/L NH_4-N are considered toxic to fish (EPA, 1991).

Nitrate is not toxic in the aquatic environment; warm water fish tolerate up to 90 mg/L NO_3-N and up to 5 mg/L NO_2-N; salmonids are more susceptible and can only tolerate up to 0.06 mg/L NO_2-N (Anderson et al., 1989).

Literature Cited

Adam, J.W.H. 1980. Health aspects of nitrate drinking water and possible means of denitrification (literature review). Water SA. 6:79-84.

Anonymous. 1988. Nitrate: rerun of an old horror. Health and Environmental Digest. 1:1-3.

Anderson, James L., James C. Balogh and Mark Waggoner. 1989. Soil conservation service manual: development of standards and specifications for nutrient and pest management; literature search and review. Soil Conservation Service. Washington D.C. 415 pp.

Arbuckle, Tye E., Gregory Sherman, and Paul Corey. 1988. Water nitrates and CNS birth defects: a population-based case-control study. Arch. Env. Health. 43: 162-167.

Bergsrud, Fred and James Linn. 1990. Water quality for livestock and poultry. Minnesota Extension Service, University of Minnesota. #AG-FO-1864-D.

Black, Charles A. 1989. Reducing american exposure to nitrate, nitrite and nitroso compounds: the national network to prevent birth defects proposal; comments from CAST. Council for Agricultural Science and Technology. Ames, Iowa.

Craun, G.F.; D.G. Greathouse; D.H. Gunderson. 1981. Methemoglobinemia levels in young children consuming high nitrate well water in the U.S. Int. J. Epidemiol. 10:309-317.

Dorsch, M. M.; I.C. Caler, D.M. Roder, and A.J. Esterman. 1984a. Birth defects and the consumption of nitrates in food and water in the lower south-east of south australia: an exploratory study. J. Food Nutr. 41:30-33. as cited in: Black, Charles A. 1989. Reducing american exposure to nitrate, nitrite and nitroso compounds: the national network to prevent birth defects proposal; comments from CAST. Council for Agricultural Science and Technology. Ames, Iowa.

Dorsch, Margaret M., Robert K.R. Scragg, Anthony J. McMichael, Peter A. Baghurst and Kenneth Dyer. 1984b. Congenital malformations and maternal drinking water supply in rural south Australia: a case control study. Am. J. Epidemiol. 119: 474-486.

Environmental Protection Agency (EPA); Nitrogen Work Group. 1991. Nitrogen action plan (draft). NTIS #40 0/3-90/003.

Follet, R.F. and D.J. Walker. 1989. Ground water quality concerns about nitrogen. p.1 - 22. In R.F. Follett. Nitrogen management and ground water protection. Elsevier. New York.

Goldman, Charles R. and Alexander J. Horne. 1983. Limnology. McGraw Hill. New York. 464 pp.

Jackson, Gary and Bruce Webendorfer, eds. 1983. Nitrate, groundwater & livestock health. University of Wisconsin-Extension. Madison, Wisconsin. #I-03-83-5MH.

Johnson, Carl J.; Pamela A. Bonrud; Terrance L. Dosch; Arthur K. Kilness; Kenneth A. Senger; Darron C. Busch; and Michael R. Meyer. 1987. Fatal outcome of methemoglobinemia in an infant. JAMA. 257:2796-2797.

Johnson, Carl J. and Burton C. Kross. 1990. Continuing importance of nitrate contamination of groundwater and wells in rural areas. Am. J. Industr. Med. 18:449-456.

Minnesota Department of Agriculture. 1990. Recommendations of the nitrogen fertilizer task force on the nitrogen fertilizer management plan. St. Paul, MN.

Olson, R.A. and L.T. Kurtz. 1982. Crop nitrogen requirements, utilization, and fertilization. in F.J. Stevenson, ed. Nitrogen in agricultural soils. Agronomy Series, No. 22. Am Soc Agron. Madison, Wisconsin. pp 567-604.

Rajagopal, R. and Graham Tobin. 1989. Expert opinion and groundwater quality protection: the case of nitrate in drinking water. Groundwater. 27: 835-847.

Ridder, William E., Frederick W. Oehme, and Donald Kelley. 1974. Nitrates in Kansas groundwaters as related to animal and human health. Toxicology. 2:397-405.

Rosenfield, A.B. and Roberta Huston. 1950. Infant methemoglobinemia in Minnesota due to nitrates in well water. Minnesota Medicine. 33:787-796.

Shuval, Hillel I. and Nachman Gruener. 1972. Epidemiological and toxicological aspects of nitrates and nitrites in the environment. AJPH. 62:1045-1052.

Stevenson, F.J. 1982. Origin and distribution of nitrogen in soil. In F.J. Stevenson (ed) Nitrogen in agricultural soils. Agronomy 22. Am. Soc. of Agron., Madison, WI.

Walton, Graham. 1951. Survey of literature relating to infant methemoglobinemia due to nitrate-contaminated water. AJPH. 41:986-995.

NITRATE IN GROUND WATER - EXISTING CONDITIONS AND TRENDS

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INTRODUCTION

Minnesota does not have a statewide ground water monitoring program in place designed specifically to determine the extent and trends of nitrate concentrations. For purposes of this report, existing water quality monitoring data sets were collected and examined. Nitrate results from individual wells were obtained and analyzed from seven computerized data sets meeting minimum criteria. Nine other data sets were examined and summarized from literature and personal communication with representatives of groups collecting the data.

Upon casual examination, it would appear that there is an abundance of data to make reasonable estimations of the current nitrate status in Minnesota. The estimated number of permanent residence domestic wells in Minnesota is 410,000.[1] Nitrate data from nearly 50,000 wells is available from various data sets. However, nearly half of this information was produced by questionable or unknown analysis methods. Fourteen data sets, representing 26,340 wells, have EPA approved laboratory analysis methods. Great differences exist between data sets regarding sampling purpose, field methodologies, areas sampled, years and frequency of sampling, data management, and documented well location and construction information. These differences limit the utility of the data in assessing statewide conditions.

Much of the data examined in this report is from domestic water supply wells, commonly sampled through county or regional efforts. These wells are most frequently tested to simply determine if the source is fit for human or animal consumption at the time of sampling. Domestic water supply sampling for nitrate is required following the completion of a new well. Data sets based on private new well construction will likely be biased towards lower nitrate concentrations. Domestic water supply testing conducted as part of county sampling programs usually tests wells with a variety of ages, depths, construction techniques and locations. Biases may exist when using private well data, rather than specially designed and installed monitoring wells, to assess ground water quality. Poor construction, lack of maintenance, and nearby pollution sources may lead to direct contamination of well water by surface or interflow water. Domestic well owners may be more likely to commit the time, effort and expense of having their wells tested if they suspect problems. Similarly, counties with an active involvement in nitrate testing are more often those with higher occurrences of contamination. On the contrary, well owners suspecting problems may be reluctant to submit water samples through government sampling programs.

There are many physical factors that will influence the nitrate concentration of water obtained in a given well. Vertical and horizontal variability can be extremely high. A very important variable controlling nitrate concentrations is the geologic and hydrogeologic conditions of the area surrounding the well.

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[1] Extrapolated from 1980 census information.

Several studies have been conducted to help us understand some of the geologic complexities affecting nitrate in Minnesota. A discussion of factors and complexities affecting nitrate concentrations precedes a description of existing nitrate data sets and discussion of existing conditions and trends.

Despite the number of caveats, and keeping these fully in mind throughout the interpretation process, a greater understanding of nitrate conditions in Minnesota can be gained by examining the assembly of data sets.

FACTORS AND COMPLEXITIES AFFECTING NITRATE CONCENTRATIONS

A nitrate (NO_3) level in any given sample of well water is the combined effect of numerous factors, including surrounding land use, soils, hydrogeology, climate, well location and construction. Therefore, the concentration of NO_3 in any given well or group of wells may or may not represent the NO_3 conditions of an area. The number of wells needed to adequately assess ground water NO_3 conditions will depend on the spatial and temporal variability of the aforementioned factors. In studies of surficial sand aquifers in Benton and Stearns counties, extreme variability of ground water NO_3 was found in areas of about one square mile (Magner et al., 1990A and Magner et al., 1990B). Nitrate-N concentrations ranged from 0.01 to over 30 mg/l in each of these two areas.

There are hundreds of studies from around the country that could be referenced for each topic in the following discussion. Some examples from Minnesota studies are included to help illustrate the complexities involved in assessing nitrate conditions in ground water.

Land Use and Management

Land use and its associated N inputs can greatly affect ground water NO_3 concentrations. The potential for various sources to impact ground water is discussed in chapters G, H, I, J, K, and L of this report. Both the land use in the immediate vicinity of the well and the broad scale land use in the area of the well can affect NO_3 concentrations. The dimension of the plume of NO_3 contamination also varies by land use. Land application of fertilizers can produce a relatively wide plume of elevated NO_3 water. This plume may widen somewhat as it moves away from the field. In these areas there will be less chance of mixing of high and low NO_3 waters. Septic systems will produce a relatively narrow plume that will gradually widen with distance from the drainfield. The further from the septic system that a well is, the greater chance that high NO_3 water has mixed with lower NO_3 water and the plume has diffused.

While land use in itself can affect NO_3 concentrations in ground water, more important is the management of the land. In residential areas, the density of septic systems is an important land management variable. In agricultural areas, the crop type, irrigation management, nitrogen fertilizer and manure rates and management are important variables affecting ground water quality.

Ground Water Flow Hydraulics

Direction and rate of ground water flow in relation to the N source is an important variable. Comparisons of NO_2+NO_3-N concentrations in major agricultural areas in central Minnesota sand plain aquifers were made using 57 up-gradient and 46 down-gradient wells sampled in four counties (Myette, 1984). County medians of wells up-gradient of agricultural fields ranged from 0.1 to 0.7 mg/l, whereas county medians of down-gradient wells were between 6.0 and 9.5 mg/l.

The natural flow direction can be altered by pumping of wells. Magner et al. (1990A) noted seasonal ground water flow direction reversals near an irrigation well. Domestic wells and municipal wells can also create localized zones of ground water flow reversal.

Ground water flow and hence contaminant transport is affected by such factors as hydraulic gradient, aquifer thickness, hydraulic conductivity, and porosity. These factors affect the degree of mixing or dilution of NO_3 enriched water with other water, the penetration depth of a NO_3 plume and the rate of movement of the NO_3 plume. The vertical flow component is a very important factor affecting nitrate transport within aquifers and nitrate concentrations in well water.

Short Term Nitrate Fluctuations

A well down-gradient of a major N source may not necessarily be impacted by NO_3. Depending on the relationship between the residence time of the water in the well and the date of introduction of the N source, it is very possible that NO_3 from an upgradient source will not be found in a well water analysis. Also, seasonal releases of N can create pulses of high NO_3 water. Wells downgradient of such sources can have great temporal variability in NO_3 concentrations. Short term temporal variability can be caused by non-constant N releases, seasonal climate changes, precipitation patterns, and complex relationships between soil and aquifer-hydraulic characteristics. The degree of short term nitrate fluctuations will vary greatly from well to well depending on the well construction and aquifer characteristics, including the residence time of the water in the well. Wall et al. (1989A) reported NO_3-N concentrations in one Big Stone County well to decrease from 30 to 18 to 5 mg/l in less than a year. Nitrate-N concentrations in other wells in the same area varied by less than 0.2 mg/l over that same time period. Anderson (1987) reported NO_3-N concentrations in a surficial sand aquifer well to decrease from 72 mg/l in May 1983 to 18 mg/l in May 1984.

Ground Water Chemistry (Denitrification)

Nitrate losses through denitrification can occur when the ground water chemistry is conducive for such a reaction. Important factors for denitrification are low redox potential, low dissolved oxygen, and high organic carbon content. Chemical conditions necessary for denitrification can vary greatly among aquifers and can change along the flow path within an aquifer. Several studies in Canada and the United States have shown denitrification to be responsible for significant NO_3 losses under certain ground water chemical conditions (see Chapter C). In Winona County, Wall and Regan (1991) found lower NO_3

concentrations in wells showing a greater potential for denitrification compared to wells with water chemistries indicating less of a potential for denitrification.

Geologic Stratigraphy

The geologic zone from which water enters a well greatly influences NO_3 levels in the well water. Layers of clay from glacial till deposits, shale units, siltstone units and other lower permeability layers can greatly retard movement of water, thereby protecting underlying aquifers from NO_3. In Winona County, Wall and Regan (1991) found lower NO_3 concentrations in areas under shale formations and units of low permeability siltstone compared to areas with no shale or siltstone. In several studies throughout different areas in Minnesota, NO_3 concentrations were found to be much lower below glacial till deposits than overlying surficial aquifers (Magner et al., 1990A; Magner et al., 1990B; Wall et al., 1989; Klaseus and Buzicky, 1988). At a nested monitoring well site downgradient of an irrigated field, Wall et al. (1989A) found NO_3-N concentrations averaging 77 mg/l in five samples taken in a surficial sand aquifer well. Below ten feet of till at the same site NO_3-N concentrations averaged 1.7 mg/l. Glacial stratigraphy can be quite complex and "protective" till units are often localized. In areas where till does not exist, water can move deeper into the aquifer and move laterally below nearby layers of clayey till, thereby impacting water below this till.

Well Depth

There are a number of important factors associated with well depth that can affect well water NO_3 levels, including the depth of casing, total well depth, interrelationship between well depth and stratigraphy and depth below the water table. A deeper well will more often have a lower NO_3 concentration than a shallower well at the same location. This is often due to the deeper well penetrating a lower permeability unit. In addition, as water moves downward in the aquifer dilution, dispersion and sometimes denitrification can contribute to lower nitrate concentrations. However, well depth is less likely to correlate with nitrate in a region where topographic or stratigraphic variability is great. Wall et al. (1989B) and Wall et al. (1991) observed no relationship between NO_3 levels and well depth from bedrock aquifers in wells scattered throughout western Winona County. Shallower wells located in valleys can often penetrate deeper into aquifers than deeper wells located on ridgetops. Also, a 250 foot well in a karst area will be much more susceptible than a 250-foot well in a glacial drift aquifer. In most cases, a 150 to 250-foot well in a glacial drift aquifer well will penetrate at least one layer of clayey material, thereby protecting the well from rapid transport of surface contaminants. In a karst area, a 250-foot well may only penetrate shallow soils and fractured carbonate bedrock offering very little protection from surface contaminants.

In unconfined aquifers, deeper wells have been shown to be more likely to have lower NO_3 than shallow wells at the same location. Two adjacent (nested) wells screened at different depths in unconfined aquifers were sampled at nine sites in central Minnesota (Klaseus et al., 1988). Average NO_3-N concentrations were 7.4 and 1.9 mg/l for the shallow and deep wells, respectively. During a study in Douglas, Pope, and Stearns counties, Anderson (1987) found higher NO_3

concentrations in eight out of eleven wells screened at the top of the water table compared to wells screened ten feet below the water table. At the Staples Irrigation Center, NO_3 concentrations were found to decrease quite dramatically with increasing depth below the water table (Myette, 1984). The stratification appeared to be greatest during periods of little recharge. Myette (1984) sampled 106 wells in Hubbard, Morrison, Ottertail, and Wadena counties and found that the mean NO_3-N concentration was 15 mg/l at the top of the aquifer and 0.01 mg/l at the bottom of the aquifer.

In some instances, NO_3 in deeper unconfined wells has been higher than shallow wells. As NO_3 enriched water moves further from a source it will often move deeper into the aquifer along with the movement of the ground water. Therefore, deeper wells further down gradient from a source could have higher NO_3 than shallower wells. In Benton County, Magner et al. (1990A) found two domestic wells 50 and 60 feet deep to have average NO_3-N concentrations of 26 and 23 mg/l, respectively. Shallower monitoring wells (20 and 39 ft.) placed next to the domestic wells had average NO_3-N concentrations of 2.4 and 1.6 mg/l. The study concluded that ground water deeper within the aquifer was water which had originated about a half mile away at the site of an irrigated field and water in the upper part of the aquifer had recharged closer to the wells in areas of trees and grass. In a nested well site along a lake in Stearns county, Magner et al. (1990B) found NO_3-N ranging from 23-27 mg/l in a deeper well (29 ft) and 8-10 mg/l in an adjacent shallow well (19 ft). Lake/ground water interaction (bank storage) and lower NO_3 inputs in the area surrounding the well was likely responsible for the lower NO_3 in the shallower well. Anderson (1987) noted that where higher NO_3 was found in the deeper wells at nested well sites, the wells were located near intermittent ponds. Anderson proposed that these ponds may drain rapidly through the sandy soils, displacing or mixing with higher NO_3 water in the aquifer.

Well Construction and Type of Well

A number of well construction factors can affect well water NO_3 levels. Wells with no grout or a poor seal around the casing, holes in the casing, or uncased wells provide direct conduits for water to move from the soil or upper parts of the aquifer to points deeper within the aquifer. These conditions often exist in older wells, and therefore the age of the well is an important factor. If an old well is located near a N source (i.e. feedlot or septic system) water quality problems in certain hydrogeologic settings will likely be further exacerbated. This problem is more likely to occur with domestic wells. Some of the other issues associated with sampling domestic wells were discussed in the introduction to this chapter. A municipal or irrigation well withdraws water from a larger area than monitoring wells and domestic wells. The distance from which a well draws in water may affect NO_3 concentrations.

Community wells are monitored more frequently and are more likely to be replaced if high NO_3 is detected than most other wells.

NITRATE RESULTS FROM EXISTING DATA SETS - CURRENT CONDITIONS

No sampling program has been undertaken to specifically assess statewide NO_3 conditions and trends. An abundance of ground water NO_3 data has been obtained from numerous individual studies and sampling programs in Minnesota during past years. A majority of this data has been collected since the late 1970's. One of the goals of this study was to summarize NO_3 results from most of the major existing data sets generated in recent years. Many of the data sets are stored on computer data bases, but have not been recently examined or reported. Other NO_3 data sets are not as readily accessible, but have been recently examined and reported by others.

Ground water nitrate results from various data sets are described in three sections of this report. The first section is a more in-depth analysis of readily available data that also met certain other conditions. Raw data were obtained and analyzed for this first section. The second section summarizes the results from other data sets that either were 1) not readily accessible, 2) did not meet the certain conditions or 3) were already described in recent reports. Towards the end of this chapter, changes in NO_3 concentration with time are described for two data sets.

Great differences exist between these data sets regarding sampling purpose, field methodologies, types of wells sampled, areas sampled, years and frequency of sampling, data management, and documented well location and construction information. Several of the data sets are from wells in geologically sensitive agricultural areas. Other data sets are from newly constructed wells. Biases exist with most of these data sets.

An Analysis of Nitrate Results from Seven Selected Data Sets

Nitrate data and associated well information were obtained and analyzed from readily accessible data bases that met the following other conditions:

1. Analysis methods are EPA approved (cadmium-reduction, ion specific electrode, ion chromatography, automated hydrazine or Brucine Sulfate - with approved quality assurance/quality control) - an exception to this is the County Well Index which may have some data generated by non-certifiable methods.

2. Nitrate data were collected since January 1, 1978. (The MPCA ambient ground water monitoring program began in 1978. Also, data collected since this date should be fairly reflective of recent conditions).

3. Site locations were identified to a minimum of township, range, and section.

4. Wells were not part of a sampling program aimed at assessing water quality around point sources of pollution.

Data from the following sampling programs and data bases met the above conditions and will be discussed in this section of the report:

1. U.S. Geological Survey collected data stored in WATSTORE (USGS)
2. MPCA ambient ground water monitoring program (Ambient)

3. County Well Index on file at Minnesota Geological Survey (CWI)
4. MPCA Nonpoint Source Studies (NPS)
5. U.S. Forest Service data stored in STORET (NFS)
6. Southeastern Minnesota Regional Laboratory (seven-county sampling program) (SEMN)
7. MDA pesticide/nutrient monitoring program (MDA)

Characteristics of Wells from the Seven Selected Data Sets

The number of analyses and wells, years of sampling, well location, age of well and well depth information is summarized for each of the seven data sets in Tables B1 to B6. This section is followed by descriptions of each sampling program and associated NO_3 results.

Number of Analyses and Wells

The number of analyses and wells varied tremendously between sampling programs (Table B-1). These data were provided to MPCA during the summer of 1990 and thus information obtained since that time are not included in this analysis. The township range and section (TRS) was known for most state and federal program sampling sites. However, about one-third of SE MN sampling analyses did not have associated TRS information and thus are not included in the summary of NO_3 results.

Table B-1 Number of NO_3 analyses and wells with NO_3 analyses for each of the selected data sets

Data Set	# NO_3 Analyses Total (as of 7-90)	# Analyses Since 1978	# Analyses Since 1978 TRS Known	# Wells Since 1978 TRS Known
County Well Index (CWI)	11,073	9,600	9,291	8,085
MDA Pest/Nutrient (MDA)	444	444	413	95
MPCA Ambient (Ambient)	1,032	990	990	484
MPCA Nonpoint Source (NPS)	384	384	384	71
U.S. Forest Service (NFS)	623	502	502	114
U.S. Geological Survey (USGS)	3,247	2,226	2,226	841
SE Minn. Reg. Lab (SEMN)	8,525	8,523	5,727	4,728
Total	25,328	22,669	19,533	14,418

Since many wells were sampled more than once, the number of NO_3 analyses column is greater than the number of wells column in Table B-1. **Where more than one NO_3 analysis was available for a given well, the average concentration was used to represent that well for this study.** The number of wells sampled varies greatly by program. From the seven data sets, county Well Index (CWI) and S.E. Regional Lab files are the two data sets with the greatest number of wells. The MPCA-NPS and MDA data sets are relatively small.

Well Locations by County

The number of wells in each county sampled by the various programs is listed in Table B-2. Sixteen counties had at least 150 wells meeting the criteria and seven counties each had over 1000 wells (Anoka, Brown, Blue Earth, Goodhue, Nicollet, Olmsted, and Washington counties). The data from CWI are most representative of six counties, each with NO_3 from over 600 wells. The MPCA ambient and USGS sampled wells are more evenly distributed throughout the state. There are, however, 29 counties with less than 10 wells included in the seven data sets.

A map was generated to show the distribution of well locations from the combined seven data sets (Figure B-1). A majority of the wells from these data sets are in central Minnesota, the Twin Cities area, and southeastern Minnesota. Few wells are located in Northern Minnesota. South-central and south western Minnesota do not have many wells in the selected data sets, but do have county sampling program results which are described later in this chapter.

Date of Sampling

The number of analyses found for each sampling program each year (1978 to 1990) is shown in Table B-3. In general, more data were available for the period 1985 to 1989.

Table B-3 Number of samples collected each year since 1978 for the selected data sets.

SAMPLE YEAR	CWI	MDA	AMBIENT	NPS	NFS	USGS	SE MN	TOTAL
1978	365		138		44	10	1	558
1979	414		100		24	224	0	762
1980	348		61		42	456	0	907
1981	482		90		62	304	15	953
1982	436		94		63	83	53	729
1983	616		113		62	248	247	1,286
1984	740		149		64	244	31	1,228
1985	1,043		66		54	190	562	1,915
1986	2,717		63		59	95	503	3,437
1987	1,159	10	18	101	16	120	676	2,100
1988	660	153	60	248	12	108	1,576	2,817
1989	311	127	38	35		132	1,297	1,940
1990		123				3	766	892
Total	9,291	413	990	384	502	2,217	5,727	19,524

Figure B-1

Number of Wells per Township

from the Seven Selected Data Sets

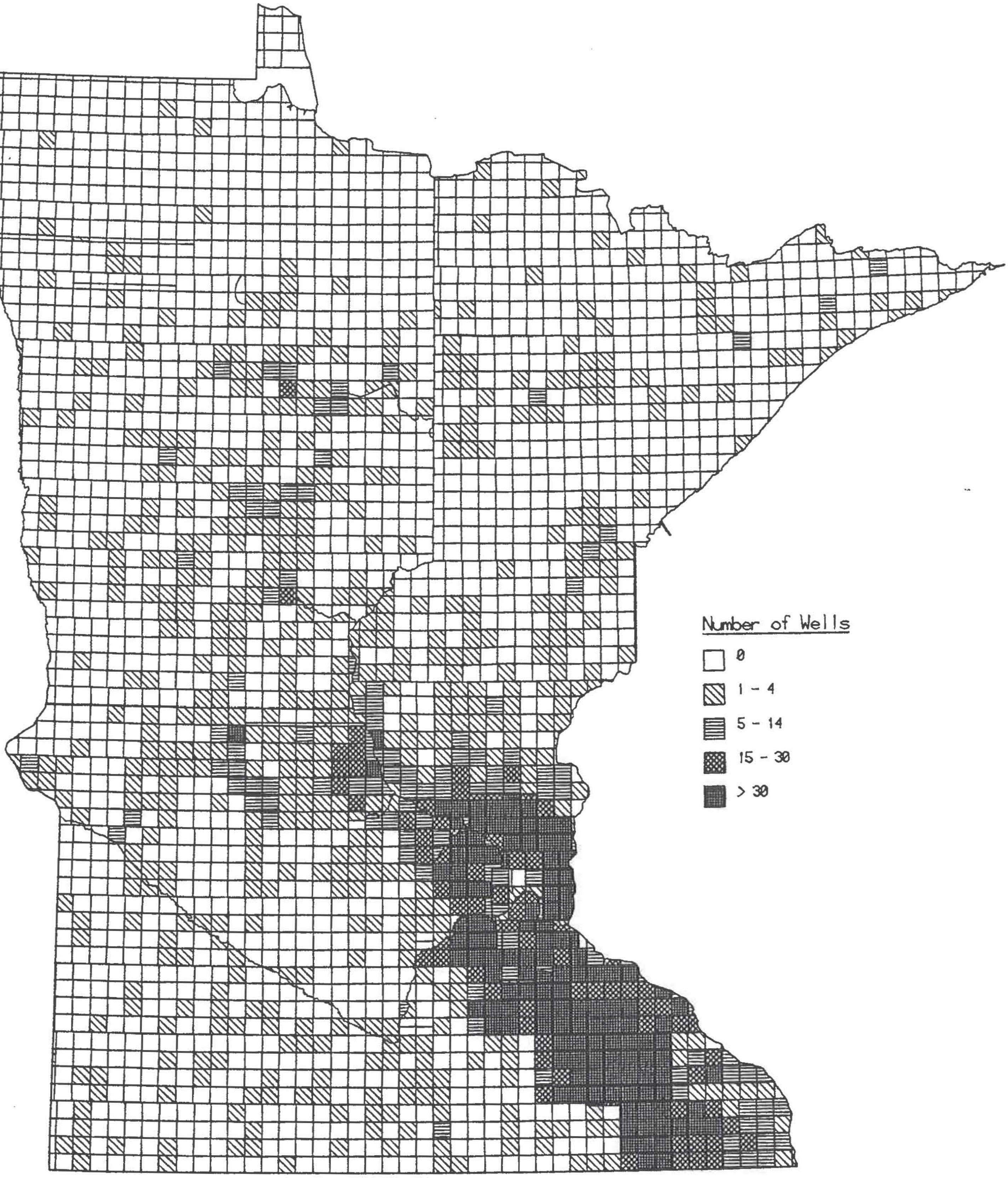

Table B-2 Number of wells in each county from the seven selected data sets.

CO. #	CO. NAME	CWI	MDA	AMBIENT	NPS	NFS	USGS	SE COS	TOTAL
1	AITKIN	7		1					8
2	ANOKA	1512		13			72		1597
3	BECKER	8	5	1			23		37
4	BELTRAMI	7		3		9	46		65
5	BENTON	67	1	3	27				98
6	BIG STONE			3	16		4		23
7	BLUE EARTH	22		3			4		29
8	BROWN	3		4			1		8
9	CARLTON	8		3			26		37
10	CARVER	27		6					33
11	CASS	17		6		26	9		58
12	CHIPPEWA	1		5			2		8
13	CHISAGO	64		1			11		76
14	CLAY	3		3					6
15	CLEARWATER			1			14		15
16	COOK	3		5		21			29
17	COTTONWOOD			9			1		10
18	CROW WING	23		4			1		28
19	DAKOTA	762	3	28			11		804
20	DODGE			2			2	553	557
21	DOUGLAS	7	2	1			14		24
22	FARIBAULT	8		4			6		18
23	FILLMORE	50		28			1	834	913
24	FREEBORN			1			3		4
25	GOODHUE	132		21			3	1018	1174
26	GRANT			1			3		4
27	HENNEPIN	638		25			62		725
28	HOUSTON	19	1	6				131	157
29	HUBBARD	13	5	8			44		70
30	ISANTI	78	3	4			16		101
31	ITASCA	9		1		28	8		46
32	JACKSON		1	3			1		5
33	KANABEC	16		1			7		24
34	KANDIYOHI	17	2	2			20		41
35	KITTSON			1					1
36	KOOCHICHING			2					2
37	LAC QUI PARLE			2			2		4
38	LAKE			4		14	2		20
39	LAKE OF WOODS			1					1
40	LE SUEUR	6		2			6		14
41	LINCOLN			1					1
42	LYON	3		4					7
43	MCLEOD	3		4					7
44	MAHNOMEN	2		1			6		9
45	MARSHALL			2					2
46	MARTIN	1		3					4
47	MEEKER	17		2			1		20
48	MILLE LA	37		2					39
49	MORRISON	22	3	12			29		66
50	MOWER			9			4		13
51	MURRAY			2					2
52	NICOLLET	14		1			2		17
53	NOBLES		1	5					6
54	NORMAN			2			2		4
55	OLMSTED	883		4				1084	1971
56	OTTER TA	7	7	8			38		60
57	PENNINGTON			4					4
58	PINE	17		5			14		36
59	PIPESTONE		2	2					4
60	POLK	7		2			1		10
61	POPE		7	7			72		86
62	RAMSEY			23			25		48
63	RED LAKE			1					1
64	REDWOOD	3	3	10			3		19
65	RENVILLE	1		2			4		7
66	RICE	25		6			2	497	530
67	ROCK		2	4					6
68	ROSEAU			2					2
69	ST. LOUI	4		14		16	20		54
70	SCOTT	972		8			2		982
71	SHERBURN	282	13	7			45		347
72	SIBLEY			2			1	127	130
73	STEARNS	265	6	3	15		53		342
74	STEELE			4			4		8
75	STEVENS		5	1			3		9
76	SWIFT	2	6	9			10		27
77	TODD	6		3			12		21
78	TRAVERSE			1					1
79	WABASHA	95		3			2	679	779
80	WADENA	1	7	15			45		68
81	WASECA			3			3		6
82	WASHINGT	1551		21			4		1576
83	WATONWAN			3			1		4
84	WILKIN			1			2		3
85	WINONA	115	10	28	12		5		170
86	WRIGHT	242		4	1				247
87	YELLOW M	2		3			1		6

Aquifers

Aquifer information was known for many wells from CWI, MDA, Ambient, NPS, and USGS data (Table B-4). A description of the major Minnesota aquifers is found in Appendix A. The MDA, USGA and NPS wells were primarily from surficial drift aquifers. CWI, Ambient, and USGS wells represent many different aquifers. Aquifers/formations with the most data include glacial drift aquifers, Prairie du Chien, Jordan, St. Peter and Franconia Formations.

Table B-4 Number of wells in various aquifers for each of the selected data sets. Some wells were in aquifers not listed in this table.

AQUIFER	CWI	MDA	AMBIENT	NPS	NFS	USGS	SE MN	TOTAL
Surficial Drift	379	79	96	51		363		1119
Buried Drift	517	1	79	8		60		680
Unspec.Glac.>50 Ft	26							26
Cretaceous	1		31			10		42
Cedar Valley-Maqu. Dubuque-Galena			26			9		35
Decorah-Plattville-Glenwood	111		2			17		126
Platteville-St.Peter and St. Peter Prairie	39		1					40
St. Peter	311		47			21		379
Prairie Du Chien	1118	13	22			3		1156
Prairie Du Chien-Jordan			25	7		31		63
Jordan	492	1	43	4		6		546
St. Lawrence & St. Lawr-Franc.	68		5			1		74
Franconia-Ironton-Galesville	229		29			2		260
Mt.Simon/Hinckley	13		28			13		55

AQUIFER	CWI	MDA	AMBIENT	NPS	NFS	USGS	SE MN	TOTAL
No. Shore Volcanics			8			1		9
Biwabik Iron Fmt.			7			1		8
Sioux Quartzite			8					8
Precambrian Other			13			12		25
Total	3304	94	470	70	0	713	0	4651

Age of Well

Approximately three-fourths of all wells in the seven data sets for which age was reported were constructed since the well code went into effect in 1974 (Table B-5). Nitrate analyses in CWI tend to be mostly from wells constructed since 1974. Fifteen percent of the wells analyzed by S.E. regional lab for which the date of construction was known were constructed before 1945. It is likely that well owners not knowing the date of construction have older wells in general compared to the owners knowing the date of construction. Therefore, actual distributions of well age probably have greater percentages of older wells compared with the percentages shown in Table B-5.

Table B-5 Number of wells in various construction date categories for the selected data sets.

YEAR CONSTRUCTED	CWI	MDA	AMBIENT	NPS	NFS	USGS	SE MN	TOTAL
< 1945	36		21				257	314
1945-1959	108		93	1			237	439
1960-1974	327		124	4			468	923
> 1974	5,769		181	37			818	6,805
Total	6,240	0	419	42	0	0	1,780	8,481
Missing	1,865	95	65	29	114	841	2,948	5,957

Well Depth

The distribution of well depths for the various data sets is listed in Table B-6. The NPS, MDA, and USGS data sets each have over half of all wells, less than 50 feet deep.

Table B-6 Number of wells in various well depth categories for the selected data sets.

WELL DEPTH (FT.)	CWI	MDA	AMBIENT	NPS	NFS	USGS	SE MN
< 50	175	70	46	35		402	133
50 - 99	1,560	5	63	15		97	324
100 - 149	1,433	0	53	2		41	294
150 - 199	1,396	0	44	1		27	270
200 - 299	943	5	61	7		26	445
300 - 400	605	8	63	12		24	399
> 400	248	1	118	3		39	277
Total	6,360	89	448	68		656	2,144
Missing	1,725	6	36	3		185	2,584

Descriptions and Results of Individual Data Sets

U.S. Geological Survey Collected Data

The United States Geological Survey (USGS) has been involved in over 25 Minnesota ground water studies, most of them conducted since the 1970's. References are provided at the end of this chapter that list the reports where these data and the associated geology are described. Sampling frequencies varied by project, with most wells having been sampled several times over a two-month to two-year period. Most of the data for the USGS studies were collected in an effort to define hydrogeologic and water quality characteristics of major aquifers throughout the state, focusing mostly on sand plain aquifers.

Observation wells were installed for many of the studies and were often installed near the top of the water table. Other studies have utilized domestic and public wells in order to obtain a general understanding of water quality within a given area and aquifer. Wells were sampled in various land use settings including agricultural, residential, and forested/grassland. Over seventy percent of the wells are surficial drift aquifer wells, most of which are less than 50 feet deep.

All laboratory analyses have been performed at the USGS Central Laboratory in Denver, Colorado using cadmium reduction techniques. A total of 841 wells met the data selection criteria.

The NO_3-N concentrations from the USGS data set are presented in Table B-7 and Figure B-2.

Table B-7 Nitrate-N in USGS wells meeting selection criteria.

Number Of Wells	Total # Analyses	Mean[1] NO_3-N mg/l	Median[2] NO_3-N mg/l	% Wells With NO_3-N Conc. (mg/l)			
				0-1	1.01-5	5.01-10	> 10
841	2,226	3.5	0.2	61	18	10	11

Nitrate data from USGS is scattered around many areas of the state (64 counties), but is not found in southwestern and the very northwestern counties in Minnesota. Clusters of data shown in Figure B-2 are primarily from individual sand plain studies. From this data set, the percent of wells with NO_3-N exceeding 1, 10, 20, and 30 mg/l were 39, 11, 4 and 1.2 percent, respectively.

Elevated NO_3 levels from the USGS wells are found throughout many counties in the state, often in clusters, such as in southern Hubbard County, central Ottertail, eastern Pope and western Sherburne counties. Other areas show very little NO_3 (in USGS wells), such as in Beltrami, Clearwater and Swift counties.

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[1]The mean is the same as the average concentration (i.e. the sum of all concentrations divided by the number of wells).

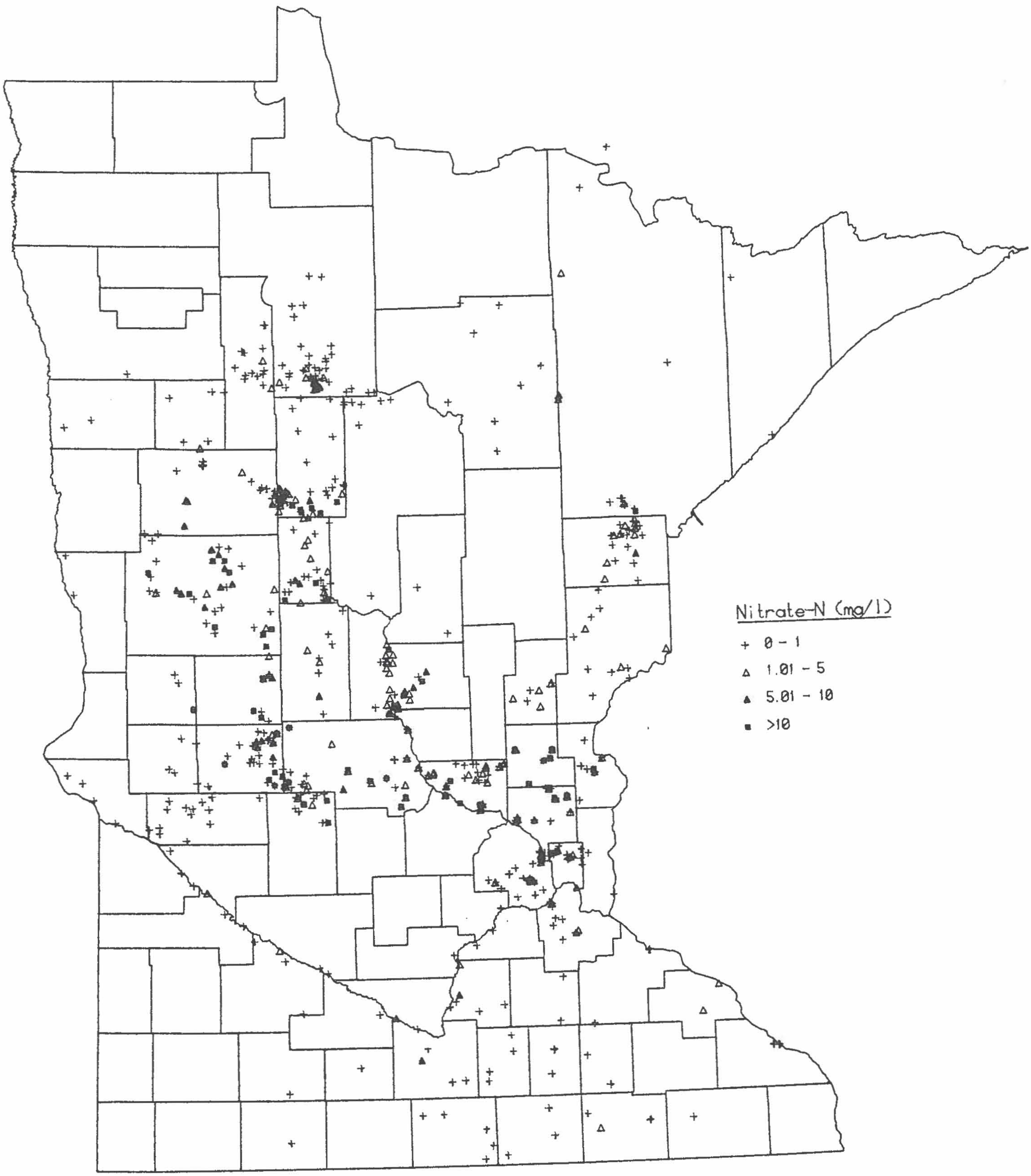
[2]The median is the middle value (i.e. half of the wells have concentrations greater than the median and half less than the median).

Figure B-2

U.S. Geological Survey

Well locations and Nitrate levels

(1978 - 1989)



Minnesota Pollution Control Agency
Ambient Ground Water Quality Monitoring Program

The overall goal of the Minnesota Pollution Control Agency (MPCA) ambient ground water quality monitoring program was to define the time and space variation of water quality in the principal aquifers of the state. Samples have been collected by trained MPCA staff using a stabilization procedure developed in conjunction with the United States Geological Survey (USGS). The USGS, through an October 1977 agreement, provided the MPCA with the design of the ambient ground water quality monitoring network. Data from a total of 484 wells (domestic, public, industrial and municipal) and springs sampled between 1978 and 1989 were obtained. Most of these stations have been sampled at least twice, for a total of 990 analyses since 1978. Each of Minnesota's 87 counties is represented by at least one sampling station.

Stations were selected for the network on the basis of aquifer, well construction, and separation from known or possible sources of ground water contamination. All analyses were performed at the Minnesota Department of Health laboratory which uses cadmium reduction with a detection limit of 0.01 mg/l. The individual stations' water quality information is maintained in the U.S. Environmental Protection Agency's (EPA) computerized water quality data base (STORET). The MPCA ground water ambient monitoring program was re-evaluated during the 1989-91 biennium and significant changes were made in the monitoring network design. The new name for the program is the "Ground Water Monitoring and Assessment Program."

All of the major aquifers in Minnesota are represented by at least two ambient wells, with most major aquifers having at least 25 wells in the ambient data set (see Table B-4). Over 30 percent of the ambient wells were constructed before 1960. There is a wide range and fairly even distribution of well depths in ambient wells (Table B-6).

The NO_3-N concentrations from the MPCA-ambient data set are presented in Table B-8 and Figure B-3.

Table B-8 Nitrate-N in MPCA-ambient wells meeting selection criteria.

Number Of Wells	Total # Analyses	Mean NO_3-N mg/l	Median NO_3-N mg/l	% Wells With NO_3-N Conc. (mg/l)			
				0-1	1.01-5	5.01-10	> 10
484	990	2.2	0.02	76	12	5	7

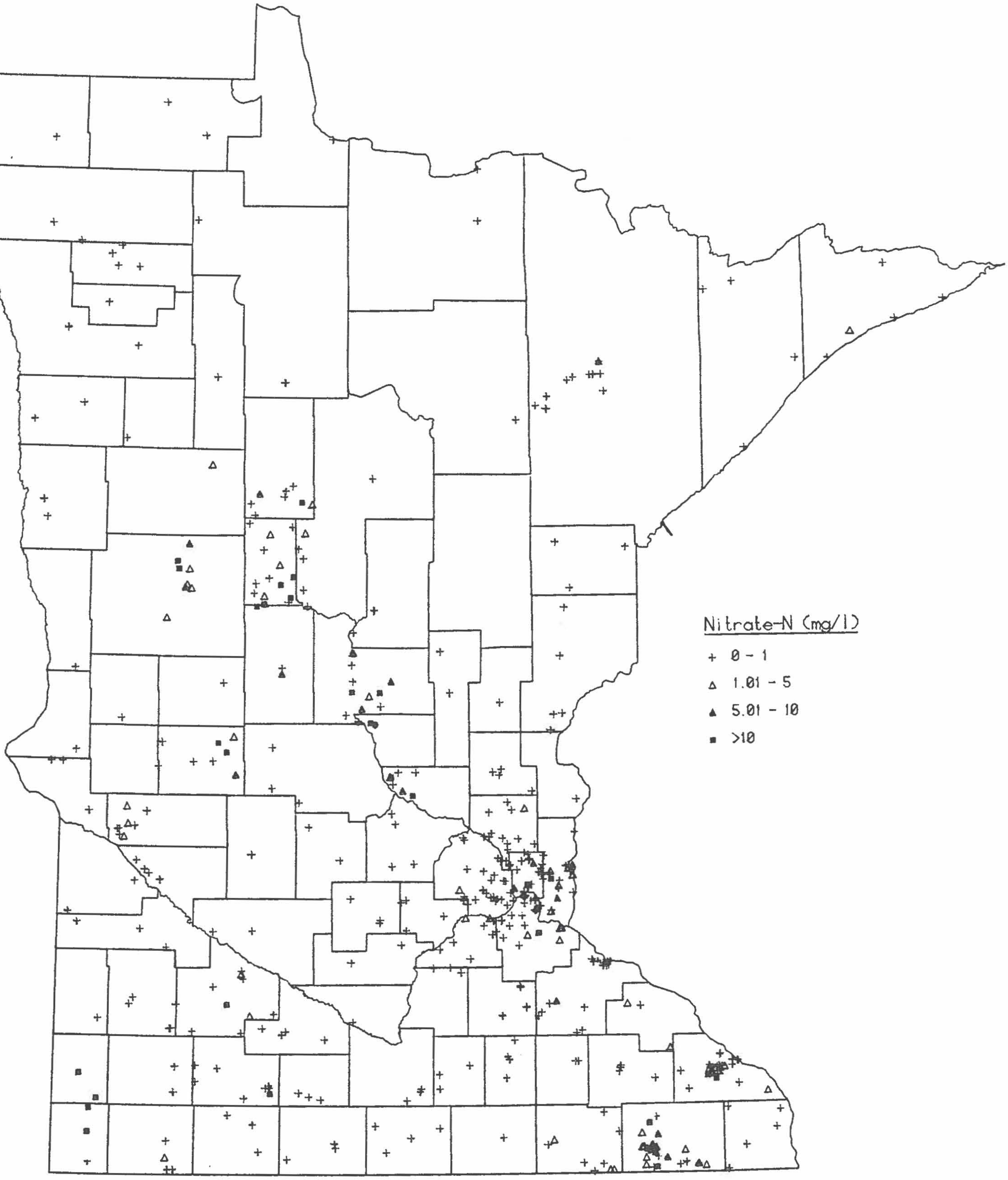
From this data set, the percent of wells with NO_3-N exceeding 1, 10, 20 and 30 mg/l were 24, 7, 2 and 1 percent, respectively. More than half of all wells had NO_3-N at or below 0.02 mg/l.

Figure B-3

MPCA Ambient Groundwater Monitoring

Well locations and Nitrate levels

(1978 - 1990)



County Well Index

County Well Index (CWI) is a computerized data base of well location and construction with associated water quality information for some wells. The system, developed by the Minnesota Geological Survey, currently (1991) has over 130,000 well records that have been submitted to the state by well drillers. A well record usually includes information about well depth and construction, but does not contain water quality information. While most of the records are from new well construction, there are also numerous records from well repairs that have been entered into CWI.

The water quality component of the system was added in 1987. Numerous NO_3 analysis results were recorded digitally between 1984 and 1987 and were then input into CWI in 1987. A majority of CWI NO_3 data is from samples required by the Minnesota Department of Health (MDH) to be taken following the completion of a new well. There have been three major efforts to input water quality results from new well construction into the CWI system. Each of these three efforts are described below.

1. During county geologic atlas preparation by the Minnesota Geological Survey, well records and water quality data have been extracted from MDH paper files and input into CWI. Completed county geologic atlases include Olmsted, Winona, Scott, Dakota, Hennepin and Washington counties. Additional NO_3 and bacteria data (other than new well construction) was added from county records to the CWI files for Olmsted, Scott, and Winona counties. Results from 177 Olmsted County wells from the SE Minnesota regional laboratory are duplicated in CWI.

2. As part of a multi-agency study on ground water sensitivity funded through the Legislative Commission on Minnesota Resources for the 1989-91 biennium, MGS has mapped out the extent of the Prairie du Chien-Jordan aquifer and analyzed existing NO_3 data for this aquifer. As part of the NO_3 analysis effort, MGS added 1,000 new well construction NO_3 analyses from MDH paper files into CWI for the 16 counties overlying the Prairie du Chien-Jordan aquifer. The unique well number on the MDH well record file was matched with existing well construction entries on CWI.

3. The MPCA, through this study, sought to include additional new well construction data from counties where: 1) geologic atlases were not developed, and/or 2) the Prairie du Chien-Jordan aquifer does not exist. Therefore, for these other counties, 3,500 NO_3 records were computerized from the MDH paper files of new wells constructed between 1980-1985. Nearly 3,000 of these wells were matched with existing CWI well location and construction files. Many other new well construction NO_3 analyses are on file at MDH.

A total of 9291 analyses from 8085 wells met the data selection criteria from CWI. Since a majority (70 to 90 percent) of the wells in CWI were constructed after the well code went into effect in 1975, these data are likely to be biased towards lower NO_3 as compared to a random sample of wells. Most experienced well drillers have an idea of the depth and aquifer needed in order to be reasonably sure of attaining acceptable water quality. Some drillers check the NO_3 levels with quick test strips before completing the well. For these reasons, this data set is likely to be further biased towards wells with low NO_3.

While most of the data in CWI was generated by EPA approved laboratory methods, it is likely that some of the data in CWI was produced by less reliable methods.

CWI is **not** biased towards shallow wells, and about three fourths of the CWI wells utilized in this study have a well depth between 50 and 300 feet (table B-6). While many aquifers are represented in the CWI data, the data are largely from glacial drift, Prairie du Chien, and Jordan aquifer wells. Over half of the samples were taken from 1985 to 1987. CWI NO_3 data represents counties throughout the state, but nearly 80 percent of the data is from counties in the Twin Cities area and Olmsted County. The NO_3-N data from CWI are presented in Table B-9 and Figures B-4(a), B-4(b), B-4(c) and B-4(d).

Table B-9 Nitrate-N in CWI wells meeting selection criteria.

Number Of Wells	Total # Analyses	Mean NO_3-N mg/l	Median NO_3-N mg/l	Wells With NO_3-N Conc. (mg/l)			
				0-1	1.01-5	5.01-10	> 10
8085	9,291	2.1	0.5	67	21	8	4

From this data set, the percent of wells with NO_3-N in excess of 1, 10, 20 and 30 mg/l were 33, 4, 1 and 0.3 percent, respectively.

CWI wells on the northwestern side of the Twin Cities (Hennepin and Anoka counties) appear to have very low NO_3. However, many wells with elevated NO_3 concentrations are found on the southeastern side of the Twin Cities (Dakota and Washington counties). Elevated NO_3 CWI wells are also found scattered throughout central and southeastern Minnesota. Very few CWI wells with NO_3 data exist in western and northern Minnesota.

Figure B-4(a)

County Well Index

Well locations and Nitrate levels

(1978 - 1989)

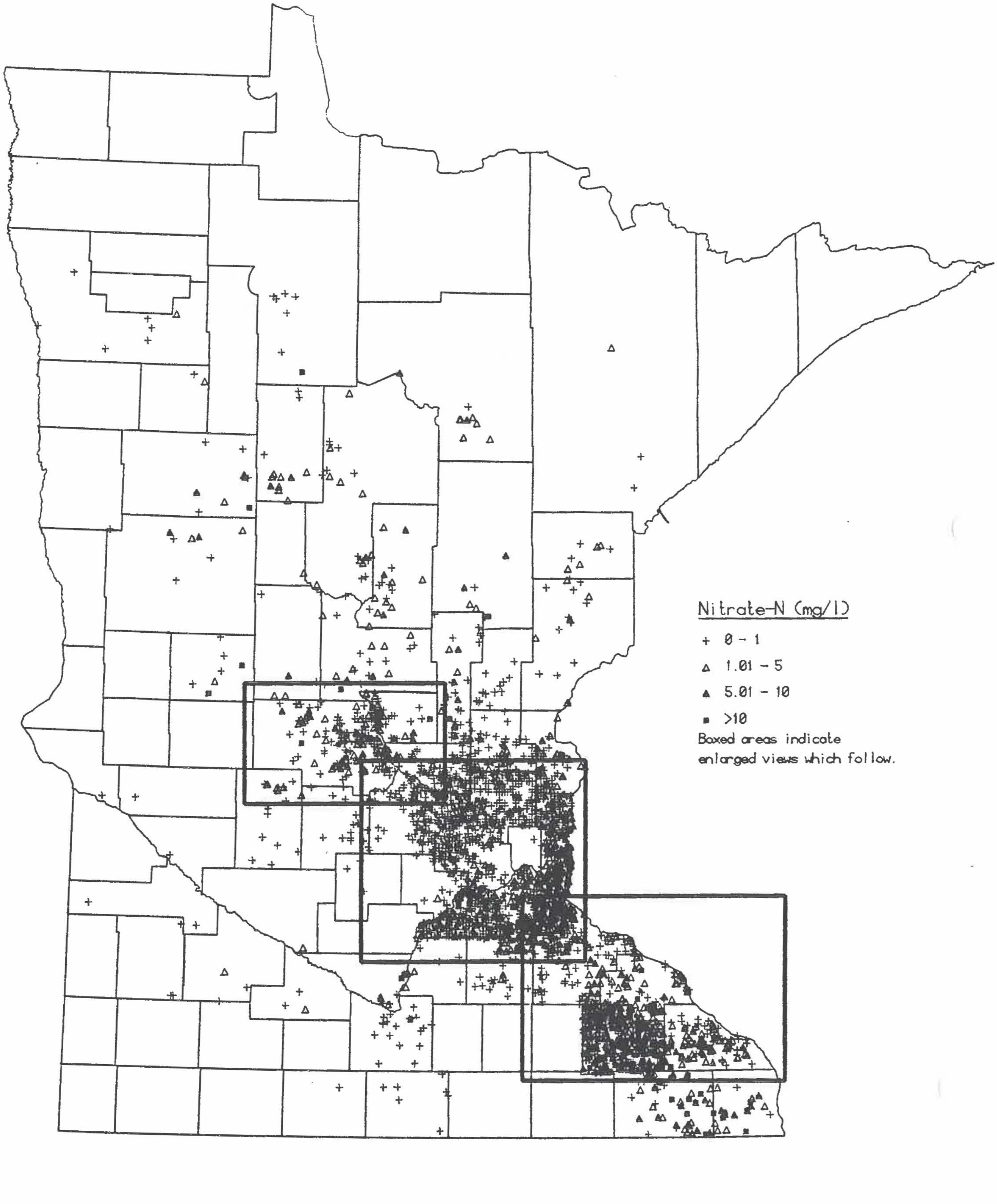


Figure B-4(b)

County Well Index

Well locations and Nitrate levels

(enlarged map of the Metro Area)

(1978 - 1989)

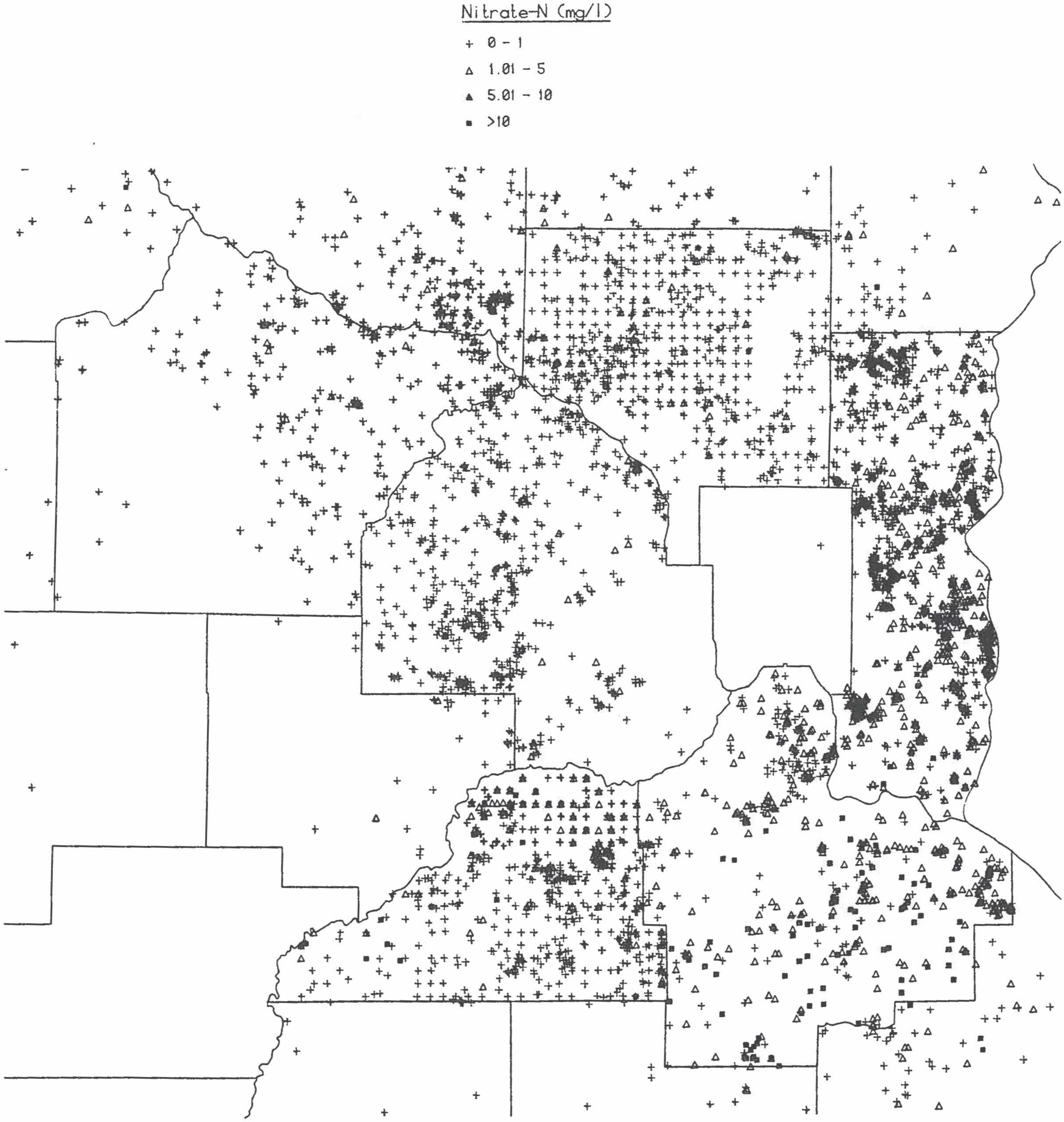


Figure B-4(c) and B-4(d)

County Well Index

Well locations and Nitrate levels

(enlarged map of the Stearns/Benton and Winona/Goodhue areas)

(1978 - 1989)

Nitrate-N (mg/l)

- + 0 - 1
- △ 1.01 - 5
- ▲ 5.01 - 10
- ■ >10

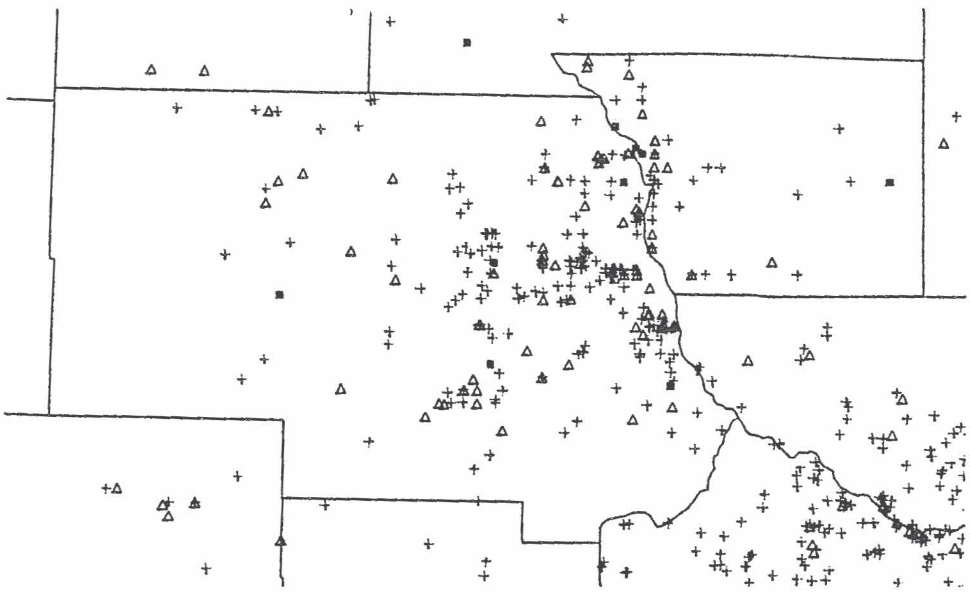


Figure B-4(c)

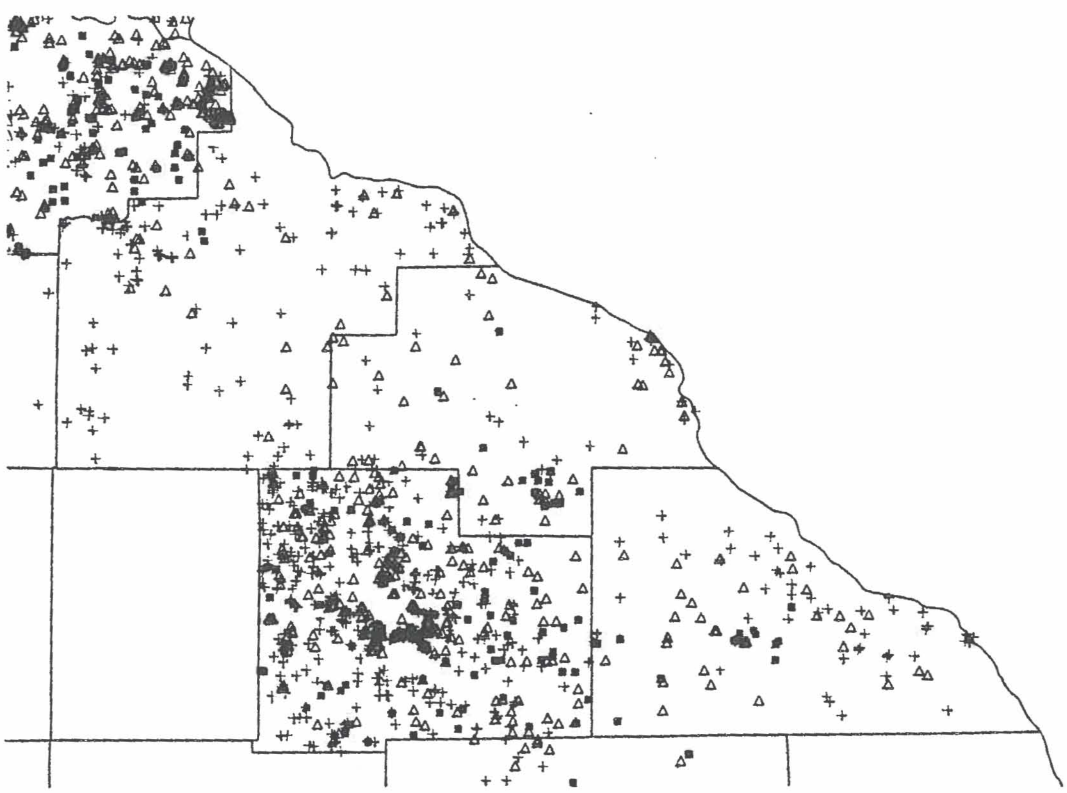


Figure B-4(d)

U.S. Forest Service

The U.S. Forest Service (NFS) stores its water quality data on the national data base STORET. This study utilized 502 NFS NO_3 analyses from 114 wells. Twenty-eight wells were eliminated from the data set (used for this study) due to their association with monitoring wastewater spray irrigation sites or water treatment plants. Selected wells are mainly potable water supply wells at U.S. Forest Service area administrative sites, campgrounds, and picnic areas. Most of the analyses were conducted at a U.S. Forest Service laboratory, Region 9 laboratory in Winton, Minnesota.

Aquifer and well depth information for many of these wells was not provided in STORET. Several wells were analyzed for NH_4, organic nitrogen, bacteria and several other parameters. While not a large data set (114 wells), the U.S. Forest Service information is important because it provides data for north central and northeastern Minnesota, where NO_3 data are more scarce. All NFS data are from six counties in north central and northeast Minnesota. Well age and depth information were unavailable for this data set.

Nitrate data from NFS are presented in Table B-10 and Figure B-5. Of the 114 wells sampled by the NFS, no wells had NO_3 exceeding 10 mg/l and only four percent of all wells had average NO_3-N in excess of 1 mg/l.

Table B-10 Nitrate-N in U.S. Forest Service wells meeting selection criteria.

Number Of Wells	Total # Analyses	Mean NO_3-N mg/l	Median NO_3-N mg/l	% Wells With NO_3-N Conc. (mg/l)			
				0-1	1.01-5	5.01-10	> 10
114	502	0.29	0.1	96	3	1	0

Minnesota Pollution Control Agency Nonpoint Source Ground Water Studies

The Minnesota Pollution Control Agency has conducted four ground water monitoring projects between 1987 and 1989 to help local governmental units define nonpoint source (NPS) impacts on ground water and factors affecting their ground water quality (see reference section). All sampling was conducted in Big Stone, Benton, Stearns and Winona counties. Aquifers with elevated NO_3 were known to exist previous to MPCA sampling in Benton and Winona county study areas. Therefore, these data will be biased towards higher NO_3 concentrations compared to a random sampling in those two counties. A majority of the wells were surficial sand aquifer wells, with half of all wells less than 50 feet deep.

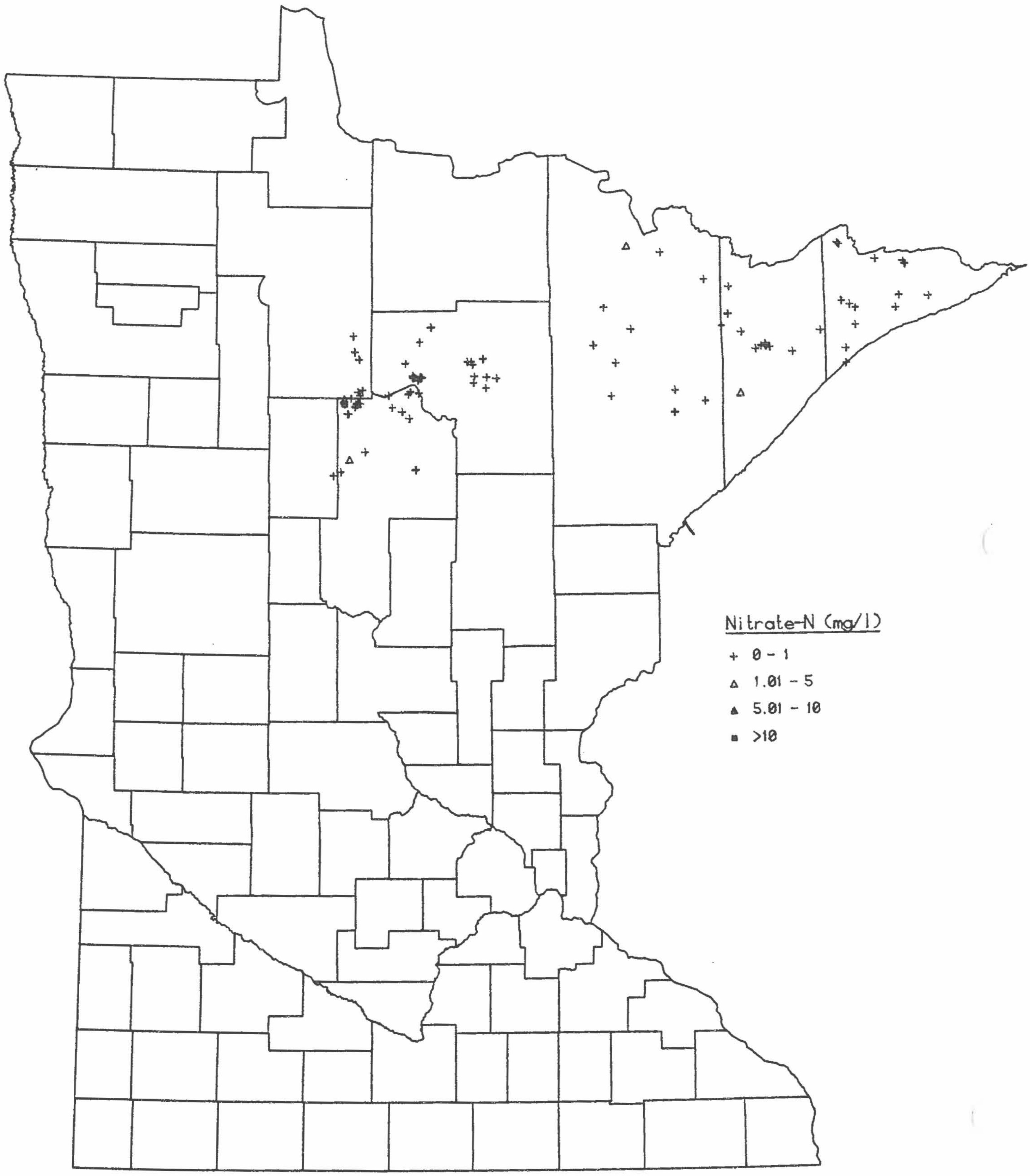
A total of 71 wells (384 analyses) were included from the four counties. Most of the wells were sampled two to seven times for NO_3 and once for NO_2, NH_4, total Kjeldahl nitrogen, and organic nitrogen. The type of wells sampled include primarily monitoring and domestic wells. Laboratory analyses were performed at the Minnesota Department of Health and the University of Minnesota Research Analytical Laboratory. All data are stored in the national EPA data base STORET.

Figure B-5

U.S. Forest Service

Well locations and Nitrate levels

(1978 - 1990)



The NO_3 data from NPS are presented in Table B-11. All four areas studied had at least a few wells above 10 mg/l. Each area also had at least a few wells with less than 1 mg/l NO_3-N. Thirteen wells had mean NO_3-N above 20 mg/l.

Table B-11 Nitrate-N in NPS wells (mg/l).

Number Of Wells	Total # Analyses	Mean NO_3-N mg/l	Median NO_3-N mg/l	% Wells With NO_3-N Conc. (mg/l) 0-1	1.01-5	5.01-10	> 10
71	384	10.2	6.8	21	17	18	44

Hundreds of wells are currently being sampled for NO_3 and other NPS contaminants through the MPCA administered Clean Water Partnership Program. These data were not yet readily accessible for use in this study.

Southeastern Minnesota Regional Laboratory

Olmsted, Dodge, Houston, Goodhue, Fillmore and Wabasha counties formed a regional well water testing program in 1983 to provide water quality data testing services to area residents. The Olmsted County Health Department offered to expand its county water laboratory to provide the regional laboratory testing services of a few selected parameters to the other counties.

The regional laboratory at the Olmsted County Health Department has been conducting NO_3 analyses for the six southeastern Minnesota counties since 1983 and Rice County since 1988.

The purpose of conducting NO_3 and bacteria analyses in the regional lab is primarily to provide a service for southeastern Minnesota private residents and community water supplies. The basis for having a regional lab rather than county labs was to minimize laboratory equipment and personnel costs associated with running numerous labs, provide for more consistency in analyses, collect regional information about the well location and construction, and make region-wide ground water quality data available to local and regional decision makers.

The individual counties are responsible for informing residents and businesses of the service being provided by the regional laboratory. Participation levels vary from county to county. Sample bottles and forms were provided to interested well owners. Information on the county, owner's name and address, township name, section number, well depth, year drilled, distance between the well and various pollution sources, date of sample collection, and reason for taking the sample are noted. The sample is then mailed or hand delivered to the laboratory in Rochester where the analyses are then conducted within 24 hours. Laboratory methods for NO_3 analysis were cadmium-reduction for all data through 1987 and HPLC (high pressure liquid chromatography) for most samples submitted between 1988 and 1990. Detection limits have varied, generally ranging between 0.1 and 0.4 mg/l.

The regional laboratory has been computerizing and managing the data and currently enters all data on data base software. Township and range numbers were determined by Olmsted County staff from the township name. Time and money did not allow field verification of location data. A unique well number was not assigned to each analysis. Therefore, multiple analysis wells were identified in this study by matching the township, range and section information, the first three letters of the owners last name, and the well depth.

This data set does not represent a random sampling and there are likely to be biases in the data due to voluntary submission of samples and use of private wells.

A total of 4728 wells met the selection criteria from seven counties. Most analyses were conducted between 1985 and 1990. Many wells from this data set were constructed before 1960 (see Table 5). Nitrate results from SE MN are presented in Table B-12 and Figure B-6.

Table B-12 Nitrate-N from Southeastern Minnesota Regional Laboratory.

Number Of Wells		Total # Analyses	Mean NO_3-N mg/l	Median NO_3-N mg/l	% Wells With NO_3-N Conc. (mg/l) 0-1	1.01-5	5.01-10	> 10
Dodge	542	638	1.6	<1.0	79	9	6.5	5.5
Fillmore	826	1005	6.5	4.4	30	23	23	24
Goodhue	1004	1189	4.2	2.0	42	28	17	13
Houston	131	154	5.5	3.1	40	23	22	15
Olmsted	1062	1355	2.2	<1.0	67	17	10	6
Rice	490	567	1.9	<1.0	77	9	8	6
Wabasha	673	819	5.2	2.7	33	32	17	18
All	4728	5727	3.8	0.8	52	21	14.5	12.5

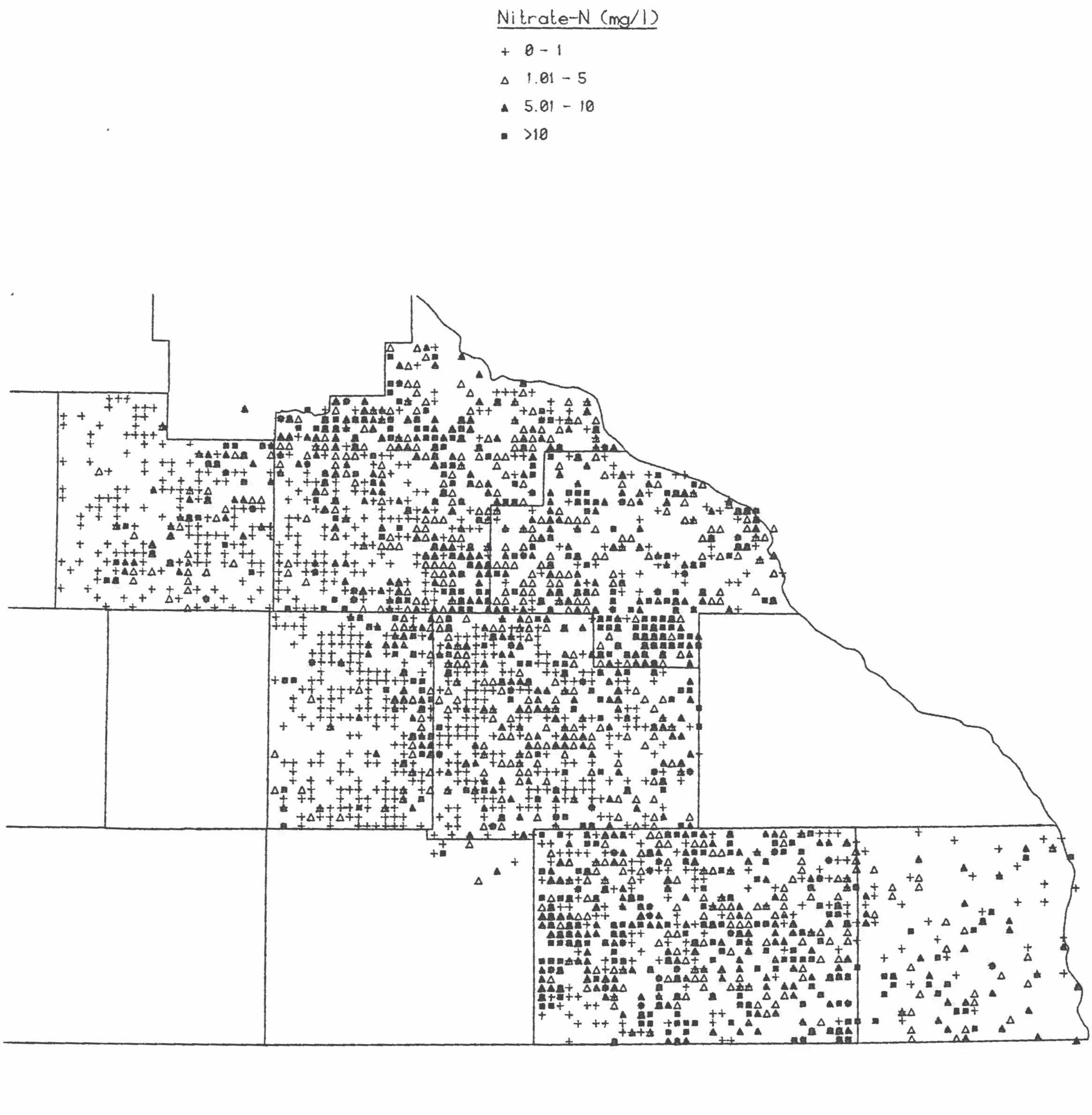
From this data set, the percentage of wells exceeding 1, 10, 20 and 30 mg/l were 48, 12.5, 3.0 and 0.7 percent, respectively. Major differences were noted in results among the seven counties. Dodge, Rice and Olmsted counties each had about six percent of wells exceeding 10 mg/l. Wabasha and Fillmore counties had 18 and 25 percent of wells exceeding 10 mg/l. With the exception of areas in Rice County and Dodge County which appear to have few wells with elevated NO_3, most townships in the southeastern Minnesota counties appear to have a wide range of NO_3 levels among area wells. Certain areas of Goodhue, Wabasha, and Fillmore county stand out as having many high NO_3 wells.

Figure B-6

Southeastern Minnesota Regional Lab

Well locations and Nitrate levels

(1981 - 1990)



Minnesota Department of Agriculture Pesticide/Nutrient Sampling Program

The Minnesota Department of Agriculture (MDA) maintains a ground water monitoring program designed to study the long-term effects of normal agricultural pesticide use on ground water quality. The MDA's ground water quality monitoring networks are carefully designed based upon statistical data analysis requirements. Network wells are selected based, in part, upon pesticide use and land management practices on adjacent lands. The monitoring program networks are not designed, in a strict statistical sense, to study the effects of fertilizer use on ground water quality.

For the Minnesota diagnostic ground water monitoring network, the state is divided into 24 regions, or county clusters, consisting of two to six counties each. The boundaries of these regions were determined based on similarity in hydrogeology, soils, cropping patterns, and other land uses. To date, the network has been developed and maintained in the county clusters in central, southeastern, and southwestern Minnesota. Wells are selected within these county clusters based upon several criteria; one of the primary requirements is that a well must be down gradient of agricultural fields that have received pesticide applications within the previous five growing seasons. Information on fertilizer use in the area upgradient of the well is not required. The number of wells sampled in each county cluster varies depending on hydrogeologic, soil, and pesticide use characteristics. The network provides baseline information on how current pesticide use practices affect ground water quality; this information is used to guide policy decisions on pesticide use management. The data should not be used to assess average water quality conditions for a given aquifer, region, or the state.

About eighty percent of the network wells are Quaternary Water Table Aquifer (QWTA) wells less than 50 feet deep. Quaternary water table aquifers are composed of unconsolidated sand and gravel deposits left by the melting of the most recent glaciers, and have no confining layer between the water table and the ground surface. Water table depths may range from less than ten feet to greater than 40 feet. All network QWTA wells are monitoring wells. Domestic drinking water wells are often used in the southeastern clusters where the karst bedrock aquifers are monitored.

When samples are collected for pesticide analysis from MDA network wells, ground water samples are also collected and submitted for NO_3-N analysis. Nitrate analysis is performed by the MDA Laboratory Services Division of the MDA using cadmium reduction method with a reporting limit of 1 ppm. The NO_3 data generated from the period 1988 to 1990 are presented in Table B-13 and Figure B-7. Sampling frequency varied for different wells. Wells with high NO_3 concentrations (> 10 ppm) were distributed throughout many of the county clusters where sampling occurred. **It is important for the reader to recognize that these data were not generated by a monitoring network statistically designed to address the effects of nitrogen fertilizer use on ground water quality.** Further information on the MDA water quality monitoring program can be found in Hines et al., (1990).

Table B-13 Nitrate-N from MDA wells.

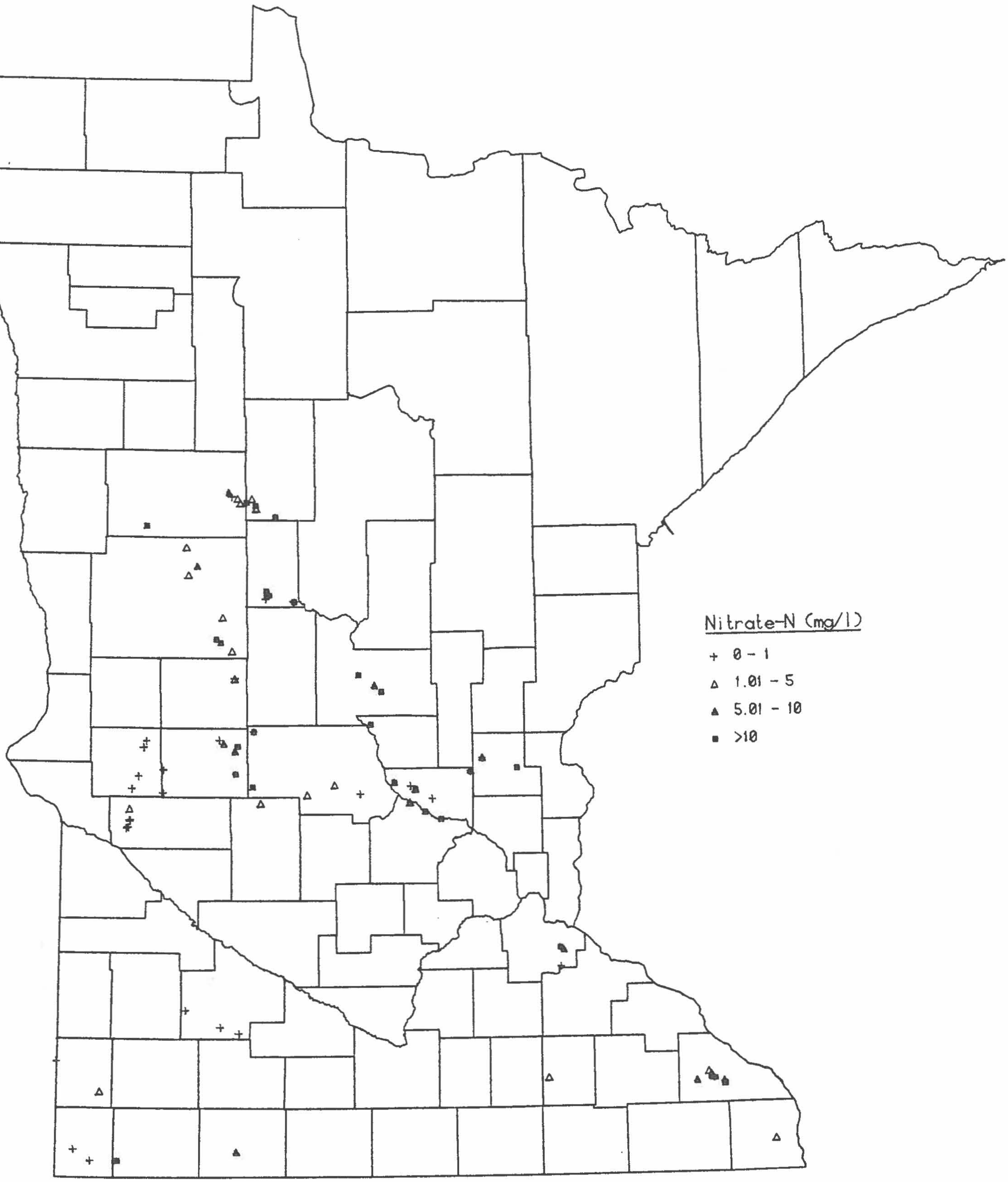
Number Of Wells	Mean Total # Analyses	Median NO_3-N mg/l	% Wells NO_3-N mg/l	% Wells With NO_3-N Conc. (mg/l) 0-1	1.01-5	5.01-10	> 10
95	413	7.4	4.0	34	21	15	30

Figure B-7

Minnesota Department of Agriculture

Well locations and Nitrate levels

(1987-1990)



Miscellaneous Ground Water Nitrate Data Sets

Nitrate results from nine other major data sets that were not described in the previous section are summarized below. The actual NO_3 data were not obtained for all but one of these data sets. The results included within this section are from existing reports and personal communication with representatives of groups collecting the data. Many of the data sets described in this section did not have readily accessible well location information. Data from two data sets were produced by methods not approved by EPA. Wells sampled for one data set were located within about one mile from used and unused landfills.

Community Public Water Supply Systems (MDH)

MDH obtains NO_3 data for all public water supply wells (e.g. cities, restaurants, schools, gas stations, etc.). MDH[1] provided nitrate result information for 1678 community public water supply wells. By definition public water supply wells provide water for human consumption to at least 15 service connections used by year round residents, or regularly serves at least 25 year round residents (e.g. municipality, subdivision, mobile home park). The percentages of wells having < 1, 1-5, 5.01-10 and > 10 mg/l NO_3-N were 78.6, 17.2, 3.0, and 1.2, respectively. These data were mostly from one-time samples analyzed between 1985 and 1988.

Community wells generally have lower NO_3 than private wells primarily because they are often deeper and are better constructed, maintained, and monitored than many existing private wells. Community wells are more likely to be relocated if NO_3 problems are found than are domestic wells.

MDH records of approximately 4000 non-community public water supply systems show 65 (1.6%) with NO_3-N greater than 10 mg/l.[1]

Newly Constructed Private Wells (MDH)

MDH also has been collecting paper files of new well logs and associated NO_3 and bacteria levels as reported by the well driller. Eight to ten thousand NO_3 analyses from newly constructed wells are received by MDH each year. MDH compiled the results from 6,899 new private well construction NO_3 analyses from samples submitted by the well driller or pump installer to the Minnesota Department of Health laboratory. Analyses were from the periods June 1988 to December 1989 and October 1990 to June 1991. A majority of the wells were drilled within 100 miles of the Twin Cities. Table B-14 shows the percentage of analyses that were found within various NO_3-N categories.

Table B-14 Nitrate-N concentrations analyzed at MDH laboratory from 6899 recently constructed private wells (personal communication with Steve Ring, MDH).

Nitrate-N mg/l	Percentage of Wells
< 0.4	87.1
0.4 - 5	9.4
5.01 - 10	2.1
> 10	1.4

---

[1]Personal communication with Phil Moroukian.

Metropolitan Landfill Study (MDH)

Metropolitan landfill monitoring by MDH produced a valuable computerized data set for the Twin Cities area. Klaseus (1991) reported that 1,302 wells had been sampled (235 public wells and 1,067 private wells) within about one-mile from 155 dump sites in the seven-county metropolitan area from 1985 through 1990. Most of the dump sites or landfills were inactive. Laboratory analyses were performed at the MDH lab. Table B-15 shows the number of wells in various categories of NO_3 concentration from the MDH dump site monitoring.

Table B-15 Nitrate in wells surveyed for the MDH metropolitan landfill study.

Nitrate-N mg/l	# Wells	% of Total
< .4	940	72.2
0.4 to 10	265	20.4
> 10	97	7.4

MDA Pesticide Survey Nitrate Results

The Minnesota Department of Agriculture sampled 65 observation wells, 31 drinking water wells, and four irrigation wells during 1986 and 1987 in unconfined surficial sand aquifers and in karst areas of Minnesota (Klaseus, et al., 1988). Most wells were in areas of intensive pesticide use and were sampled four times. The primary purpose of the sampling was to evaluate the possibility of pesticide movement to ground water in Minnesota in susceptible regions. Nitrate analyses were also performed on the water samples. The NO_3 results are presented in Table B-16. Some of these wells are duplicative of MDA pesticide/nutrient monitoring wells previous described.

Table B-16 Nitrate-N from the 1986-87 MDA pesticide survey.

Region	# Wells	# Wells NO_3-N < 1 mg/1N	# Wells NO_3-N 1-10 mg/l	# Wells NO_3-N > 10 mg/l
Northwestern MN	8	8	0	0
SW and SC MN	17	10	6	1
Southeastern MN	21	2	12	7
Central MN	54	16	19	19
Total	100	36	37	27

MDH Pesticide Survey Nitrate Results

Public Wells

The Minnesota Department of Health sampled 224 municipal wells and 176 other public water supply wells (offices, schools, churches, restaurants, etc.) between May 1986 and June 1987 (Klaseus et al. 1988). Wells were selected mostly based on their apparent susceptibility to pesticide contamination. Nitrate analyses were conducted on 395 of these wells. Just over 30 percent of the wells were sampled twice and other wells were sampled once. Nitrate-N was detected above 0.4 mg/l in 187 wells (47.3 percent), and exceeded 10 mg/l in 28 wells (7.1 percent). The locations and general NO_3-N concentration of the MDH sampled public water supply wells is shown in Figure B-8.

Private Wells

In a separate study, the Minnesota Department of Health conducted a survey of 225 private wells for pesticides between April 1986 and May 1987 (Klaseus and Hines, 1989). Nearly all (224) wells were also sampled for NO_3. Twenty-five wells were sampled eight times each. A majority of the wells were located in geologically-sensitive agricultural regions. However, some wells were also sampled in less sensitive areas. Results from the one or two-time sampling of 199 wells showed 71.4 percent of all wells having NO_3-N above the detection limit of 0.4 mg/l, with 42.2 percent of all wells having NO_3-N above 10 mg/l.

Figure B-9 shows the general sampling site locations and the NO_3-N detection status of each well. The median NO_3-N concentration exceeded 10 mg/l in seven out of the ten areas surveyed. Two areas stood out as having very low NO_3, northwestern Minnesota and Martin County.

Of the 25 multiple analysis wells, 18 had NO_3-N exceeding 10 mg/l in at least one out of the eight samples, four wells had 0.4 to 10 mg/l NO_3-N and three wells had less than 0.4 mg/l. The NO_3 levels were fairly consistent over the course of the study in 21 of the 25 wells.

County Sampling Programs

Community Health Services Reports

Community Health Services (CHS) and/or local water planners for most counties in the state provide or coordinate NO_3 testing as a service to well owners. Counties have a number of ways of advertising the service, handling the samples, analyzing the water for NO_3 and tracking the data. Some counties submit samples to certified laboratories and other counties perform NO_3 analyses with their own equipment such as a Hach Colorimetric kit. **The reliability of data from CHS's varies greatly among counties and should be used with much discretion.**

Most CHS's submit their results to the Minnesota Department of Health on an annual basis. Sixty-seven Community Health Services submitted at least 20 NO_3 results to MDH during 1988 and 1989. These results are shown in Table B-17.

Figure B-8
OCCURRENCE OF NITRATES
PUBLIC WELLS, MDH SURVEY
(From Klaseus et al., 1988)

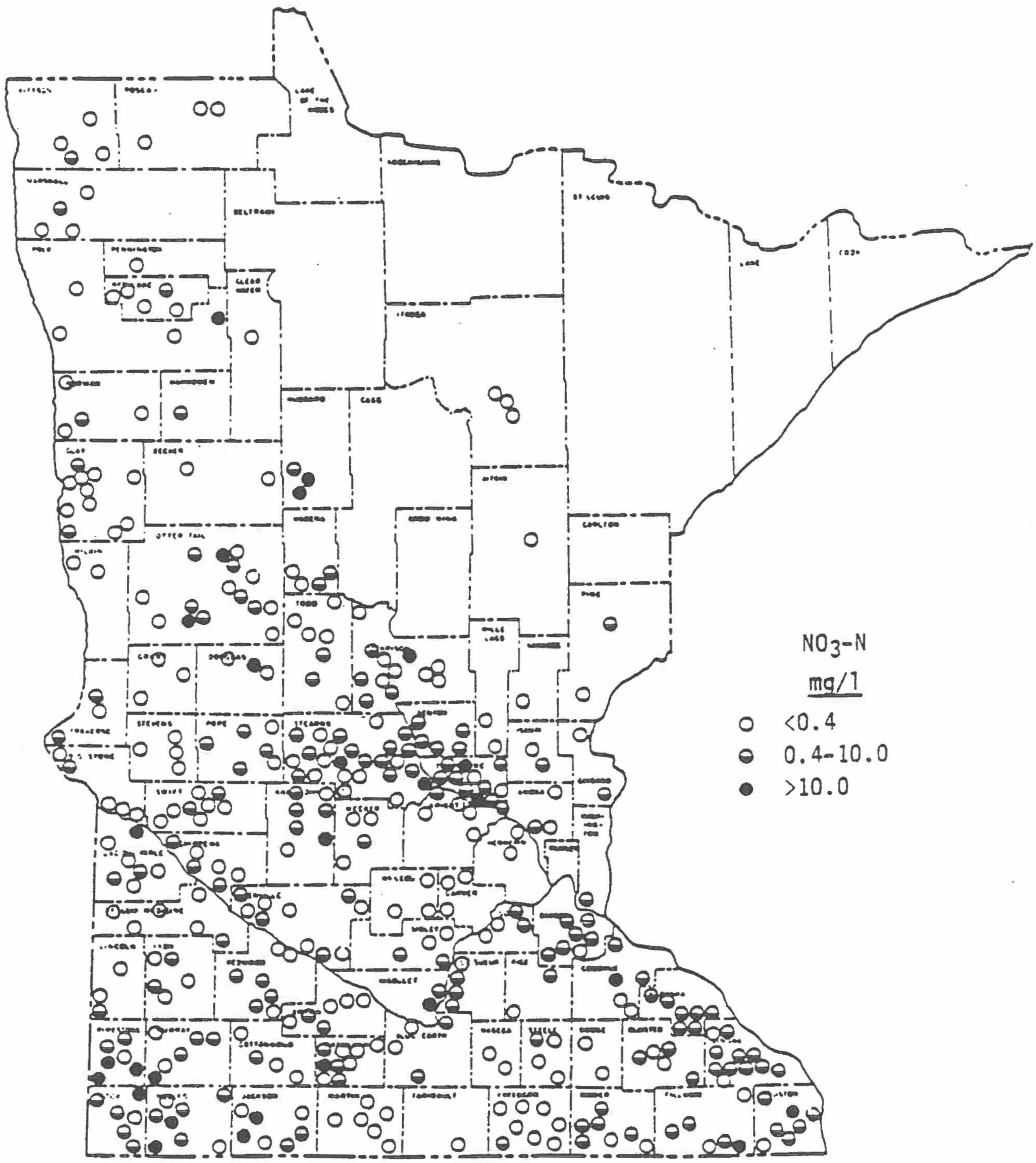


Figure B-9
OCCURRENCE OF NITRATES (NO_3-N)
ONE-TIME SURVEY
PRIVATE DRINKING WATER WELLS
(From Klaseus & Hines, 1989)

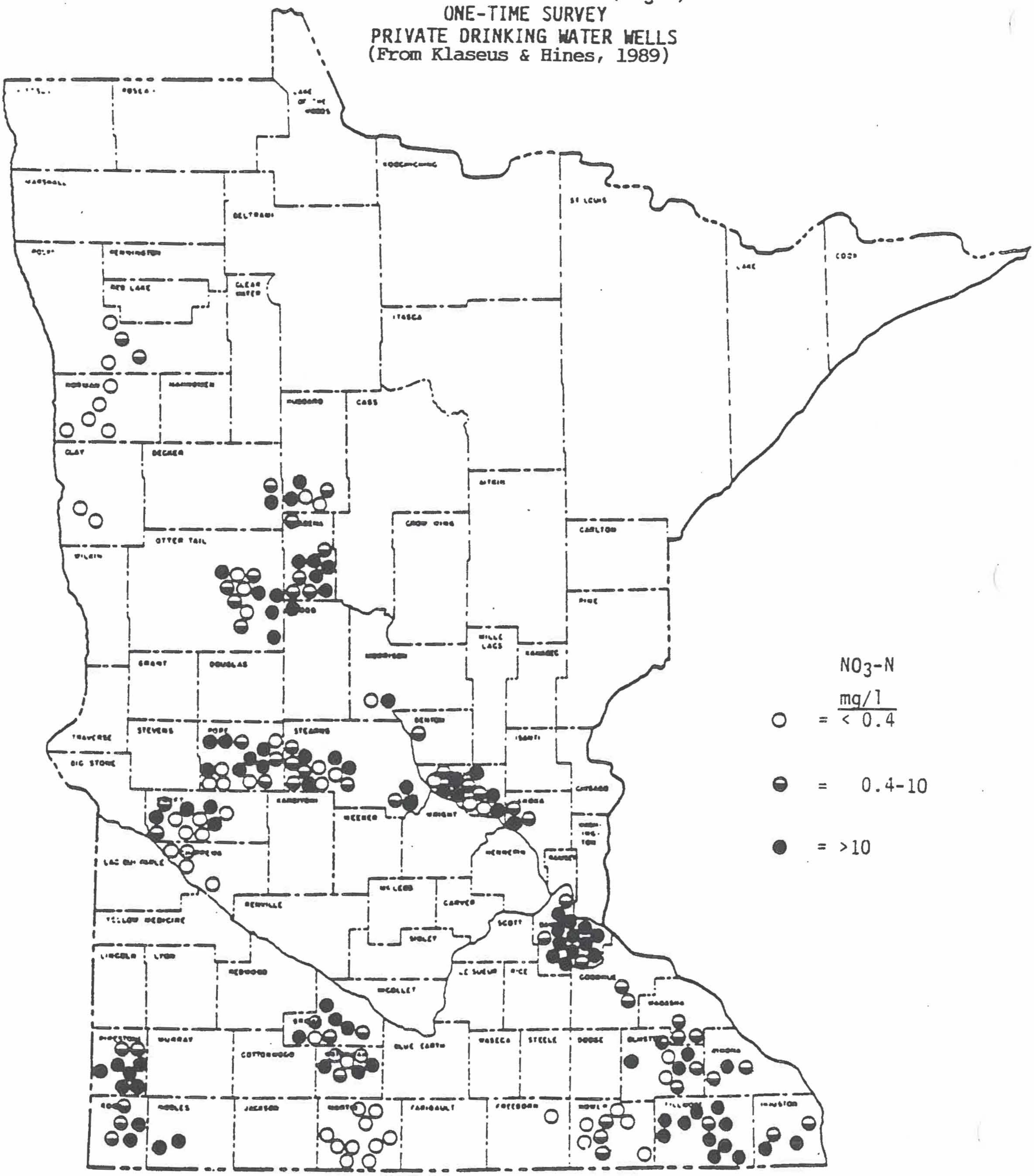


Table B-17 Community Health Services reports to MDH for years 1988 and 1989. The number of wells is unknown. Use this information with a great deal of discretion.

COUNTY	1988 & 1989 NO3-N Total # Samples Taken	1988 & 1989 percent NO3-N >10 mg/l
NORTHWEST		
BELTRAMI	317	19.2
HUBBARD	74	13.5
KITTSON	28	14.3
LAKE OF WOODS	52	11.5
MARSHALL	59	6.8
PENNINGTON	69	4.3
ROSEAU	179	1.7
NORTHEAST		
AITKIN	383	2.9
COOK	157	0.0
ITASCA	707	0.3
LAKE	127	4.7
ST. LOUIS	2612	0.5
WEST CENTRAL		
CLAY	894	1.6
OTTER TAIL	1288	3.0
CENTRAL		
BENTON	168	4.8
CASS	642	1.2
CROW WING	263	1.5
ISANTI	25	12.0
MILLE LACS	148	4.7
MORRISON	563	8.9
SHERBURNE	446	7.0
STEARNS	923	3.9
TODD	150	4.7
WADENA	110	6.4
WRIGHT	337	1.2
SOUTHWEST		
BIG STONE	90	6.7
CHIPPEWA	211	5.2
KANDIYOHI	40	2.5
LAC QUI PARLE	244	3.3
LINCOLN	34	23.5
LYON	266	27.1
MURRAY	118	33.1
NOBLES	219	22.4
PIPESTONE	92	20.7
REDWOOD	207	12.1
RENVILLE	88	6.8
ROCK	22	18.2
SWIFT	305	3.0
YELLOW MEDICINE	186	8.6
SOUTH CENTRAL		
BLUE EARTH	305	2.3
BROWN	1018	7.9
FARIBAULT	51	3.9
LE SUEUR	396	0.0
MARTIN	107	4.7
MC LEOD	165	0.6
MEEKER	329	0.9
NICOLLET	1855	5.0
SIBLEY	123	9.8
WATONWAN	49	12.2
WASECA	254	0.4
SOUTHEAST		
DODGE	330	4.2
FREEBORN	389	1.5
GOODHUE	275	24.0
HOUSTON	125	11.2
MOWER	982	1.3
OLMSTED	1450	6.1
RICE	473	5.9
STEELE	329	2.7
WABASHA	102	22.5
WINONA	361	5.5
METRO		
ANOKA	873	2.1
CARVER	154	0.6
DAKOTA	365	15.1
HENNEPIN	378	0.8
RAMSEY	814	1.2
SCOTT	593	0.8
WASHINGTON	818	2.4
Total	26306	4.65

CHS's submit information on the total number of analyses rather than the total number of wells. Since wells tested with high NO_3 are more likely to retest, the data are likely skewed to higher NO_3 levels.

A total of 26,306 NO_3 results were reported by CHS's for years 1988 and 1989. Fifteen counties each submitted over 500 NO_3 results for the two years. Results reported from certain counties are also included within data sets discussed in two other sections of this chapter, Southeastern Minnesota Regional Laboratory and south central counties - individual sampling programs. Counties with partial duplicative results include Dodge, Goodhue, Houston, Olmsted, Rice, Steele, Wabasha, Mower, Blue Earth, Brown, Nicollet, Sibley and Steele (7265 analyses). About 4.7 percent of all CHS analyses had reported NO_3 concentrations above 10 mg/l. The reported percent of wells exceeding 10 mg/l NO_3-N varies greatly among counties. A number of counties in northeast Minnesota, south central Minnesota, and in the metro area had less than two percent of analyses exceeding 10 mg/l. Other counties, primarily in southwestern and southeastern Minnesota, had over 20 percent of analyses exceeding 10 mg/l.

South Central Minnesota - Individual County Sampling Programs

While most of the Community Health Services do not have their NO_3 data on computer, several counties are beginning to keep better track of water quality results. Data from seven southeastern MN counties was previously described. Through the process of developing and implementing Comprehensive Local Water Plans, several counties in south-central Minnesota have also had their NO_3 data sets computerized. Mankato State University computerized domestic well NO_3 data for Brown, Nicollet, Cottonwood, Blue Earth, Jackson, Steele and Sibley counties. The amount of associated well information entered onto the computer varied by county and well owner. Well owners filled out a form that usually asked for the well location by township, range, and section, well depth, date of construction, and distance from nearby pollution sources.

The computerized South Central Counties raw data were obtained and analyzed for this study. The sampling program from which each county's data were obtained is described for each county below. Since each county sampled water by the request of the owner rather than randomly choosing wells, the data are likely to be biased. Wells sampled more than once were identified by matching the location, owner's name, and well depth. Most of the samples were collected by the well owner and therefore some caution must be used when analyzing and reporting these results. Year of construction information was provided for about one-third of all wells. From the information gathered, there is a wide range and even distribution of dates of construction, with many wells constructed prior to 1945. Well depth information indicated that nearly 90 percent of all wells were between about 30 and 300 feet. The only data set produced by EPA approved methods was from Steele County.

Brown and Nicollet counties analyzed 3491 samples between June 1988 and April 1989. Nitrate analyses were offered to residents that brought water samples to their town hall on a specified date. At the town hall, samples were analyzed using a Hach kit and colorimeter. While this is not a certifiable method by

MDH, split samples sent to certified labs reportedly gave similar results. Most of the wells for which water was submitted were private domestic wells. Nitrate results from Brown, Nicollet and Cottonwood counties is summarized elsewhere (Holtz, 1990).

Since 1979, Steele County has offered NO_3 and bacteria analyses to well owners submitting a water sample. Due to lack of participation at the programs conception, the data are more representative of the last seven years. All samples were analyzed at Minnesota Valley Testing Laboratory in New Ulm, Minnesota using cadmium reduction techniques (EPA approved methods). It was estimated that 30 to 40 percent of the samples submitted were for real estate transactions and that nearly every sample submitted is from a domestic well.[1]

The Sibley County Public Health Office has offered NO_3 testing for non-municipal water supplies since 1979 and has supplied data from 124 wells where the location was known. Samples are brought to the public health office where the samples are analyzed immediately on-site by a consulting sanitarian. As with the other county sampling programs, the well owner is asked to provide well location and construction information.

Jackson County has tested water samples for NO_3 since 1980 using a Hach Colorimetric kit. Most samples are collected and submitted by residents wanting to know the NO_3 concentration of their domestic well.

Since 1972, Blue Earth County has offered to analyze private residence water for NO_3 using a Hach colorimetric kit with a meter. Nitrate results have been compared with those at laboratories using EPA approved methods and found to be similar. The samples are collected by county people in some wells and samples are brought into the county offices by well owners from other wells.

Residents of Cottonwood County may obtain a Whirlpack water sample bag and directions for sampling their wells from the county extension office. Once samples are collected, they are sent to Jackson County where the NO_3 analysis is performed using a Hach colorimetric kit.

The NO_3 results for each county and the combined data set are provided in Table B-18. Detections of NO_3 in domestic wells appear to be quite low in Steele County which has only six percent of all wells with NO_3-N greater than 1 mg/l. Other counties, including Cottonwood and Jackson had over twenty percent of wells exceeding 10 mg/l NO_3-N.

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[1] Personal communication with Scott Goldberg, Steele County Environmental Services Director.

Table B-18 Nitrate-N in certain South Central Minnesota Counties (mg/l).

	# Analyses Since 1978	# Analyses Since 1978 & TRS Known	# Wells Since 1978 & TRS Known	% Wells 0-1	% Wells 1.05-5	% Wells 5.01-10	% Wells > 10
S. Central Counties	6072	4085	3588	68	21	5	6
Blue Earth	515	154	134	82	8	6	4
Brown/Nicollet	3491	2814	2413	65	26	5	4
Cottonwood	222	136	129	61	11	7	21
Jackson	440	315	297	54	18	6	22
Sibley	240	137	124	50	33	6	11
Steele	649	529	491	94	2	1	3

Faribault, Martin and Watonwan Counties

Like many counties in the state, Faribault, Martin and Watonwan counties have been conducting water quality educational programs. Participants of two educational sessions during March 1988 were offered an opportunity to bring in water to test for NO_3, bacteria, sulfate, and seven pesticides. Nine percent of 336 samples had NO_3-N above 10 mg/l. Wells with concrete tile and clay tile casing tested positive for bacteria much more than wells with steel and plastic casing. Depth of well and well age did not show any strong correlation with NO_3. These results were described in a report "FMW-Water Project" by the Extension Service, Soil and Water Conservation Districts, and Soil Conservation Service in the three counties.

Rock and Nobles Study

A cooperative study, involving the Nobles-Rock Health Service, the Nobles and Rock County Extension Service, Nobles and Rock Soil and Water Conservation Districts, local township boards and affected watershed districts, was conducted to 1) inform homeowners of certain characteristics of their well water and 2) establish base-line ground water data applicable to comprehensive county water plans.

Between May 1990 and December 1990, 1,350 water samples from wells in Rock and Nobles counties were analyzed for NO_3. A cadmium reduction and a spectrophotometer was used for NO_3 analysis. The average NO_3-N concentration from all 1,350 samples was 9.6 mg/l.[2] Table B-19 lists the percentage of wells falling into various ranges of NO_3.

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[2]All results presented from Rock-Nobles counties are from personnel communication with Lee Carlson, Public Health Sanitarian (Rock-Nobles counties).

Table B-19 Rock-Nobles Counties Nitrate-N Sampling Results (1990)

	Rock County	Nobles County
# wells	356	994
nitrate-N		
% 0-1.5 mg/l	18%	15%
% 1.6-10 mg/l	46%	52%
% > 10 mg/l	36%	33%

About 34 percent of all wells sampled in the two counties had NO_3-N which exceeded 10 mg/l. Twelve percent of all wells had NO_3-N concentrations exceeding 20 mg/l. Augured or dug wells had higher average NO_3 concentrations than drilled wells for all wells over 60 feet deep.

Other Projects

Other NO_3 data are currently being collected for many Clean Water Partnership Projects, the Anoka Sand Plain Regional Ground Water Assessment, and the Minnesota River Assessment Project that were not readily available for use in this study.

There are several other data sets that exist from research projects or local studies. It was beyond the scope of this project to track down and include all existing data.

Discussion of Existing Conditions

Degree of Problem

Major differences in ground water NO_3 conditions are found when comparing results from the various data sets that were generated between 1978 and 1990 (Table B-20, Figures B-10 to B-13). Sampling programs that target private wells and/or monitoring wells in sensitive geologic areas under agricultural production had between 27 and 44 percent of the wells exceeding 10 mg/l NO_3-N (MPCA-NPS, MDA-pesticide survey wells, and MDH-private well pesticide survey). Data sets that have NO_3 data from municipal wells or primarily newer constructed wells throughout the state showed from 1 to 4 percent of wells exceeding 10 mg/l NO_3-N. The other data sets had generally 4 to 33 percent of wells with excess NO_3 (> 10 mg/l). MPCA ambient monitoring program results from 484 wells in different aquifers throughout the state showed NO_3-N exceeding 10 mg/l in 7 percent of sampled wells. The National Forest Service data set was an exception, with no wells above 10 mg/l and only 4 percent of wells with NO_3-N above 1 mg/l.

Some of the NO_3 variability between the various data sets is likely due to aquifer differences. However, even when comparing data from a similar aquifer type, such as surficial sand aquifers, there are great differences in NO_3 levels (Table B-21).

Table B-21 Nitrate-N results for surficial sand aquifers (SSA) wells sampled for various sampling programs

Data Set	#SSA Wells	Mean mg/l	Median mg/l	% > 10 mg/l
USGS	363	5.9	2.6	19
MPCA - Ambient	96	4.1	0.4	15.6
MPCA - NPS	51	11.0	8.3	45.1
MDA	79	6.9	3.4	26.6
CWI	379	3.9	3.4	4.5

It is difficult to get an accurate estimate of the percentage of wells exceeding 10 mg/l in the state since none of the sampling programs were designed to obtain such an estimate. There are approximately 410,000 permanent residence private water supply wells in Minnesota.[3] For this study, the number of wells with nitrate information obtained using EPA approved analysis methods is 26,340. About 7.3 percent of these wells had NO_3-N exceeding 10 mg/l. Despite the fact that we have reliable NO_3-N data from over five percent of permanent residence wells in the state, we still do not know the percent of all wells in the state exceeding 10 mg/l. This is due to the fact that we do not know how representative the sampled wells are of the ages, aquifers, and locations of other wells in the state. **It should not be inferred that 7% of the state's drinking water supplies exceed 10 mg/l NO_3-N. What can be inferred is that about 7% of wells included within the more "reliable" data sets described in this report exceed 10 mg/l NO_3-N.**

[3] Extrapolated from 1980 census information.

Table B-20 Nitrate data sets discussed in this report.

DATA SET (* Obtained and analyzed raw data for this study)	EPA APPROVED LABORATORY METHODS	DATA COMPUTERIZED	PRIMARY TYPES OF WELLS	STUDY TARGETED AROUND SPECIFIC LAND USE(S)	SAMPLING PERIOD (SINCE 1978)
* U.S. Geological Survey	YES	WATSTORE	Monit, Domes.	some studies	1978 to 1990
* MPCA Ambient	YES	STORET	Domes, Public	NO	1978 to 1989
* County Well Index	MOST	CWI	Domestic	NO	1978 to 1989
* U.S. Forest Service	YES	STORET	Campgrd, Picnic, Admin	NO	1978 to 1988
* MPCA Nonpoint Source	YES	STORET	Domestic, Monit.	Most Areas Agric	1987 to 1989
* SE Minn. Regional Lab	YES	PC Olmsted	Domestic	NO	1981 to 1990
* MDA pest/nutrient wells	YES	PC MDA	Monitoring	Agric. Fields	1987 to 1990
MDH Municipal Well Rec.	YES	NO	Municipal	NO	1985 to 1988
MDH New Well Construction	YES	MDH LAB	Domestic	NO	1988 to 1991
MDH Metro Landfill Study	YES	MDH	Domestic, Public	Dump Sites	1985 to 1990
MDA Pesticide Survey	YES	MDH network	Monit, Domest, Irrig.	Agric. Fields	1986 to 1987
MDH Pest. Survey (Public)	YES	MDH network	Munic, Public	Agr. Areas of MN	1986 to 1987
MDH Pest. Survey (Private	YES	MDH network	Domestic	Agr. Areas of MN	1986 to 1987
Community Health Services	SOME	NO	Domestic	NO	1988 to 1989
* South Central Counties	FEW	Mankato State	Domestic	NO	1978 to 1989
Rock and Nobles	YES	PC R&N cty's	Domestic	NO	1990

DATA SET	WELL LOCATION ON COMPUTER (* location from driller or owner)	AQUIFER INFORM. AVAILABLE	WELL DEPTH INFORM. AVAILABLE	# OF CO'S WITH DATA	NUMBER OF WELLS	NUMBER OF ANALYSES SINCE 1978	% WELLS WITH NO3-N > 1 MG/L	% WELLS WITH NO3-N >10 MG/L
U.S. Geological Survey	YES	MOST	MOST	62	841	2226	39%	11%
MPCA Ambient	YES	YES	MOST	87	484	990	24%	7%
County Well Index	* YES	HALF	MOST	55	8085	9291	33%	4%
U.S. Forest Service	YES	NO	NO	6	114	502	4%	0%
MPCA Nonpoint Source	YES	YES	MOST	4	71	384	79%	44%
SE Minn. Regional Lab	* MANY (67%)	NO	SOME (owner)	7	4728	5727	48%	12.5%
MDA pest/nutrient wells	YES	YES	MOST	22	95	413	66%	30%
MDH Municipal Well Rec.	NO	SOME	MANY	87	1678	-	21%	1.2%
MDH New Well Construction	NO	NO	NO	-	6899	9899	13% >0.4	1.4%
MDH Metro Landfill Study	YES	SOME	SOME	7	1302	-	28% >0.4	7.4%
MDA Pesticide Survey	NO	YES	YES		100	-	64%	27%
MDH Pest. Survey (Public)	YES	SOME	MANY (owner)	77	395	515	47% >0.4	7.1%
MDH Pest. Survey (Private	YES	SOME	MANY (owner)	30	199	399	71% >0.4	42.2%
Community Health Services	NO	NO	NO	67	-	26306	-	4.7%
South Central Counties	* SOME	NO	MANY (owner)	7	3588	4085	32%	6%
Rock and Nobles	YES		MANY (owner)	2	1350	1350	84% >1.5	34%

Figure B-10 Percent of Wells with Nitrate-N greater than 10 mg/l from Selected Data Sets

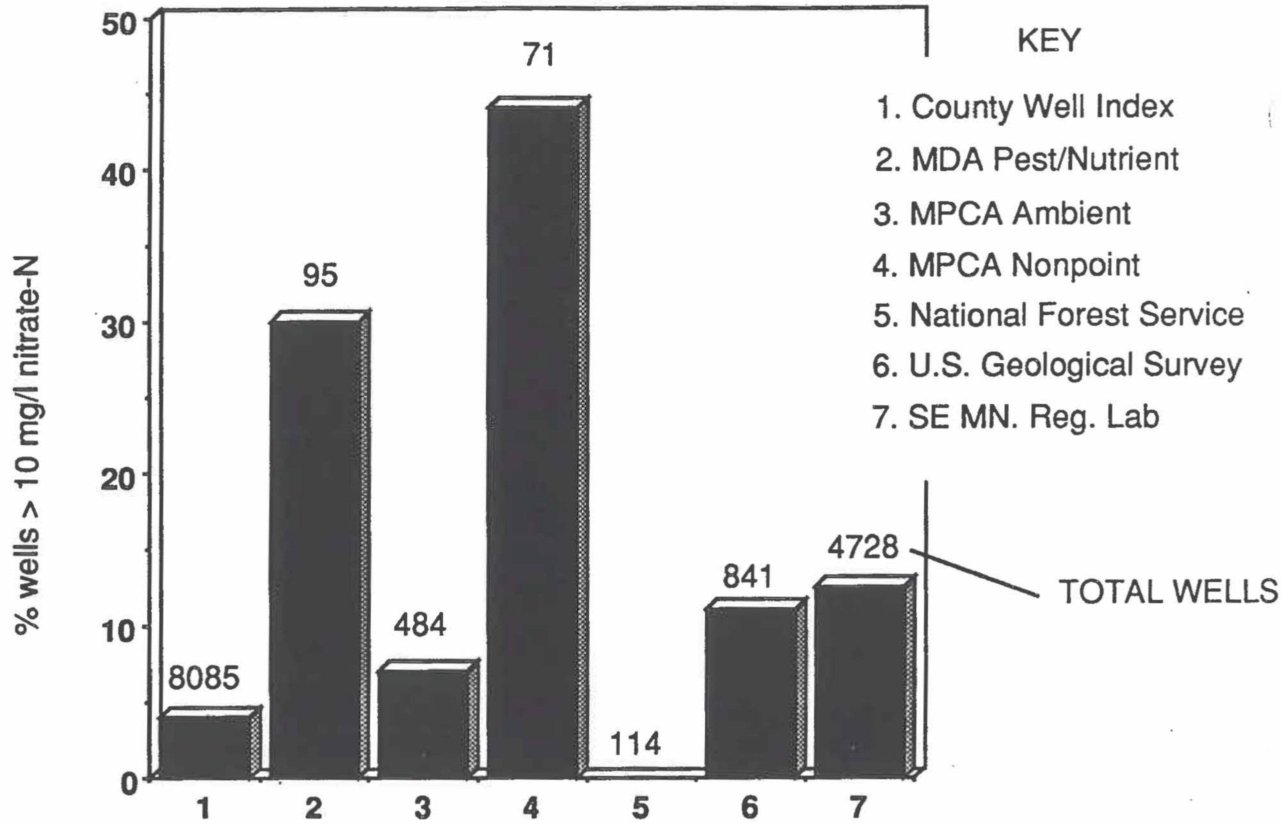


Figure B-11 Percent of Wells with Nitrate-N greater than 10 mg/l from Miscellaneous Data Sets

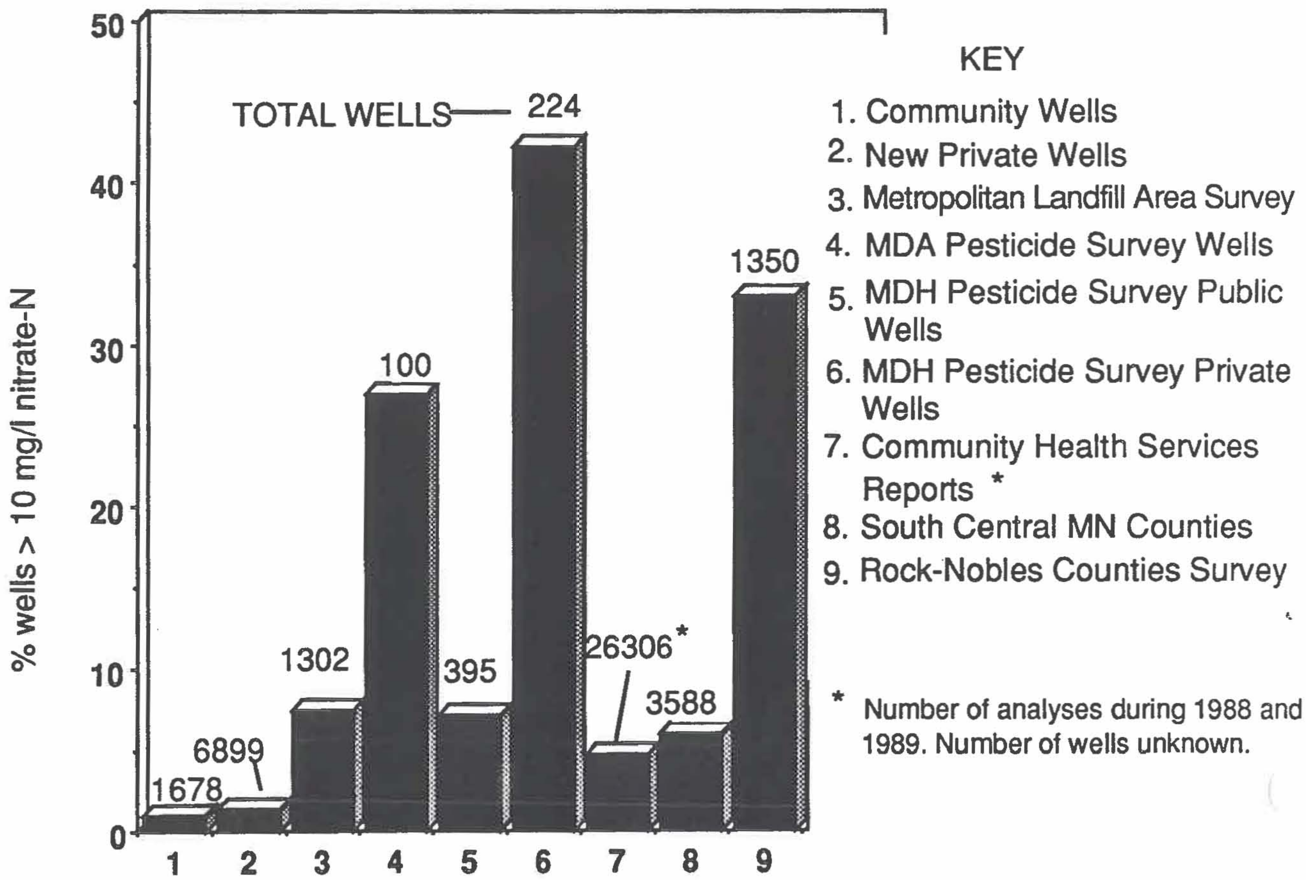


Figure B-12 Percent of Wells with Nitrate-N greater than 5 mg/l from Selected Data Sets

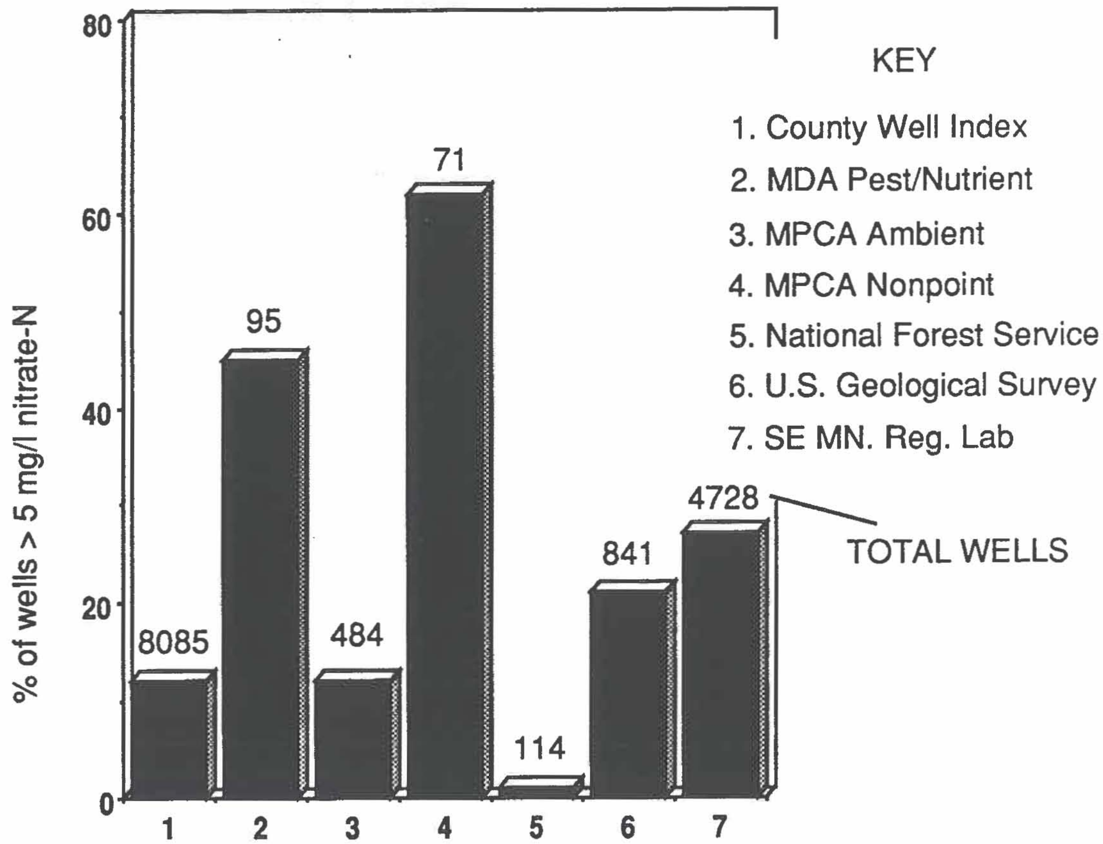
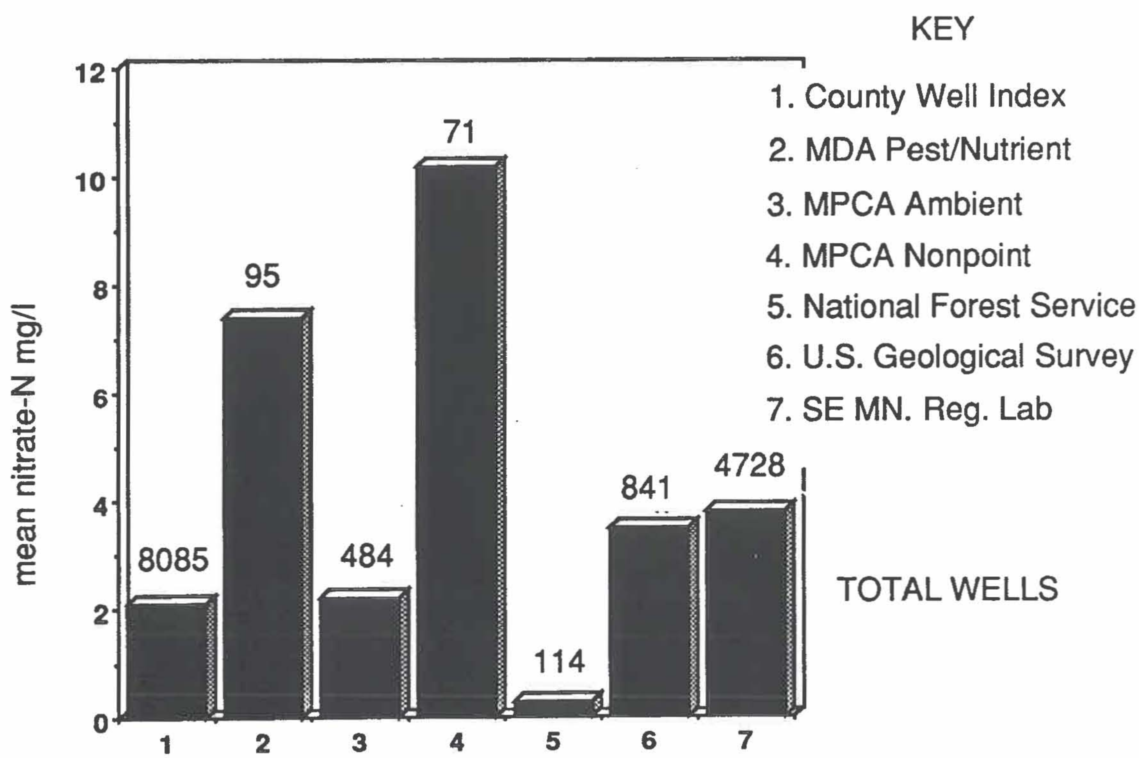


Figure B-13 Mean Nitrate-N from Selected Data Sets



While several counties appear to have over fifteen percent of sampled wells with NO_3-N above 10 mg/l, many other counties have less than two percent of wells exceeding 10 mg/l. Since most municipal water supply systems have low NO_3 water, the percentage of the **state's population** drinking high NO_3 water is well below the percentage of **wells** with NO_3-N above 10 mg/l.

Spatial Trends Across Minnesota

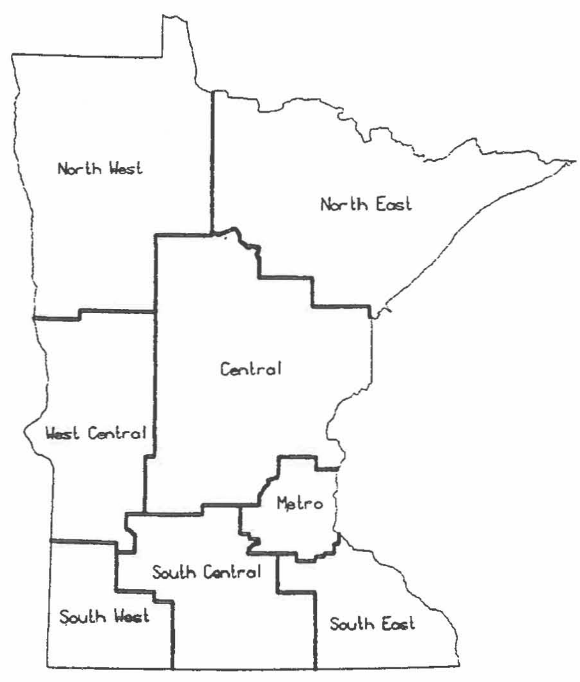
Due to issues discussed in the first section of this chapter entitled "Factors and Complexities Affecting Nitrate Concentrations," data from numerous wells sampled several times each is required to adequately assess the degree of NO_3 problems in a particular area. From the information presented in this report, some areas of the state appear to have severe NO_3 problems and other areas appear to have very minor impacts. In other areas there is very little information to assess the situation.

Nitrate information from **Northwest** Minnesota is sparse. Six of seven data sets with NO_3 information show very few NO_3 impacted wells, with the possible exception of Southern Beltrami County. The Community Health Services reports, however, show many elevated NO_3 levels in Beltrami County wells and a few high NO_3 wells in the other counties of this area.

Northeast Minnesota also had a limited number of wells in the various data sets. Based largely on community health services reports, National Forest Service monitoring and scattered wells from a few other sampling programs, this area of the state appears to have very few high NO_3 wells, with most wells having less than 1 mg/l NO_3-N.

A limited amount of NO_3 data is also available from **West-Central** Minnesota. While this area of the state does not appear to be as severely impacted as many other areas of the state, there are some areas of high NO_3 wells.

Central Minnesota has a wide range of ground water NO_3 conditions. Wells in many townships show no indication of NO_3 contamination. In other areas a relatively high percentage of wells exceed the NO_3 drinking water standard. Yet in other areas there is a mix of low and elevated NO_3 wells. Counties straight west and northeast of the Twin Cities metro area appear to be less impacted than counties in a line northwest of the Twin Cities.



The **southwestern** corner of the state appears to be one of the most severely impacted areas of the state according to Community Health Service reports, county surveys, and some wells sampled by MDH and MPCA. Some counties, such as Rock and Nobles have an abundance of NO_3 data and other counties such as Lincoln have few wells in existing data sets. Since much of the data from this area was produced by non-certifiable methods, further study may be needed.

In general, **south central** Minnesota has fewer high NO_3 wells than southeastern and southwestern Minnesota. Many townships in counties such as Waseca, Steele, Freeborn, Martin, Blue Earth, LeSueur and Faribault appear to have very few NO_3 impacted wells. Other areas in south central Minnesota have a significant number of high NO_3 wells. A fair amount of NO_3 data are available for this area from county sampling programs; however, the reliability of some of these data are questionable.

Much NO_3 data are available from the **Twin Cities** area. The northern half of the Metro area appears to have generally lower NO_3 ground water than the southern half of the metro area. Information from several data sets shows Dakota County with numerous high NO_3 wells. Southern Washington and some areas of Scott County also appear to have many elevated NO_3 wells. Hennepin County appears to have very few high NO_3 wells.

Nitrate information from a large number of wells are available for **southeastern** Minnesota through county sampling programs and County Well Index and other existing data sets. Southeastern Minnesota had many areas with numerous high NO_3 wells, especially in Goodhue, Wabasha, Winona, Fillmore, and Houston Counties. Rice, Dodge, Mower and western Olmsted County appear to have generally lower NO_3 than counties along the Mississippi River. However, high NO_3 wells are found in each of the southeastern Minnesota counties.

Differences Among Aquifers

Three data sets in the state had enough NO_3 data collected from different aquifers to allow limited comparison of NO_3 between aquifers. These three data sets, all of which were described earlier in the report, include County Well Index, MPCA-ambient, and U.S. Geological Survey data. The mean, median, and percent exceeding 10 mg/l, NO_3-N for each aquifer is shown in Tables B-22 to B-24. Descriptions and maps showing the extent of the principal aquifers in Minnesota are provided in Appendix A.

County Well Index - Aquifer Comparison

County Well Index NO_3 information from nine aquifers is shown in Table B-22. The most NO_3 impacted aquifer of the nine was the Decorah-Plattville-Glenwood, which is a bedrock aquifer found in areas of southeastern Minnesota. The Franconia-Ironton-Galesville and Mt. Simon-Hinckley aquifers have very few NO_3 impacted wells. These aquifers are older and deeper bedrock aquifers that are utilized throughout southeastern and parts of south central Minnesota. In most areas, the Franconia-Ironton-Galesville and Mt. Simon-Hinckley formations are protected by overlying confining units. The younger bedrock aquifers in southeastern Minnesota all showed some degree of NO_3-N impact, with between about three and eight percent of wells exceeding 10 mg/l.

Surficial drift aquifers had many more high NO_3 wells than the buried drift aquifers. Buried drift aquifers, which are "protected" by at least 10 feet of overlying clayey till, had a median NO_3-N concentration of 0.5 mg/l and one percent of the wells had NO_3 exceeding 10 mg/l. Surficial drift wells had a median NO_3-N concentration of 3.4 mg/l and 4.5 percent of wells exceeded 10 mg/l.

Table B-22 Comparison of Nitrate-N among aquifers for County Well Index.

Aquifer	# Wells	Mean (mg/l)	Median (mg/l)	% > 10 mg/l
Surficial Drift	379	3.9	3.4	4.5
Buried Drift	517	1.0	0.5	1.0
Decorah-Plattville-Glenwood	111	8.7	5.0	30.6
Platteville-St. Peter and St. Peter-Prairie	39	4.2	1.8	7.7
St. Peter	311	1.9	0.5	2.6
Prairie Du Chien	1118	2.6	1.4	3.4
Jordan	498	2.6	1.4	3.0
Franconia-Ironton-Galesville	229	1.0	0.4	0.4
Mt. Simon/Hinckley	13	0.17	<0.1	0

MPCA Ambient - Aquifer Comparison

MPCA ambient program NO_3 information from 16 aquifers is shown in Table B-23. Six aquifers are represented by less than 20 wells each and may not accurately reflect NO_3-N conditions in those aquifers. The most impacted aquifers are the younger bedrock aquifers in southeastern Minnesota, surficial drift aquifers, and the Sioux quartzite. The older formation aquifers in southeastern and northeastern Minnesota and Cretaceous aquifers were unimpacted or minimally impacted by NO_3. Buried drift, St. Peter Formation, and Prairie du Chien-Jordan aquifers had some NO_3 impacted wells, but had low overall mean and median NO_3 concentrations.

Table B-23 Comparison of Nitrate-N among aquifers for MPCA ambient wells.

Aquifer	# Wells	Mean (mg/l)	Median (mg/l)	% > 10 mg/l
Surficial Drift	96	4.1	0.4	15.6
Buried Drift	79	1.6	<0.01	5.1
Unspec.Glac.>50 Ft				
Cretaceous	31	0.02	<0.01	0
Cedar Valley-Maqu. Dubuque-Galena	23	5.1	4.9	9
Decorah-Plattville-Glenwood	5	4.7	0.44	20
St. Peter	47	0.52	<0.01	2
Prairie Du Chien	22	5.0	0.06	18
Prairie Du Chien-Jordan	25	0.8	0.02	4
Jordan	43	1.15	0.5	0
St. Lawrence & St. Lawr-Franc.	5	0.29	0.02	0
Franconia-Ironton-Galesville	29	0.26	0.01	0
Mt. Simon/Hinckley	23	1.9	0.01	4
No. Shore Volcanics	4	0.12	0.02	0
Biwabik Iron Fmt.	7	0.3	0.19	0
Sioux Quartzite	8	5.6	0.18	37.5
Precambrian Other	4	1.7	0.32	0

U.S. Geological Survey - Aquifer Comparison

U.S. Geological Survey NO_3 information from nine aquifers is shown in Table B-24. Four of these aquifers are represented by less than 20 wells. Surficial drift aquifers are the only aquifer classification with mean and median NO_3-N concentrations greater than 0.2 mg/l. Nineteen percent of USGS monitored surficial sand aquifer wells exceeded 10 mg/l NO_3-N. Only one of 60 buried drift wells exceeded 10 mg/l NO_3-N.

Table B-24 Comparison of Nitrate-N among aquifers for USGS wells.

Aquifer	# Wells	Mean (mg/l)	Median (mg/l)	% > 10 mg/l
Surficial Drift	363	5.7	2.4	19%
Buried Drift	60	0.6	0.05	2%
Cretaceous	10	0.03	0.01	0
Decorah-Plattville-Glenwood	17	0.08	0.03	0
St. Peter	21	0.06	0.05	0
Prairie Du Chien-Jordan	31	0.5	0.2	0
Jordan	7	0.6	0.2	0
Mt. Simon/Hinckley	13	0.21	0.2	0
Precambrian Other	7	0.06	0.02	0

Overall

In all three data sets the unconfined surficial sand aquifer wells were collectively much more NO_3 impacted than the buried drift wells. Very low NO_3 was a consistent trend in the older bedrock formations, including the St. Lawrence, Franconia, Ironton, Galesville, Mt. Simon and Hinckley Formation. Varying degrees of NO_3 contamination are evident in the other major bedrock aquifers, including the Cedar Valley-Maquoketa-Dubuque-Galena, Decorah-Plattville-Glenwood, St. Peter Sandstone and Prairie du Chien-Jordan.

Surficial Drift Aquifers

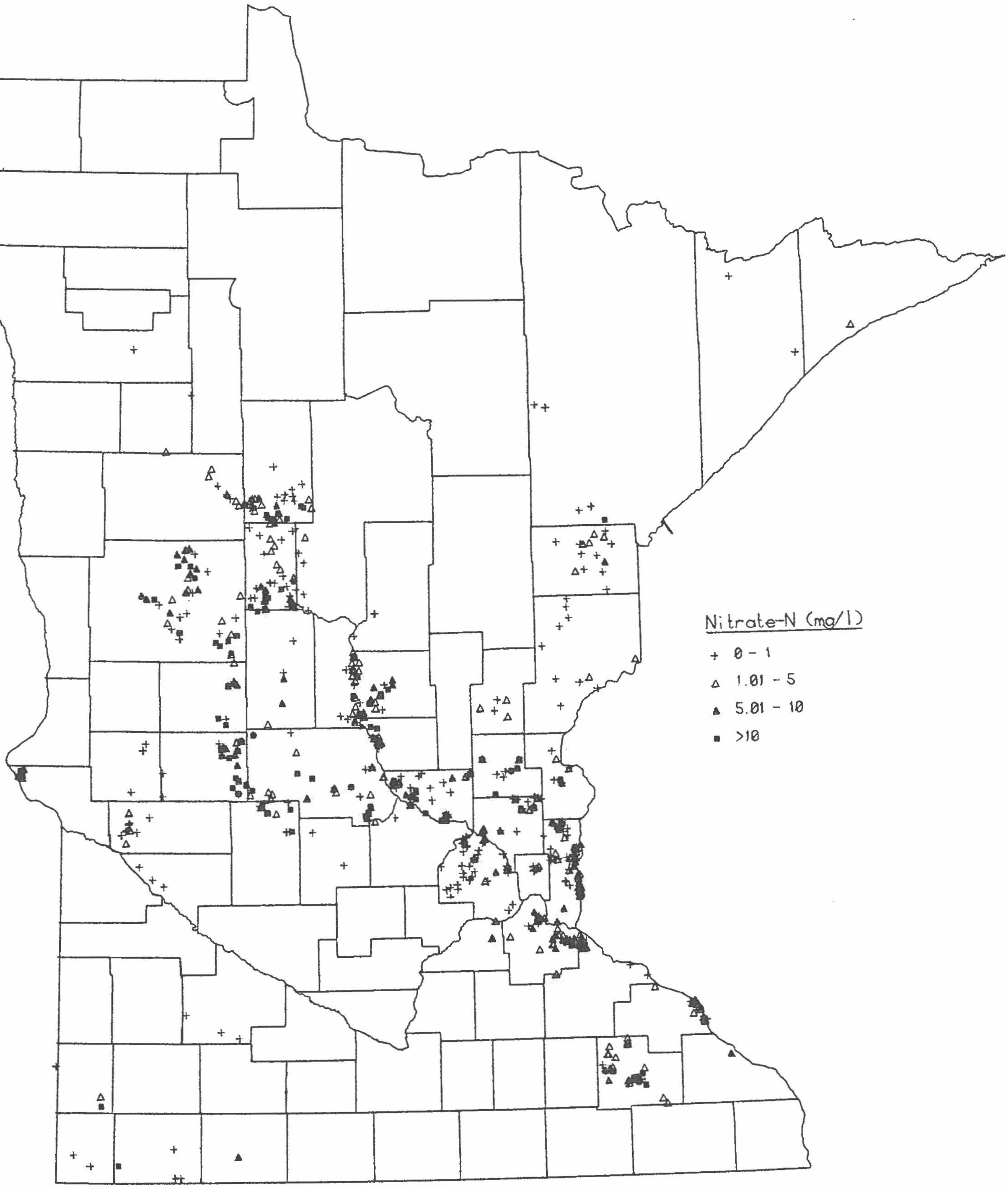
Surficial drift aquifers are found as large glacial outwash sand plain aquifers throughout much of central Minnesota and as alluvial aquifers along River Valleys throughout much of the state. Surficial drift aquifers are often vulnerable to contamination from activities at the land surface. These aquifers are very important to Minnesota because they usually yield high amounts of water with good natural water quality, are found throughout much of the state, and well construction costs are lower than other aquifers. Figure B-14 shows the locations and NO_3 levels in surficial drift aquifers throughout the state from five data sets (County Well Index, MPCA-Ambient, USGS, MPCA-Nonpoint Program, and MDA -- all previously described). Wells in these surficial aquifers display a wide range in NO_3 concentration, even within township sized areas. While most areas have at least some NO_3 impacted surficial drift wells, a few areas stand out as being more severely impacted.

Figure B-14

Surficial Sand Aquifer Wells

(Ambient, NFS, MDA, USGS, CWI)

Well locations and Nitrate levels



Geologic Sensitivity

Criteria and guidelines for assessing geologic sensitivity of ground water in Minnesota have recently been developed (MDNR, 1991). Geologic sensitivity criteria are proportional to the time required for a contaminant to move vertically from the ground surface to an aquifer. One might generally expect lower NO_3 concentrations in less sensitive areas since: 1) not enough "travel time" may have elapsed since nitrogen input increases to land that have occurred throughout the past few decades and 2) a longer travel time to the aquifer may allow a greater chance for denitrification to occur.

A statewide ground water contamination susceptibility map was created from maps of aquifer materials, recharge potential, soil materials, and vadose zone materials (Porcher, 1989). The susceptibility rankings were developed for the upper-most aquifers only and the resolution of the map units is on the order of one square mile. The map, which shows five different susceptibility rankings, is stored on computer at the Land Management Information Center (LMIC). For this study, NO_3 results from several data sets including Ambient, CWI, USGS, MDA, NPS, NFS and SEMN were compared with the susceptibility ranking at the well location (Table B-25). Nitrate concentrations were found to be generally higher in areas ranked in the two highest susceptibility categories compared to the middle susceptibility category. Not enough wells were located in the lowest two susceptibility categories to allow comparison of NO_3 for all five categories. Low nitrate inputs are often found in highly susceptible areas, and may help to explain the great number of wells in susceptible regions that have low nitrate.

Table B-25. Relationship between nitrate-N in CWI, ambient, USGS, MDA, NPS, NFS, and SEMN program wells and the susceptibility ranking at each well location.

Nitrate-N mg/l	# Wells	Ground Water Contamination Rank Susceptibility (Increasing to Right) 1	2	3	4	5	Total
0 - 1	8,589	0.1%	2.7%	29.0%	31.9%	36.2%	100%
1.01 - 5	2,848	0.0%	0.8%	13.0%	45.8%	40.4%	100%
5.01 -10	1,452	0.0%	0.4%	9.0%	47.7%	42.8%	100%
> 10	1,043	0.1%	0.4%	8.2%	57.6%	33.7%	100%
# Wells	13,932	13	268	3080	5338	5233	

The Minnesota Geological Survey (MGS) investigated geologic factors affecting the sensitivity of the Prairie du Chien-Jordan aquifer, which is used extensively throughout southeastern Minnesota (Setterholm et al., 1991). Nitrate was used as an indicator of sensitivity in the MGS study. A strong correlation was observed between NO_3 and the degree of protection provided by overlying glacial deposits. High NO_3 values tended to be found below thin or hydraulically conductive glacial sequences. The existence of a lower permeability unit of regional extent within the Prairie du Chien-Jordan aquifer was also found to affect NO_3 concentrations. Lower NO_3 concentrations were observed below the lower permeability unit where the unit was over ten feet thick. In a recent USGS study in Olmsted County, Jordan wells were found to be generally much less impacted than Prairie du Chien wells.[1]

Nitrate concentrations in 54 western Winona County wells were compared with a susceptibility map created as part of the Winona County geologic atlas (Wall and Regan, 1991). Mean and median NO_3 concentrations were found to be generally higher in the high sensitivity areas compared to areas ranked as moderate sensitivity.

It appears from these efforts that there is some sort of general relationship between sensitivity and NO_3 concentrations. However, sensitivity should not be equated with potential for NO_3 contamination. Other factors also greatly influence the likelihood of ground water NO_3 contamination (e.g., land use and management, water chemistry, and well construction and location).

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[1]Personal communication with Jim Stark, U.S. Geological Survey, St. Paul, Minnesota.

CHANGE IN NITRATE CONCENTRATION WITH TIME

Very few wells exist in Minnesota that have continuous sampling records to allow an analysis of changes in NO_3 concentration over a period of several years or more. In this chapter two data sets are discussed which provide very limited information regarding long-term NO_3 trends (12 to 40 year records) and trends since the mid 1980's.

Another long-term NO_3 record exists from the Big Spring Basin in northeast Iowa. Since land use, soils and geology in the Big Spring Basin are similar to many areas of southeast Minnesota, the Big Spring Basin NO_3 trends are pertinent to this study. The wholly agricultural basin is 103 square miles in size and has ground water NO_3 measurements dating back to the 1930's. A discussion of the NO_3 trends in the Big Spring Basin is provided in Chapter G (pg. G-33).

Municipal Well Nitrate Records

Introduction

Records of municipal well water chemistry are kept on microfiche files at the Minnesota Department of Health. Some of the NO_3 records go back as far as 1947, making this data set unique for the purposes of assessing long-term trends. For this study, NO_3 data from the microfiche were extracted for wells that had elevated NO_3 (greater than 5 mg/l NO_3-N) during the most recent testing. The primary purpose for analyzing these data was to see if wells with currently elevated NO_3 show any consistent long-term NO_3 trends. There is also some question about the integrity of some of the data obtained in the 1940's and 1950's.

While municipal water distribution systems, which are often a mix of water from several wells, are regularly analyzed for NO_3 and other parameters, the individual municipal wells are usually sampled less frequently. However, for trends analysis, it was necessary to look at NO_3 concentrations from individual wells rather than the distribution systems. Twenty-nine municipal well records (with NO_3-N > 5 mg/l) were found to have had at least five NO_3 measurements taken over a 12 to 40 year period. No regularity or consistency in sampling dates and frequency were evident. There is also some question about the integrity of some of the data obtained in the 1940's and 1950's.

Discussion of Results

A number of inherent difficulties exist when trying to use data from these municipal well records to draw conclusions about long-term NO_3 trends in Minnesota aquifers. These difficulties and some noted trends are discussed in the following paragraphs.

Nitrate concentrations in many wells show great variability over relatively short periods of time (Figures B-15 to B-18). This variability can be due to a number of factors that could include seasonal variability, short and long-term climatic conditions, land use changes, complex relationships between timing of nitrogen releases and soils and hydrogeologic conditions, laboratory error, and changes in pumping rate. A large number of data points are needed to assess long-term trends in wells with such great short-term variability.

Several long-term municipal well records have big gaps where no NO_3 measurements were made in over 12 years of the period of record (see Figures B-19 to B-21). Unless many data points are collected before and after the sampling lapse period, it is difficult to conclude much about the long-term trends where such data gaps exist.

It is evident from the municipal NO_3 measurements that elevated NO_3-N (7 to 24 mg/l) occurred as far back as the late 1940's in some cities (Figures B-22 to B-24). Further evidence of high nitrate in well water during the late 1940's is found in Bosch et al., 1950. There could have been several potential sources for this elevated NO_3 in the 1940's. It was noted in letters found in the well record microfiche files at MDH that municipal sewage problems were thought to be likely sources of contamination of many wells in the 1940's, 50's and 60's. Great strides have been made in municipal wastewater treatment since this period. It is possible that different nitrogen sources could be responsible for elevated NO_3 in a given well throughout its history.

Some municipal wells have had fairly consistent NO_3 levels throughout their period of record (Figures B-25 to B-28). Other wells had fairly stable levels for many years and then showed a sudden increase that may be due to an anomalous data point (Figures B-29 to B-30). It is apparent in other wells that NO_3 increases have occurred with time (Figures B-31 to B-35). Yet NO_3 concentrations in other wells have shown decreasing trends (Figure B-36 and Figure B-18).

This analysis of municipal well NO_3 data is inconclusive about the long-term temporal trends of NO_3 in Minnesota. While there appears to be more increasing trends than decreasing trends in municipal wells, the relatively small number of wells analyzed and inconsistency in trends limits the utility of this data set in drawing regional or statewide conclusions regarding long-term NO_3 trends in ground water.

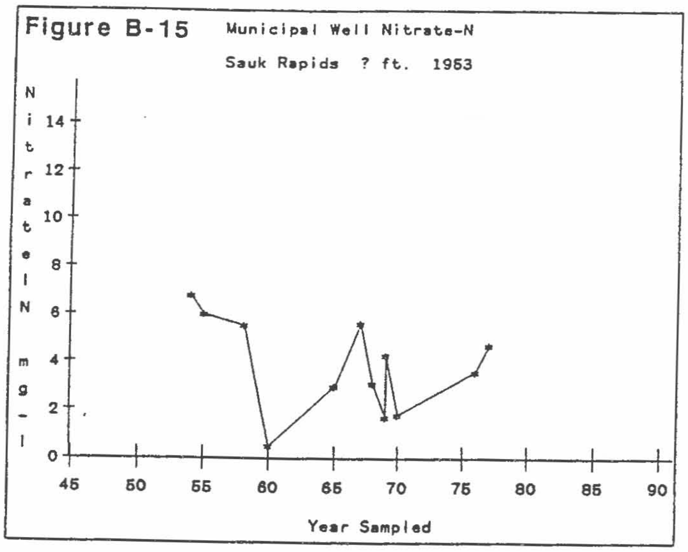


Figure B-15
Municipal Well Nitrate-N
Sauk Rapids ? ft. 1953
Nitrate N mg/l
Year Sampled

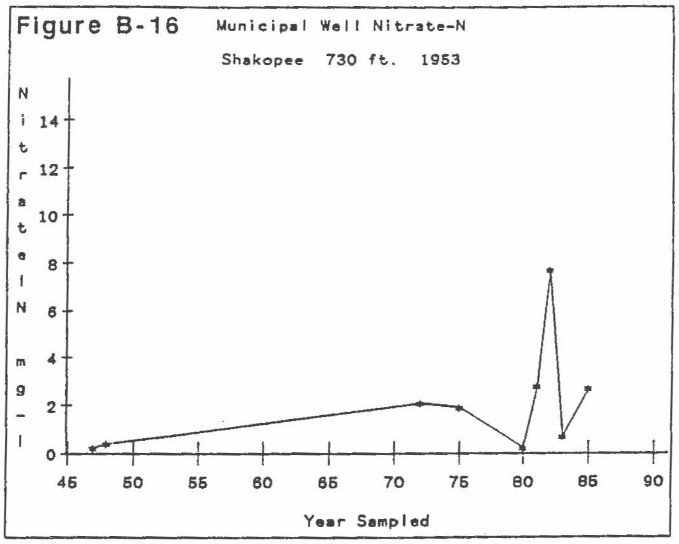


Figure B-16
Municipal Well Nitrate-N
Shakopee 730 ft. 1953
Nitrate N mg/l
Year Sampled

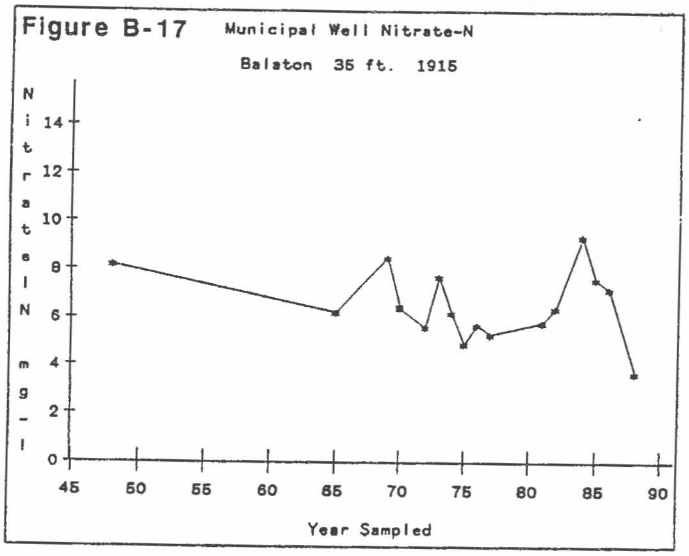


Figure B-17
Municipal Well Nitrate-N
Balaton 35 ft. 1915
Nitrate N mg/l
Year Sampled

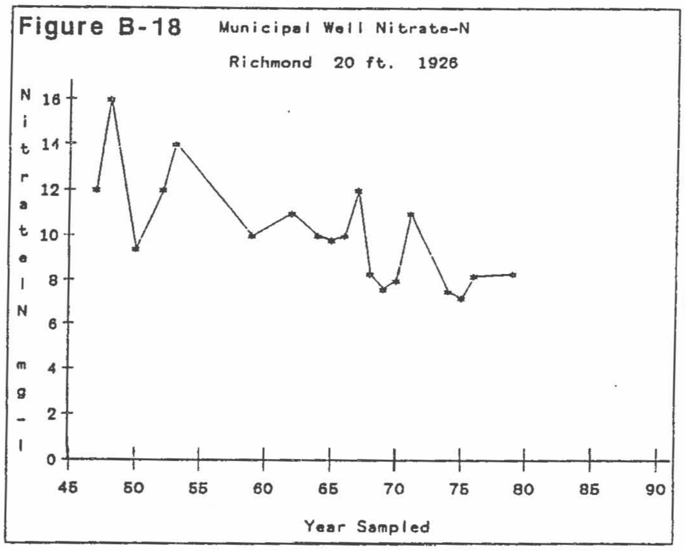


Figure B-18
Municipal Well Nitrate-N
Richmond 20 ft. 1926
Nitrate N mg/l
Year Sampled

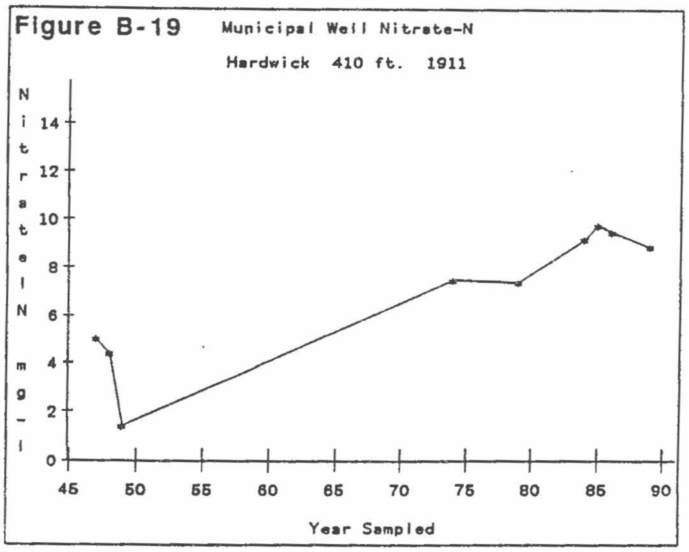


Figure B-19
Municipal Well Nitrate-N
Hardwick 410 ft. 1911
Nitrate N mg/l
Year Sampled

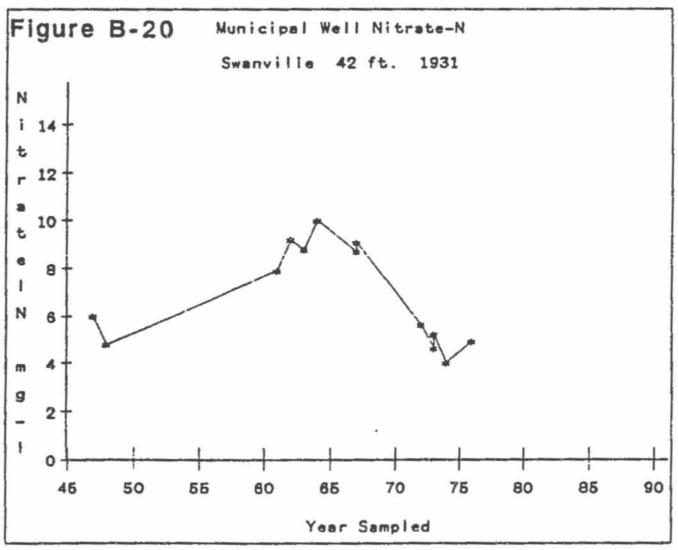
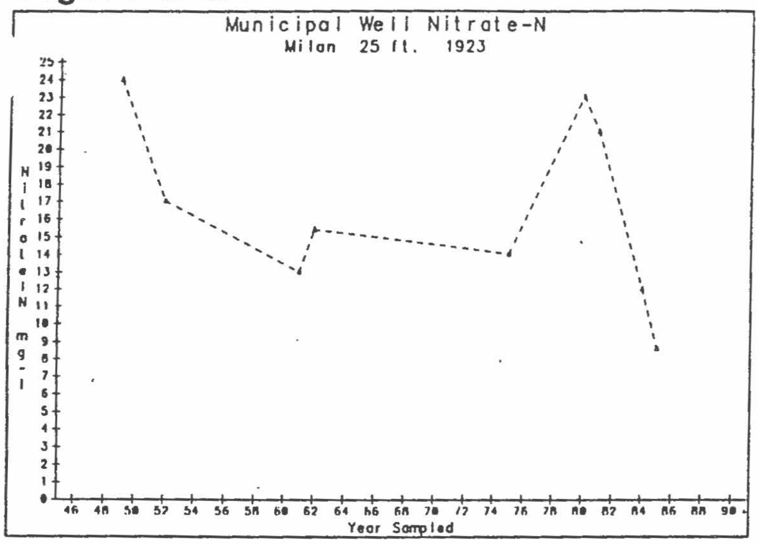


Figure B-20
Municipal Well Nitrate-N
Swanville 42 ft. 1931
Nitrate N mg/l
Year Sampled

Figure B-21


Municipal Well Nitrate-N
Milan 25 ft. 1923
Nitrate-N mg-l
Year Sampled

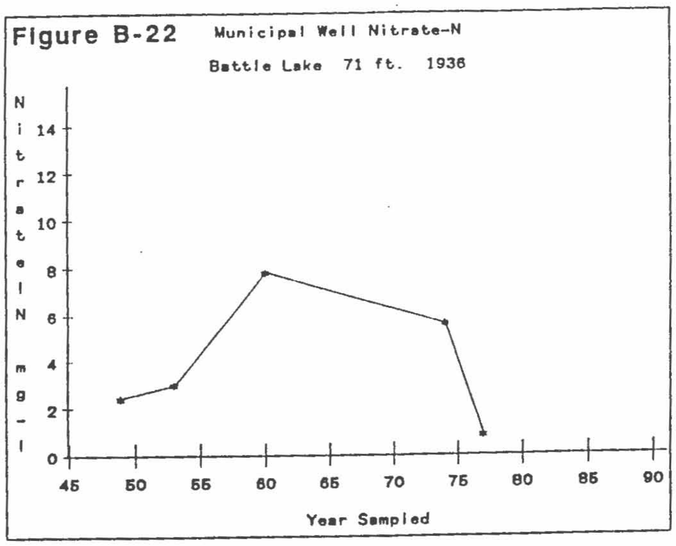


Figure B-22
Municipal Well Nitrate-N
Battle Lake 71 ft. 1936
Nitrate-N mg-l
Year Sampled

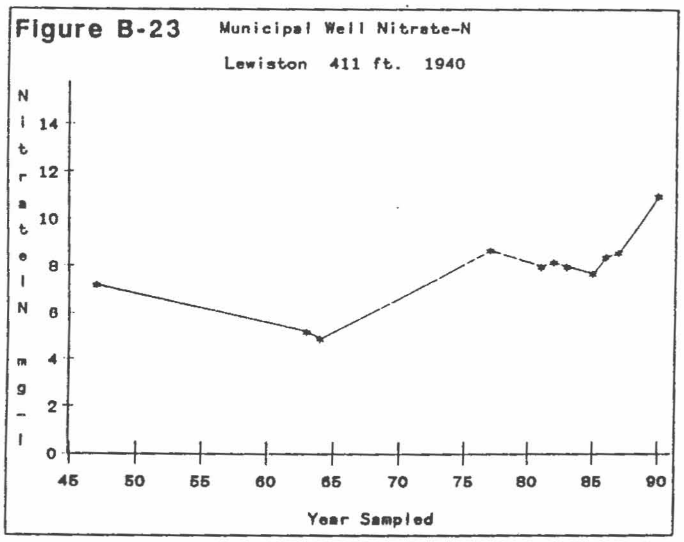


Figure B-23
Municipal Well Nitrate-N
Lewiston 411 ft. 1940
Nitrate-N mg-l
Year Sampled

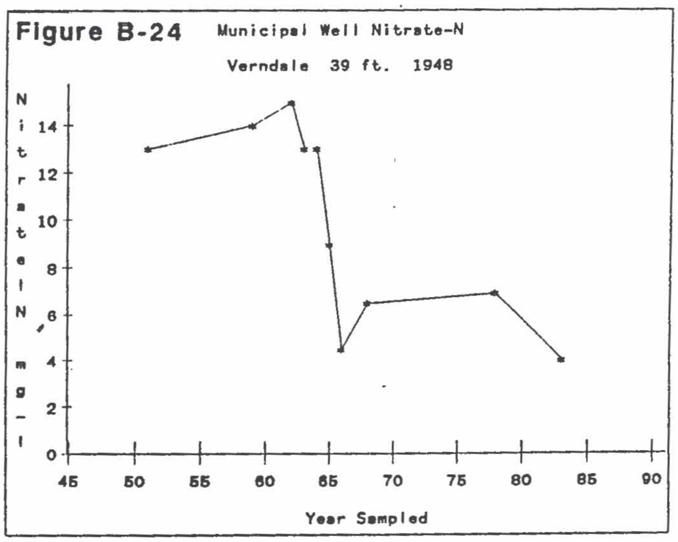


Figure B-24
Municipal Well Nitrate-N
Verndale 39 ft. 1948
Nitrate-N mg-l
Year Sampled

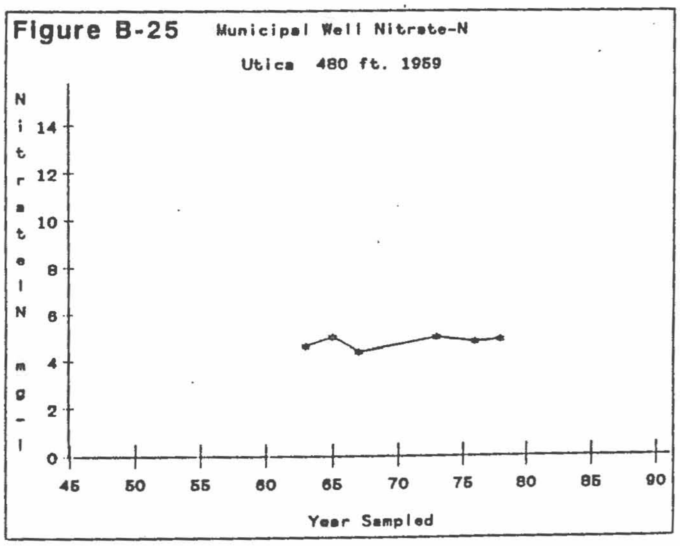


Figure B-25
Municipal Well Nitrate-N
Utica 480 ft. 1959
Nitrate-N mg-l
Year Sampled

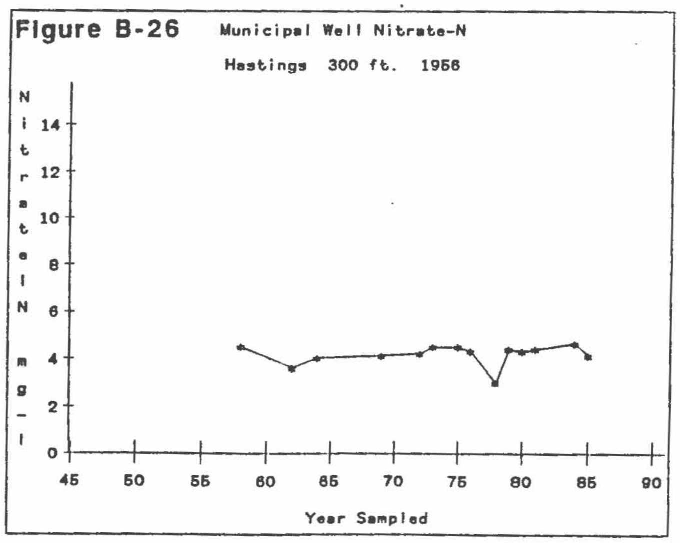


Figure B-26
Municipal Well Nitrate-N
Hastings 300 ft. 1956
Nitrate-N mg-l
Year Sampled

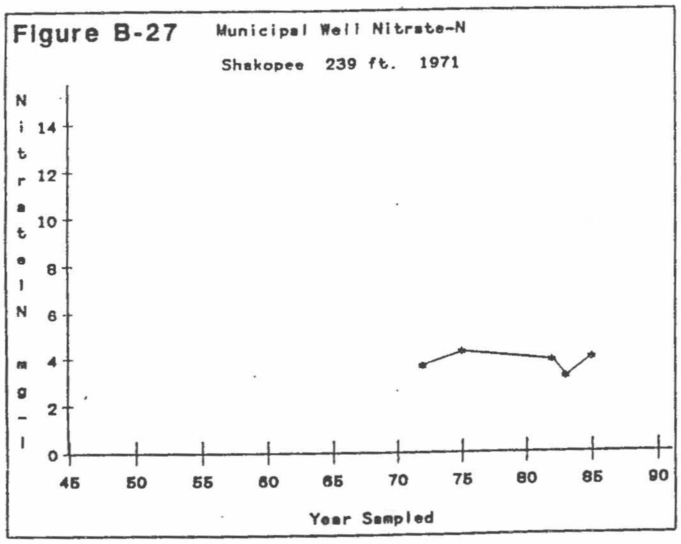


Figure B-27 Municipal Well Nitrate-N
Shakopee 239 ft. 1971

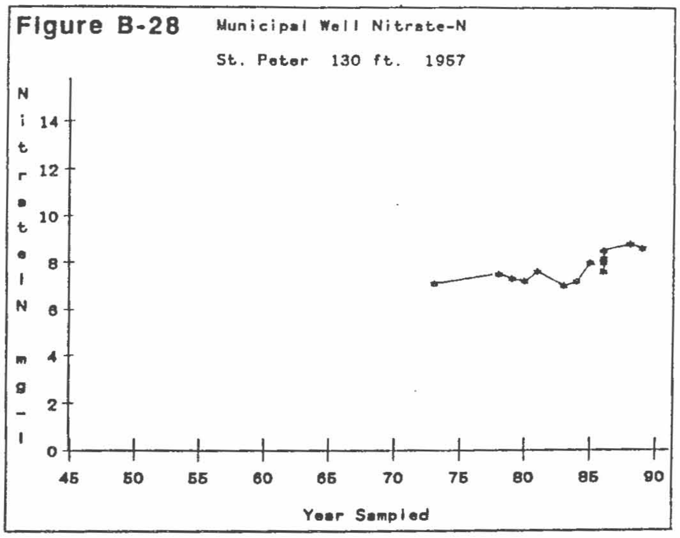


Figure B-28 Municipal Well Nitrate-N
St. Peter 130 ft. 1957

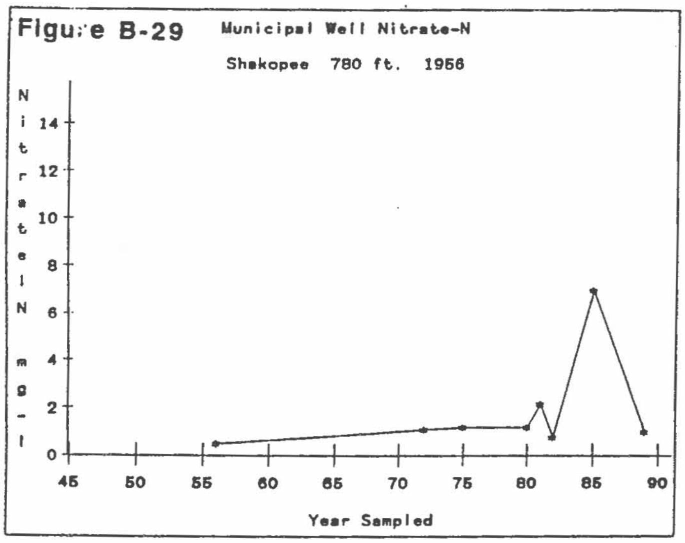


Figure B-29 Municipal Well Nitrate-N
Shakopee 780 ft. 1956

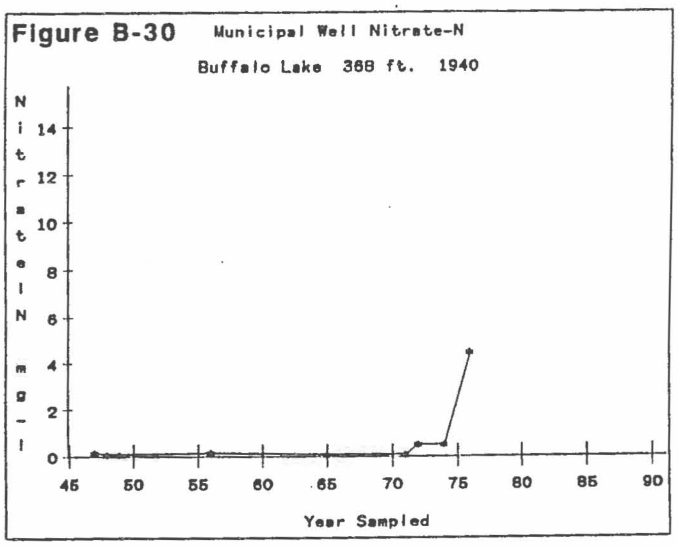


Figure B-30 Municipal Well Nitrate-N
Buffalo Lake 368 ft. 1940

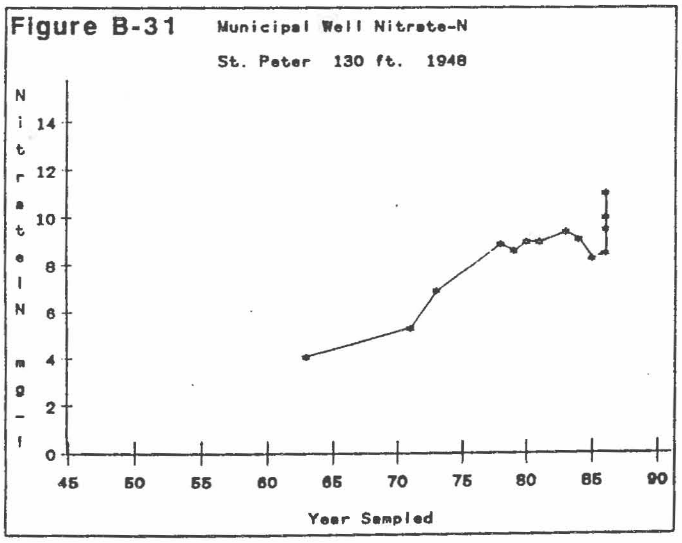


Figure B-31 Municipal Well Nitrate-N
St. Peter 130 ft. 1948

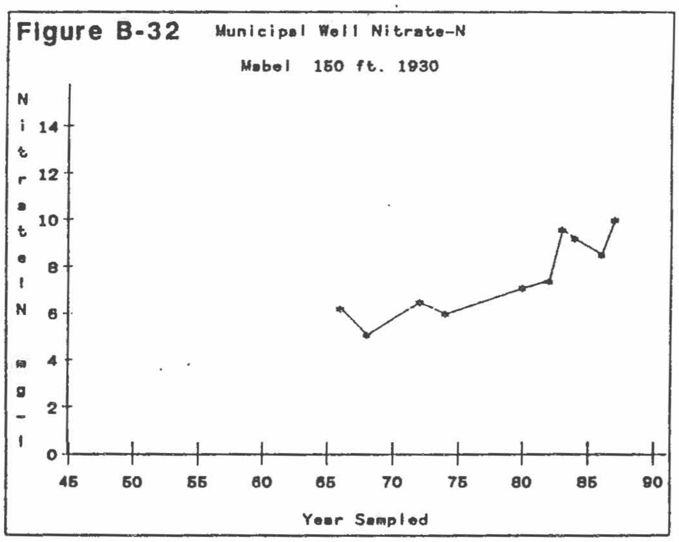


Figure B-32 Municipal Well Nitrate-N
Mabel 150 ft. 1930

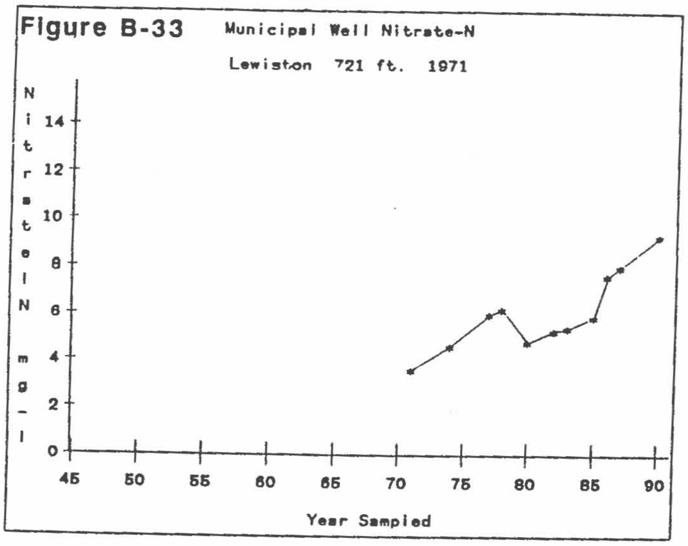


Figure B-33
Municipal Well Nitrate-N
Lewiston 721 ft. 1971
Nitrate-N mg-l
Year Sampled

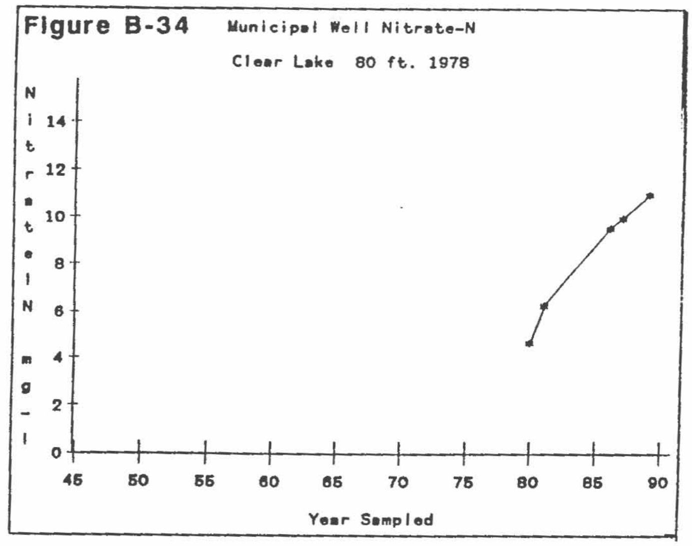


Figure B-34
Municipal Well Nitrate-N
Clear Lake 80 ft. 1978
Nitrate-N mg-l
Year Sampled

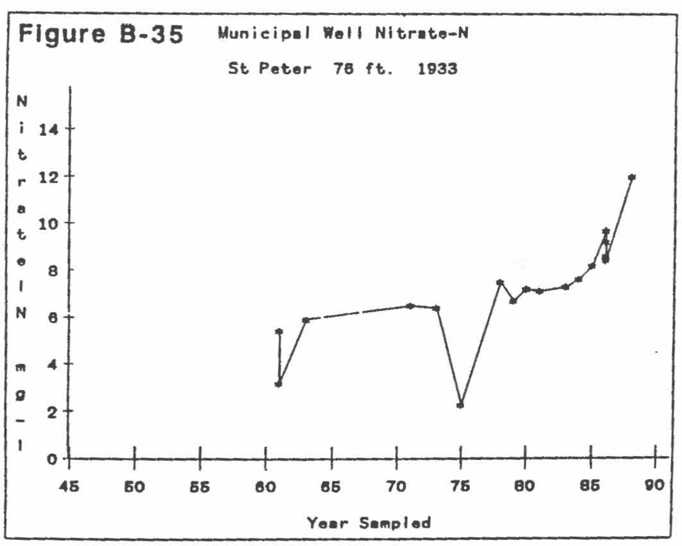


Figure B-35
Municipal Well Nitrate-N
St Peter 76 ft. 1933
Nitrate-N mg-l
Year Sampled

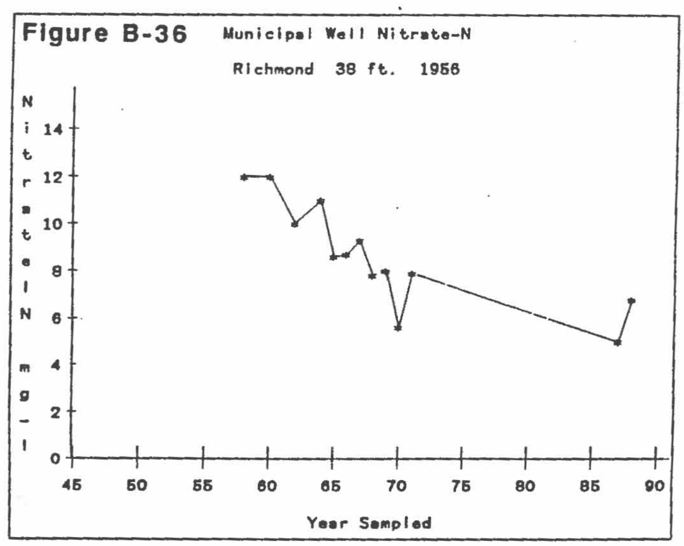


Figure B-36
Municipal Well Nitrate-N
Richmond 38 ft. 1956
Nitrate-N mg-l
Year Sampled

Minnesota Department of Agriculture Time Trend Analysis Network

The Minnesota Department of Agriculture Time Trend Analysis Monitoring Network has been active since 1986. The objective of the Network is to determine whether observed trends in pesticide concentrations in contaminated wells over time are statistically significant rather than the result of random (naturally occurring) variability.

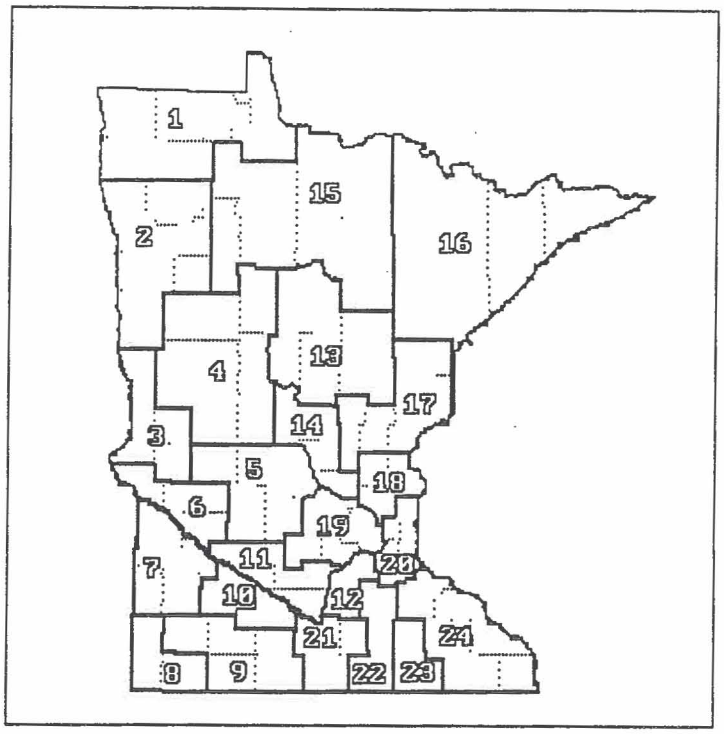
The term "trend" is defined, for the purposes of this network, as a general increase or decrease in the observed water quality variable (in this case pesticide concentration) over time. The program tests for monotonic (one directional), gradual (linear) trends.

The Time Trend Network wells are selected from those wells already included in the diagnostic network (see page B-26). The diagnostic network wells are selected based, in part, upon pesticide use and land management practices on adjacent lands. The protocol for time trend network well selection and sampling frequency are far more rigorous than those for the diagnostic network due to the statistical techniques central to time series analysis. Time trend network wells must have a history of pesticide detections of sufficient frequency to determine a concentration trend, or must appear to have a high likelihood of recurring pesticide detections during consecutive sampling quarters. Network wells are sampled quarterly.

For the Minnesota ground water monitoring networks, the state is divided into 24 regions, or county clusters, consisting of 2 to 6 counties in each (Figure B-37).

Figure B-37

Figure B-37
Delineation of County Clusters
Minnesota Department of Agriculture
Ground Water Monitoring Program



Currently, the time trend analysis network is maintained in four county clusters (Figure B-37). The majority of the wells represent quaternary water table aquifers (county clusters 4, 14, and 15); a smaller subnetwork is maintained in the southeast karst terrain (county cluster 24).

Quaternary water table aquifers (QWTA) are composed of unconsolidated sand and gravel deposits left by the melting of the most recent glaciers, and have no confining layer between the water table and the ground surface. Water table depths may range from less than ten feet to greater than 40 feet.

Karst terrain is characterized by fractured limestone or dolomitic bedrock, disappearing streams, springs, and sinkholes. The fractured bedrock, overlain by silty, loess-based soils, is the surface aquifer in this region. Bedrock fractures allow for rapid and unpredictable water movement to and within the aquifer. The network wells in this region are primarily domestic drinking water wells in the Prairie du Chien-Jordan aquifer.

Although the network is not statistically designed to study the occurrence of NO_3-N concentrations in ground water, samples are collected and submitted for NO_3-N analysis during the quarterly sampling events. The 1986 to 1990 NO_3-N data are summarized in Table B-26.

For the quaternary wells sampled over time, there is a statistically significant increasing trend (summary data, "all QWTA"), although results vary widely for individual QWTA wells. For the karst wells sampled over time, there is no statistically significant trend ("all karst") and results vary widely for individual wells within the region.

The reader should note that this monitoring network was not designed specifically to study occurrences and trends in NO_3-N data; nor should results be extrapolated to larger populations.

Table B-26 Minnesota Department of Agriculture Time Trend Analysis Network Linear Monotonic NO_3-N Trend Data

Quaternary Water Table Aquifer Wells and Southeastern Karst Wells

Cluster/Well	Significant at 80% Confidence Interval?	Slope ppm/yr	Direction of Slope
QWTA			
4/ 031002	No	0	NA
291001	Yes	1.45	-
801001	Inconclusive	0.35	-
211001	No	0.23	-
14/ 051001	No	1.3	-
491001	Inconclusive	0.5	+
491002	Yes	2.1	-
491003	Inconclusive	0.5	-
711002	Yes	5.9	+
711007	Yes	0.85	-
711008	Yes	3.1	+
5/ 341002	No	0.67	+
611001	Yes	1.7	+
611002	No	0.57	+
611003	Yes	1.22	+
611005	*	*	*
731001	Inconclusive	0.3	+
731003	Inconclusive	1.68	-
Karst			
24/ 852001	Yes	0.9	-
852002	Yes	1.49	-
852004	No	0.1	+
852005	Yes	0.75	-
Summary			
All QWTA	Yes	1.1	+
All Karst	Inconclusive	0.82	+

QWTA = Quaternary Water Table Aquifer
* = Nitrate has never been detected

RECOMMENDATIONS - NITRATE MONITORING NEEDS

The water quality data summarized in this report clearly illustrates that NO_3 contamination of ground water resources is a problem in Minnesota. However, existing data does not provide information necessary for implementing effective water resource protection measures to ensure a safe source of drinking water for current users of domestic water wells and future generations. Existing data do not provide information to adequately answer the following questions. Where should the state target ground water protection and drinking water protection activities? What are the long term trends in concentration of nitrate in Minnesota? Are current ground water protection efforts improving the water quality? The purpose of this section is to provide a long term and statewide strategy for collecting the information that will be useful for addressing these issues in the decades ahead.

The overall goal of the strategy is to fulfill local and state government needs for reliable and useful information for managing Minnesota's ground water resources in the upcoming decades in a cost efficient manner. Ground water monitoring is expensive; not every well can be analyzed frequently enough to assure drinking water protection for the user. A cost-effective means of providing drinking water protection is by monitoring and managing the natural resource.

The following management objectives target the need for useful information with several specific recommendations that are necessary to fulfill each objective. The recommendations that are feasible approaches that build upon existing programs and emphasize coordination between state and local government to monitor the ground water resources. Successful implementation of these recommendations requires that the State make monitoring for nitrate a priority in Minnesota. The recommendations have been organized into three fundamental management objectives.

Management Objective #1: Identification of Nitrate Priority Areas.

Minnesota needs reliable, statewide information to identify areas and aquifers where concentrations of NO_3-N currently exceed or are approaching drinking water guidelines established by the Minnesota Department of Health. This information is necessary to target implementation of best management practices and other ground water protection efforts, and provide an increased level of drinking water protection for domestic water supply uses.

This objective requires long term monitoring at locations that systematically cover the state with frequency of sample collection based upon existing water quality data, surrounding land uses, and hydrogeologic conditions. The well information must be adequate to identify the aquifer and location of the monitoring point. Each water sample must be analyzed by the most reliable laboratory techniques. The data must be maintained on a computerized data base that is available to all interested parties. Strict quality assurance and quality control measures should be followed throughout sample collection, laboratory analysis and data management.

Recommended Actions:

#1A Establish a long term and stable funding source for the collection and evaluation of ground water quality data on a statewide basis.

There are currently two programs that collect and evaluate ground water quality data on a long term and statewide basis in Minnesota. The MDA Pesticide and Nutrient Monitoring Program is designed to evaluate agricultural impacts on ground water quality. The MPCA Statewide Ground Water Quality Monitoring Program (previously known as the ambient monitoring program) assesses baseline conditions in Minnesota's 14 principal aquifers. The success of these programs to generate useful information for managing Minnesota's ground water resources relies on long term, stable funding.

#1B Establish statewide standards for collection and analysis of NO_3 data.

Statewide standards should address locational information, documentation of well construction, laboratory techniques used to analyze water samples, and management of NO_3 data. The standards should be applied to all programs and projects receiving public funds.

#1C Maintain a statewide computerized NO_3 data registry for water quality results that meet selected data standards.

These data should be maintained by a single program in order to ensure that proper quality assurance and quality control practices are followed. The data should be available to all interested parties. Computerized data that meets the minimum standards (#1B) for NO_3 would be useful to existing state programs for identifying NO_3 hotspots.

#1D Enhance current state programs, such as the MDA Pesticide and Nutrient Monitoring Program, to include providing technical assistance to local units of government for monitoring NO_3.

Many local units of government are initiating ground water monitoring programs to supplement monitoring currently being conducted by the State. A pilot project in southeastern Minnesota is bringing together local units of government, academia, and state agencies to assess the regional ground water quality conditions. This project integrates the ground water monitoring expertise found in state government and academia with local knowledge and local commitment to ground water protection. Cooperative monitoring ensures that data collection efforts are coordinated in Minnesota. The MDA contributions to cooperative efforts are currently limited by insufficient staff resources.

Management Objective #2: Evaluate Long Term Changes in Nitrate Concentrations in Minnesota.

What are the long term trends in concentration of nitrate in Minnesota?

Reliable information is needed to evaluate nitrate concentration time trends in Minnesota. Long term trend analysis requires uninterrupted sample collection and analysis over a long period of time (usually at least five years) at a regular interval (usually four to twelve times per year from each well). This high level of commitment is needed to distinguish long term overall trends in levels of nitrate from seasonal and annual fluctuations in nitrate concentrations. Because time trend analysis is expensive to complete and is most meaningful on a local scale, it is recommended that this type of monitoring target nitrate hotspots identified in objective #1.

Time trend analysis is useful for predicting when a hotspot may become a public health concern. This information is useful for water resource managers to develop appropriate techniques for mitigating nitrate impacts on ground water and protecting drinking water sources. For instance, this information can be used to predict when a community wastewater treatment system should replace individual septic systems, or when and where testing of domestic supply wells serving infants should be conducted. This information will also help ensure ground water protection measures are appropriate - not too strict or permissive.

Recommended Actions:

#2A Amend the Ground Water Protection Act to clearly delegate authority for the MDA Pesticide and Nutrients Monitoring Program to work with local units of government to evaluate the occurrence of nutrients in ground water quality conditions.

#2B Enhance the MPCA Ground Water Monitoring and Assessment Program to conduct time trend analysis at nitrate priority areas.

This program is currently funded to evaluate the concentration of volatile organic compounds in heavily developed areas. Nitrate monitoring in problem areas could be easily included in the operation of this program with the addition of funds to cover the collection and analysis of water samples.

Management Objective #3: Evaluate the Effectiveness of Statewide and Current Ground Water Protection Efforts in Targeted Areas.

Minnesota needs reliable ground water quality information to evaluate the effectiveness of the nitrogen fertilizer management plan and best management practices for preventing contamination of ground water resources with NO_3-N. This information is critical for prioritizing future ground water protection efforts.

This objective requires uninterrupted collection of ground water samples for several years at locations where ground water protection measures have been employed. Monitoring efforts for shorter periods of time are useless for this type of evaluation. The frequency of sample collection is determined by the hydrogeologic conditions at the site and may range from four to twenty-four times per year.

Recommended Actions:

#2A Conduct long term monitoring at Clean Water Partnership Project areas where best management practices are implemented.

Several Clean Water Partnership Projects will be implementing protection measures for nitrate in ground water. Long term monitoring requires a stable source of funding and statistical expertise for data evaluation. The MPCA maintains one ground water monitoring program that collects and evaluates water quality information on a long term and statewide basis. Enhancing the Statewide Ground Water Quality Monitoring Program would be a cost effective approach to meet this need for water quality information.

#2B Conduct Long term monitoring in association with the implementation of the Nitrogen Fertilizer Management Plan.

Implementation of the Nitrogen Fertilizer Management Plan should include funding for long term regional monitoring to evaluate the effectiveness of the management plan. Long term monitoring at the MDA is currently conducted by the Pesticide and Nutrient Monitoring Program. This program would require additional funding to address this ground water informational need.

SUMMARY

The nitrate (NO_3) concentration in any given sample of well water is the combined effect of numerous factors, including surrounding land use and management, ground water flow hydraulics, ground water residence time, climatic conditions, ground water chemistry, well depth in relation to geologic stratigraphy and water table elevation, type of well sampled and well construction.

Minnesota does not have a statewide ground water monitoring program in place designed specifically to assess the extent and trends of ground water NO_3 concentrations. Nitrate data have been collected in Minnesota through various federal, state and local programs, with most of this information generated since the late 1970's. For this report, 16 data sets with NO_3 information were examined in an attempt to better understand the degree of nitrate problems in Minnesota and ground water NO_3 variability across the state. EPA approved methods were used to produce data in 14 data sets that represent a total of about 26,340 wells. Certain data sets also provided limited information regarding NO_3 differences between aquifers and changes in NO_3 concentration with time.

It may appear upon casual examination that there is an abundance of data to make good estimations of the current NO_3 status and trends in Minnesota. However, there are great differences between existing data sets in sampling purpose, field and laboratory methodologies, areas sampled, years and frequency of sampling, data management, and documented well location and construction information (Table B-20). These differences limit the utility of the data in assessing current statewide conditions.

Computerized data from seven data sets were obtained, evaluated, and described for this report. Nitrate data from these seven data sets were collected since 1978, produced by EPA approved methods and had associated well location information. Results from nine other miscellaneous nitrate data sets were also described. Summary statistics were available for eight of the nine data sets from either literature or representatives of the group collecting the data. Most of these nine data sets did not have readily accessible detailed well location information. Data from two data sets were produced by methods not approved by EPA. Wells sampled for one data set were located within a one-mile radius of dump sites.

The data summarized in this chapter clearly illustrate that nitrate contamination of ground water resources is a problem in many areas of Minnesota. Major differences in ground water NO_3 conditions are found when comparing results from the sixteen data sets (Table B-20). Sampling programs targeting wells in geologically-sensitive areas under agricultural production show a relatively high percentage (27 to 44%) of wells exceeding 10 mg/l NO3-N. Sampling programs targeting newly constructed wells or municipal wells showed a much lower percentage (1 to 4%) of wells with NO_3-N exceeding 10 mg/l. MPCA ambient monitoring program results from 484 wells in different aquifers throughout the state showed NO_3-N exceeding 10 mg/l in 7 percent of the wells sampled. County sampling program results were quite varied throughout the state.

Some areas of the state appear to have NO_3 problems and other areas appear to have only minor impacts. In other areas, there is very little information to assess the situation. A majority of the NO_3 data has been collected in the southern half of the state, particularly southeastern Minnesota (including the Twin Cities). Limited data indicate relatively few wells with elevated NO_3 in northeastern and northwestern Minnesota. Central Minnesota appears to have a wide range of ground water NO_3 conditions. Southeast and southwest Minnesota appear to be two regions where a relatively high percentage of wells are NO_3 impacted; however, there is great variability in the degree of NO_3 impact within these regions. South central and west central Minnesota have less evidence of NO_3 problems than southeast and southwest Minnesota, but both of these regions have high NO_3 wells in certain areas. The northwest Twin Cities area appears to have fewer NO_3 problem areas than the southeast Metro area.

Three data sets had enough NO_3 data collected from different aquifers to allow limited comparison of NO_3 between aquifers. In all three data sets, unconfined surficial sand aquifer wells were generally much more NO_3 impacted than buried drift wells. Low nitrate was a consistent trend in older bedrock formation aquifers of the southeastern quarter of the state (St. Lawrence, Franconia, Ironton, Galesville, Mt. Simon and Hinkley formations). Varying degrees of NO_3 contamination are evident in the other major bedrock aquifers in the southeastern quarter of the state, including the Cedar Valley-Maquoketa-Dubuque-Galena, Decorah-Platteville-Glenwood, St. Peter and Prairie du Chien-Jordan. The degree of protection provided by overlying glacial deposits appears to be an important factor affecting nitrate levels in the Prairie du Chien-Jordan aquifer.

There are very few wells in Minnesota that have continuous nitrate sampling records sufficient for time-trend analysis. Twenty-two monitoring wells have been sampled quarterly since 1986 by MDA. Trend results show some wells with increasing trends and other wells with decreasing trends. In addition to the MDA well data analysis, 29 MDH municipal well records were visually examined for this report. These 29 wells had 1) elevated (>5 mg/l) NO3-N during recent tests, and 2) at least five measurements taken over a 12 to 40 year period. Data integrity uncertainties exist with this historic data set. While some wells appear to exhibit increasing trends, other wells appear to exhibit decreasing trends. Yet other wells have very consistent concentrations with time. Erratic nitrate levels and large gaps in the period of record were found in several of the well records. While there appeared to be more increasing trends than decreasing trends in municipal well records examined, the relatively small number of wells analyzed and inconsistency in trends limits the utility of this data set in drawing regional or statewide conclusions regarding long term NO_3 trends.

A long term NO_3 monitoring program is needed in Minnesota to evaluate the effectiveness of current ground water protection efforts. Future monitoring should also focus on identifying nitrate priority areas. Statewide standards are needed for collection and analysis of NO_3 data. Reliable data should be automated and maintained in a single program.

REFERENCES

(Cited References are denoted with an *)

Minnesota Department of Health

*Klaseus, Tomas G., Greg C. Buzicky and Edward Schneider. 1988. Pesticides and ground water: surveys of selected Minnesota wells. Minnesota Department of Health and Minnesota Department of Agriculture report to the Legislative Commission on Minnesota Resources. 95 pp.

*Klaseus, Tomas G. and John W. Hines. 1989. Pesticides and ground water: a survey of selected private wells in Minnesota. Minnesota Department of Health report for the U.S. E.P.A. 76 pp.

*Klaseus, Tomas G. 1990. Information presented at an April 1990, Nitrogen Fertilizer Task Force meeting held at the Minnesota Department of Agriculture.

*Klaseus, Tomas G. 1991. Personal communication in May, 1991.

*Minnesota Department of Health. 1989. Public Water Supply Data Report.

References for USGS Studies

Adolphson, D.G. 1983. Availability and chemical quality of water from surficial aquifers in southwest Minnesota. U.S. Geological Survey Water-Resources Investigations Report 83-4030.

Anderson, H.W., Jr. 1986. Hydrogeologic and water quality characteristics of crystalline-rock aquifers of archean and proterozoic age, Minnesota. WRI 86-4033.

*Anderson, H.W., Jr. 1987. Effect of agriculture on quality of water in surficial sand-plain aquifers in Douglas, Kandiyohi, Pope, and Stearns Counties, Minnesota. USGS Water Resources Investigations Report 87-4040.

Anderson, H.W., Jr. 1991. Effects of land use on quality of water in the Anoka sand-plain aquifer, Minnesota. USGS Water Resources Investigation Report in preparation.

Anderson, H.W., Jr. and Jeff Stoner. 1989. Effects of controlled agricultural practices on water quality in a Minnesota Sand Plain Aquifer. USGS Open File Report 89-267.

Cotter, R.D. and Young, H.L. 1960. Municipal water supplies on the Mesabi and Vermilion Iron Ranges Northeastern Minnesota. Ground Water Series Basic Data Release I.

Delin, G.N. 1986. Hydrogeology of confined drift aquifers near the Pomme De Terre and Chippewa Rivers, western Minnesota. USGS Water Resources Investigation Report 86-4098.

Delin, G.N. 1987. Evaluation of availability of water from drift aquifers near the Pomme De Terre and Chippewa Rivers, western Minnesota. WRI 86-4321.

Delin, G.N. 1988. Geohydrology and water quality of confined drift aquifers in the Brooten-Belgrade area, west central Minnesota.

Helgesen, J.O. 1977. Ground-water appraisal of the Pineland Sands Area, central Minnesota. WRI 77-102.

Lindholm, G.F. 1980. Ground-water appraisal of sand plains in Benton, Sherburne, Stearns, and Wright Counties, central Minnesota. WRI 80-1285.

Miller, R.T. 1982. Appraisal of the Pelican River sand-plain aquifer, Western Minnesota. OFR-82-347.

*Myette, C.F. 1982. Baseline water-quality data for sand-plain aquifers in Hubbard, Morrison, Otter Tail and Wadena Counties, Minnesota. OFR-82-909.

Myette, C.F. 1983. Appraisal of water from surficial-outwash aquifers in Todd County and parts of Cass and Morrison counties, central Minnesota. WRI 83-4156.

Myette, C.F. 1984. Ground water quality and appraisal of sand-plain aquifers in Hubbard, Morrison, Otter Tail, and Wadena counties, Minnesota. WRI 84-4080.

Myette, C.F. 1985. Hydrogeology of sand-plain aquifers in Carlton, Kanabec and Pine counties, east-central Minnesota. WRI 85-4334.

Myette, C.F. 1986. Hydrogeology of the sand-plain aquifers in Carlton, Kanabec and Pine counties, east-central Minnesota. WRI 85-4334.

Ruhl, J.F. 1989. Water Resources of the White Earth Indian Reservation, Northwestern Minnesota. USGS Water Resources Investigation Report. 89-4074. 73 pp.

Ruhl, J.F. 1989. Water Resources of Fond du Lac Reservation, East Central Minnesota. WRI 88-4114. 42 pp.

Ruhl, J.F. and D.G. Adolphson. 1986. Hydrogeologic and water quality characteristics of the Red River-Winnipeg Aquifer, northwestern Minnesota. WRI 84-4111.

Ruhl, J.F., Wolf, R.J. and Adolphson, D.G. 1982. Hydrogeologic and water-quality characteristics of the Ironton-Galesville aquifer, southeast Minnesota. WRI-83-4080.

Ruhl, J.F., Wolf, R.J. and Adolphson, D.G. 1983. Hydrogeologic and water-quality characteristics of the Prairie du Chien-Jordan aquifer, southeast Minnesota. WRI-83-4045.

Ruhl, J.F., Wolf, R.J. and Adolphson, D.G. 1983. Hydrogeologic and water-quality characteristics of the St. Peter aquifer, southeastern Minnesota. WRI-83-4200.

Ruhl, J.F. and Wolf, R.J. 1984. Hydrogeologic and water-quality characteristics of the upper carbonate aquifer, southeastern Minnesota. WRI-84-4150.

Soukup, W.G., Gillies, D.G., and Myette, C.F. 1984. Appraisal of the surficial aquifers of the Pomme De Terre and the Chippewa River Valleys, western Minnesota. WRI 84-4086.

Stark, J.R., J.P. Busch, and M.H. Deters. 1991. Hydrogeology and water quality of glacial drift aquifers in the Bemidji-Bagley area. Beltrami, Clearwater, Cass, Hubbard counties, Minnesota. WRI 89-4136. 135 pp.

Wolf, R.J. 1976. Buried aquifers in the Brooten-Belgrade and Lake Emily area, west-central Minnesota. WRI 76-100.

Wolf, R.J. 1981. Hydrogeology of the Buffalo aquifer, Clay and Wilkin counties, west-central Minnesota. WRI 81-4.

Wolf, R.J., Ruhl, J.F. and Adolphson, D.G. 1983. Hydrogeologic and water-quality characteristics of the Mount Simon-Hinckley Aquifer, southeastern Minnesota. WRI-83-4031.

Woodward, D.G., and Anderson, H.W., Jr. Hydrogeologic and water-quality characteristics of the Cretaceous aquifer, southwestern Minnesota. WRI 84-4153.

<u>References for MPCA Nonpoint Source Monitoring</u>

*Magner, J.A., D.B. Wall, and N.E. Wilson. 1990A. Factors relating to nitrate contaminated wells in an agricultural/residential area of Benton County, Minnesota. Minnesota Pollution Control Agency, Water Quality Division. April 1990. 62 pp.

*Magner, J.A., D.B. Wall, and N.E. Wilson. 1990B. Agricultural and lake influences on water quality in outwash aquifers in the school section lake area of Stearns County, Minnesota. Minnesota Pollution Control Agency, Water Quality Division. June 1990. 61 pp.

*Wall, D.B., J.A. Magner, S.A. McGuire and N.E. Wilson. 1989A. Factors influencing ground water quality near Beardsley, Minnesota. Minnesota Pollution Control Agency, Water Quality Division. November 1989. 77 pp.

*Wall, D.B., S.A. McGuire, and J.A. Magner. 1989B. Water quality monitoring and assessment in the Garvin Brook Rural Clean Water Project Area. Minnesota Pollution Control Agency, Water Quality Division. November 1989. 287 pp.

*Wall, D.B., S.A. McGuire, and J.A. Magner. 1989C. Nitrate and pesticide contamination of ground water in the Garvin Brook area of southeastern Minnesota: sources and trends. Minnesota Pollution Control Agency, Water Quality Division. December 1989. 17 pp.

*Wall, D.B. and C.P. Regan. 1991. Water quality and sensitivity of the Prairie du Chien-Jordan aquifer in western Winona County. Minnesota Pollution Control Agency, Water Quality Division. June, 1991. 63 pp.

References for MPCA Ambient Monitoring

Sabel, Gretchen. 1985. Ground water monitoring program: a compilation of analytical data collected from 1978 to 1984. Minnesota Pollution Control Agency, Division of Solid and Hazardous Waste. 33 pp.

Sabel, Gretchen and Eric Porcher. Ground water monitoring program: an appraisal of Minnesota's ground water quality, 1987. Minnesota Pollution Control Agency, Division of Solid and Hazardous Waste. 29 pp.

References for MDA Monitoring

*Hines, John W., William L. Haugan, and Deborah B. Deluca. 1990. Minnesota Department of Agriculture Water Quality Monitoring Biennial Report. Agronomy Services Division, September 21, 1990. 30 pp.

*Klaseus, Tomas G., Greg C. Buzicky and Edward Schneider. 1988. Pesticide and ground water: surveys of selected Minnesota wells. Minnesota Department of Health and Minnesota Department of Agriculture report to the Legislative Commission on Minnesota resources.

Other References

*Holtz, Bonnie. 1990. Brown, Nicollet, and Cottonwood County Health Services - ground water analysis projects. August 1990.

*Minnesota Department of Natural Resources (1991). Criteria and guidelines for assessing geologic sensitivity of ground water resources in Minnesota. Report prepared for the Legislative Commission on Minnesota resources. 122 pp.

*Porcher, Eric. 1989. Ground water contamination susceptibility in Minnesota. Minnesota Pollution Control Agency. 29 pp.

*Setterholm, D.R., A.C. Runkel, J.M. Cleland, R.G. Tipping, J.H. Mossler, R.A. Kanivetsky, and H.C. Hobbs. 1991. Geologic factors affecting the sensitivity of the Prairie du Chien-Jordan aquifer - DRAFT report to the Legislative Commission on Minnesota Resources.

*Bosch, H.M., A.B. Rosenfield, Roberta Huston, H.R. Shipman and F.L. Woodward. 1950. Methemoglobirema and Minnesota well supplies. Journal American Water Works Association. 42:2:161-170.

FATE OF NITRATE IN GROUND WATER

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Once nitrate (NO_3) enters an aquifer, a portion can be naturally removed through biological and chemical conversion to gaseous forms of nitrogen (N). This conversion is largely dependent upon certain conditions within the aquifer which often change as water flows from a recharge area to a discharge area. Nitrate concentrations can also change as water mixes with other ground water having a different NO_3 concentration. Where mixing or dilution occurs there is not a loss of N, but through dilution NO_3 concentrations are often lowered below the drinking water standard. The following section will discuss the fate of N in ground water, focusing primarily on the relationship between residence time and nitrate and on the major N loss mechanism of denitrification.

RESIDENCE TIMES AND NITRATE

Once water percolating through the soil reaches an aquifer (recharge), it will move in response to differences in pressure within the aquifer. Under most conditions, water will eventually be pumped up in a well or discharge into a surface body of water such as a river, spring, or lake. It can be a matter of minutes or thousands of years before the recharge water moves through the aquifer system(s). While water from several wells in the Mt. Simon Formation have been age dated to be over 10,000 years old, ground water in carbonate bedrock aquifers in southeastern Minnesota has moved from point of recharge to discharge in springs in less than two hours.[1] Water that is between several years old and few hundred years old is commonly withdrawn from wells in Minnesota.

Tritium (H_3) can be a useful isotope for helping to understand the age or residence time of ground water (Bradburg, 1991; Hendry, 1988; Alexander and Alexander, 1989). Tritium is a radioactive isotope with a half life of 12.43 years which is produced naturally in the atmosphere at very low levels. Atmospheric concentrations of tritium increased considerably during the mid to late 1950's due to nuclear weapons testing. Precipitation falling since 1954 has tritium levels reflective of the high atmospheric tritium. Since ground water systems are a mixture of water from different areas and times, tritium concentrations can, and often do, reflect a mixture of pre-and post-1954 precipitation. Water samples with less than about one tritium unit (TU) entered the ground prior to about 1953 (Alexander and Alexander, 1989). Alexander and Alexander (1989) reported that in 31 samples throughout Minnesota containing less than 0.8 TU, the highest NO_3-N concentration found was 0.11 mg/l. Most of these samples had less than 0.02 mg/l NO_3-N. Most wells with NO_3-N greater than 1 mg/l and all wells with NO_3-N greater than 10 mg/l contained more than 10 TU (post-1953 water).

For this report, tritium and NO_3 data were obtained for 302 ground water samples collected[2] during 1990 for several different projects in many areas throughout Minnesota (Figure C-1). The results are very similar to the set of wells reported by Alexander and Alexander (1989). Most of the pre-1953 water (<0.8

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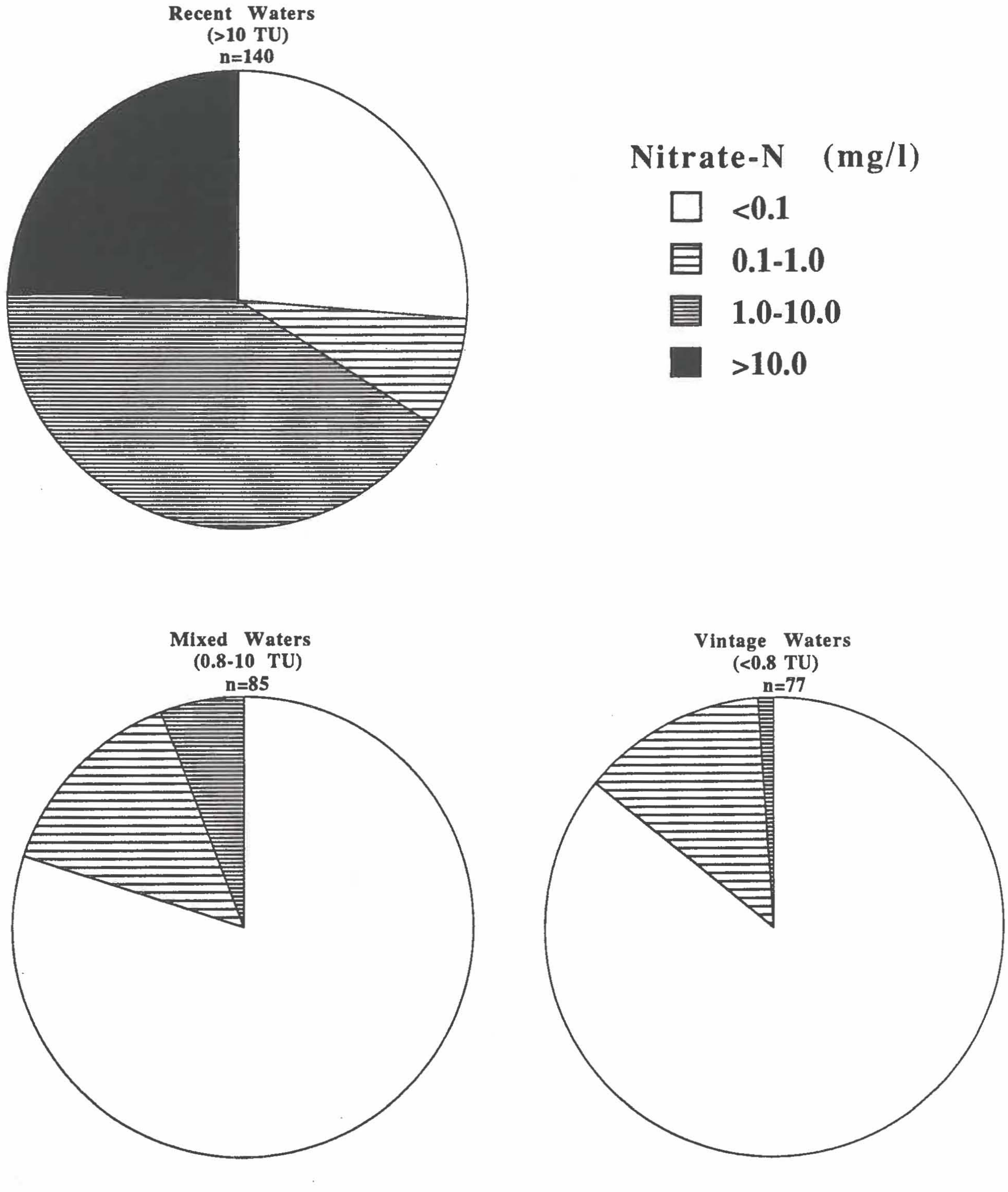
[1]Personal communication with Calvin Alexander, University of Minnesota Geology Department.

[2]These data were obtained from files kept by Calvin Alexander and Scott Alexander (University of Minnesota Geology and Geophysics Department).

Figure C-1: Nitrate/age correlation in Minnesota ground waters based on available data from samples submitted in 1990 by Calvin Alexander and Scott Alexander of the University of Minnesota - Department of Geology and Geophysics. Samples were collected by the Univ. of Minn., USGS, MPCA, MDH, Olmsted County, and Brown/Nicollet/Cottonwood Counties.



TU) had very low NO_3 with only one well having <0.8 TU and more than 1 mg/l NO_3-N (1.2 mg/l). **All 34 wells that had NO_3-N greater than 10 mg/l withdrew post-1953 water.** The residence time results suggest either 1) very little NO_3 was entering ground water before the mid-1950's, 2) NO_3 entering ground water prior to the mid-1950s was lost through denitrification, or 3) a combination of the two. The results also suggest that well water currently containing elevated NO_3 originated from water that has moved through the soil system after the mid-1950s.

DENITRIFICATION WITHIN GROUND WATER

With the exception of plant uptake of N from areas of very high water table and discharge to surface water, the only known ground water nitrogen loss mechanism is through denitrification. Denitrification, which is the reduction of NO_3 or NO_2 to gaseous N products by anaerobic bacteria, has been studied primarily in soils and waste treatment systems. Relatively few studies have examined denitrification occurring within aquifers.

Conditions required for denitrification include temperatures greater than 5 to 10°C, an anaerobic environment (indicated by low dissolved oxygen), low redox potential, denitrifying bacteria, and an organic carbon source to serve as food for the bacteria. Denitrification will result in the release of methane gas and an increase in bicarbonate and calcium in the water (Egboka, 1984; Trudell et al., 1986). In a very reducing environment (redox potential < -200 mv at pH 7) NO_3 can potentially reduce to NH_4 (Howard, 1985). Very few, if any, documented cases of NO_3 reductions to ammonium in natural aquifer settings exist.

Denitrification Within Sand And Gravel Aquifers

Many of the field studies examining denitrification within aquifers have been conducted in Canada. Perhaps the most intensively studied site is a sub-basin of Hillman Creek watershed in Southern Ontario (Gillham and Cherry, 1978; Egboka, 1984; Hendry et al., 1983). At this site, NO_3-N concentrations in an unconfined sandy aquifer are commonly between 5 and 50 mg/l in the upper six feet of the aquifer. Nitrate-N concentrations are generally less than 0.02 mg/l at depths greater than six feet. The transition zone between the high NO_3 zone and low NO_3 zone is very thin. Aquifer thickness varies from 10 to 33 feet over the 1.5 square mile sub-basin. A total of 163 observation wells were installed at 58 locations to determine the reason for the low NO_3 at depths greater than six feet below the water table.

Dissolved oxygen and NO_3 concentrations, redox potential, and methane measurements at the Hillman Creek site supported the hypothesis that denitrification is the mechanism responsible for lower NO_3 in the deeper part of the aquifer (Gillham and Cherry, 1978). In the high NO_3 zone, dissolved oxygen was about 2 mg/l, redox potential was greater than 300 mv, and methane was absent. In the low NO_3 zone, dissolved oxygen was less than 2 mg/l, redox potential was generally between 50 and 200 mv, and methane was present. In further studies, physical hydrogeologic methods of investigation, major ion analyses, environmental isotope studies and modeling showed denitrification to be responsible for the lower NO_3 in the deeper aquifer (Hendry et al., 1983; Egboka, 1984). Data from other less intensively monitored sites in Ontario also suggested that denitrification can occur in shallow ground water but is not apparent in all aquifers (Gillham and Cherry, 1978; Egboka, 1984).

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[1]Bromide will move through the aquifer similar to nitrate, but will not be lost or converted through chemical or biological processes.

Vertical NO_3 stratification was also observed in an unconfined sand aquifer near Rodney, Ontario (Trudell et al., 1986). Through an injection experiment denitrification was found to be the reason for the decreasing NO_3 in the aquifer. Nearly 15 days after high NO_3 water was injected into the aquifer, NO_3-N concentrations declined from an initial 13 mg/l to less than 0.1 mg/l. A bromide tracer injected at the same time did not show nearly the same level of decline as the NO_3. A decrease in dissolved oxygen (from > 9 mg/l to < 0.1 mg/l), an increase in bicarbonate (from < 200 mg/l to over 300 mg/l), and an increase in denitrifying organisms (from 1 to 23 per gram of soil) provided further evidence that denitrification was occurring. The measured rate of denitrification was from 0.19 to 3.12 mg/l per day. The carbon source for denitrifying bacteria was thought to be from either dissolved organic carbon (DOC) or soil organic carbon. One ground water sample had sufficient DOC for denitrification (12.4 mg/l), and aquifer soil analyses showed an organic carbon content from 0.08 to 0.16 percent by weight; which was determined to be adequate to denitrify large amounts of NO_3.

Nitrate and chloride were injected in an Iowa alluvial aquifer in an attempt to quantify denitrification (Wehmeyer, 1988). The results suggested that NO_3 was reduced in the aquifer. Core samples showed potential NO_3 reduction rates of about 1 to 9.5 mg/l per day, depending on the carbon content.

In a Massachusetts sand and gravel aquifer, NO_3 contamination resulted from more than 50 years of treated sewage disposal (Smith and Duff, 1988). While the plume from this source was 2.2 miles long, NO_3 concentrations were reported to decline below detection within 250 meters down-gradient from the contaminant source in the core of the plume. Aquifer core samples were assayed by the acetylene blockage technique and were found to display a significant potential for denitrification. Based on the laboratory results and in-situ dissolved oxygen measurement, Smith and Duff (1988) concluded that denitrification was the mechanism responsible for the drastic decrease in NO_3. Denitrifying activity was found to be carbon limited. Adelman et al. (1986) also found denitrification to be controlled largely by carbon content in aquifer materials. In the eastern sandhill regions of Nebraska, denitrification rates in soil slurry samples taken from the aquifer were found to range from 0.11 to 2.92 mg/l per day.

Denitrification was also found to be a significant nitrogen removal mechanism in a sand and gravel aquifer in Delaware (Robertson, 1980) and near Hanover, Germany (Bottcher et al., 1990).

Denitrification Within Bedrock Aquifers

A chalk limestone aquifer in England was studied using samples from 350 production wells, observation wells and springs (Howard 1985). Within the region studied, a gradual depletion of NO_3 from the recharge zone to the discharge zone was observed. The study concluded that this decrease was not likely due to denitrification, but was more likely due to mixing of waters from different origins and ages which had different chemistries and NO_3 concentrations. Howard concluded that denitrification cannot be relied upon to reduce elevated NO_3 concentrations in modern recharge waters. Only waters older than 4000 years showed any evidence of denitrification. Vogel et al. (1981) also found denitrification to have occurred in very old water (13,000 - 14,000 years old) from a sandstone aquifer in Africa.

Denitrification in a different limestone aquifer in England (Lincolnshire Limestone) was found to be a significant process resulting in NO_3 decreases of up to 33 mg/l (Wilson et al., 1990). By examining the changes in the ratio of N_2/Argon, recharge temperatures derived from noble gas measurements, and the isotope composition of the dissolved N_2 in the ground water, Wilson et al. concluded that water moving into a confined aquifer underwent significant denitrification.

The Chalk Limestone Aquifer in France was found to exhibit significant denitrification where water passes from unconfined to confined conditions (Mariotti et al., 1988). By studying nitrogen isotopes within the aquifer, it was concluded that denitrification in some areas caused NO_3-N concentrations to drop from over 10 mg/l to less than 0.2 mg/l. In other areas NO_3 concentrations declined from mixing with other low NO_3 waters.

Libra (1987) sampled 50 deep and shallow bedrock aquifer wells in eastern Iowa. Where dissolved oxygen was less than 1 mg/l, NO_3-N was less than 5 mg/l, suggesting that denitrification could be occurring. In certain Winona County Minnesota wells, Wall and Regan (1991) found sufficiently high dissolved organic carbon and sufficiently low redox and dissolved oxygen for denitrification to occur.

While very little work has been conducted in Minnesota regarding denitrification within aquifers, enough studies have been conducted in other states and countries to suggest that denitrification is probably occurring within some Minnesota aquifers.

Denitrification Near Surface Water Bodies

As ground water with elevated NO_3 approaches surface water bodies, significant denitrification can occur. In a shallow unconfined aquifer in Ontario, a high NO_3 plume of water was nearly completely attenuated within the last 6.5 feet before discharging into a river (Robertson et al., 1991). The loss of N was attributed to denitrification occurring within the organic matter enriched riverbed sediments. As high NO_3 ground water in Long Island, New York moved through sediment just before discharging into Great South Bay, NO_3 concentrations declined by 50 percent (Slater and Cavone, 1987). About 40 percent of ground water derived NO_3 in the Nottawasaga River in Canada was found to be lost through denitrification within bottom sediments (Hill, 1983). In addition to losses from denitrification, nitrogen along riparian zones may be "lost" through uptake of nitrogen by plants and trees along rivers.

IN-SITU TREATMENT OF HIGH NITRATE GROUND WATER

In-situ treatment of ground water (ground water treated directly in the aquifer) has been successful for certain contaminants by injection of nutrients and oxygen to accelerate biodegradation of waste. Since denitrifying bacteria are usually present at low numbers within an aquifer, it is possible to increase denitrification potential by introducing an organic substrate for the bacteria (Janda et al., 1988). Based on a literature review and evaluation of possible alternatives for in-situ treatment of NO_3, Mercado et al. (1988) suggested four systems of injection wells and production wells for further consideration. Two of the systems were tested and found to result, or have the potential to result, in marked decreases in NO_3. Janda et al., (1988) experimented with a system of four carbon (ethyl alcohol) injection wells and one collection well. The efficiency of NO_3 removal was found to be about 50 percent in that system. While in-situ treatment has some potential as a useful tool for NO_3 removal where the ground water resource is extremely valuable and limited, it is currently not economically feasible. Further research is needed to better develop in-situ NO_3 treatment techniques (Lowrance and Poinke, 1989; Mercado et al., 1988).

SUMMARY

Ground water that recharged more than 37 years ago rarely has NO_3-N levels greater than 0.1 mg/l. Tritium analyses suggest that elevated NO_3 levels currently found in ground water are from NO_3 loading since the mid-1950s.

By a variety of research techniques, denitrification was shown to be responsible for substantial nitrogen removal in bedrock and sand and gravel aquifers in Canada, Germany, France, England, Iowa, Massachusetts and Delaware. Not all aquifers studied showed evidence of denitrification. Denitrification in aquifers appears to be limited mostly by an organic carbon source within the aquifer. Due to organic rich sediments in the bottoms of streams, NO_3 losses can be significant as ground water discharges into streams.

Injecting a carbon source in ground water for denitrifying bacteria has shown some potential for in-situ NO_3 removal. However, this method of treatment is currently not feasible.

RECOMMENDATIONS

- More research is needed to determine denitrification rates and controlling factors in various hydrogeologic settings in Minnesota.

- Further research is needed to develop reliable, cost-effective in-situ NO_3 treatment techniques.

REFERENCES

Adelman, D.D., M.D. Jawson and J.S. Schepers. 1986. Measurement of denitrification in the eastern sandhills region of Nebraska. Proceedings of the Agricultural Impacts on Ground Water. NWWA Omaha, NE. Aug. 11-13. pp. 457-466.

Alexander, Scott C. and E. Calvin Alexander, Jr., 1989. Residence times of Minnesota ground waters. Journal of the Minnesota Academy of Science, 55:1:48-52.

Bottcher, J., O. Strebel, Susanne Voerkelius and H.L. Schmidt. 1990. Using isotope fractionation of nitrate-nitrogen and nitrate-oxygen for evaluation of microbial denitrification in a sandy aquifer. J. Hydrol., 114:413-424.

Bradbury, Kenneth R. 1991. Tritium as an indicator of ground-water age in central Wisconsin. Ground Water. 29:398-404.

Egboka, B.C.E. 1984. Nitrate contamination of shallow groundwaters in Ontario, Canada. The Science of the Total Environment. 35:53-70.

Fujikawa, James I. and M. Jim Hendry. 1990. Denitrification of clayey till.

Gillham, R.W. and J.A. Cherry. 1977/1978. Field evidence of denitrification in shallow groundwater flow systems. Water Poll. Res. Canada. 13:53-71.

Hendry, M.J. 1988. Do isotopes have a place in ground water studies? Ground Water. 26:410-415.

Hendry, M.J., R.W. Gillham and J.A. Cherry. 1983. An integrated approach to hydrogeologic investigations - a case history. J. Hydrol. 63:211-232.

Hill, A.R. 1983. Denitrification: its importance in a river draining an intensively cropped watershed. Agric. Ecosystems Environ. 10:47-62.

Howard, K.W.F. 1985. Denitrification in a major limestone aquifer. J. Hydrol. 76:265-280.

Janda, V., J. Rudovsky, J. Wanner and K. Marha. 1988. In situ denitrification of drinking water. Wat. Sci. Tech. 20:3:215-219.

Libra, Robert D. 1987. Distribution of nitrate in ground water: effects of time and denitrification. Presented at the Midwest Ground Water Conference. Madison, WI. October 28-30, 1987.

Lowrance, R.R. and H.B. Pionke. 1989. Transformation and movement of nitrate in aquifer systems. IN: Nitrogen management and ground water protection - Developments in agricultural and managed-forest ecology 21. Elvsvier, New York, NY. pp. 373-392.

Mariotti, Andre, Alain Landreau and Beatrice Simon. 1988. 15N Isotope biogeochemistry and natural denitrification process in groundwater: Application to the chalk aquifer of northern France. Geochimica et Cosmochimica Acta. 52:1869-1878.

Mercado, A., M. Libhaber and M.I.M. Soares. 1988. In situ biological groundwater denitrification: concepts and preliminary field tests. Wat. Sci. Tech. 20:3:197-209.

Robertson, Frederick. 1980. Evaluation of nitrate in the ground water in the Delaware coastal plain. Ground Water. 17:4:328-337.

Robertson, W.D., J.A. Cherry and E.A. Sudicky. 1991. Groundwater contamination from two small septic systems on sand aquifers. Ground Water. 29:1:82-92.

Slater, Jennifer M. and Douglas G. Capone. 1987. Denitrification in aquifer soil and nearshore marine sediments influenced by groundwater nitrate. Applied and Environmental Microbiology. 53:6:1292-1297.

Smith, Richard L. and John H. Duff. 1988. Denitrification in a sand and gravel aquifer. Applied and Environmental Microbiology. 54:5:1071-1078.

Trudell. M.R., R.W. Gillham and J.A. Cherry. 1986. An in-situ study of the occurrence and rate of denitrification in a shallow unconfined sand aquifer. J. Hydrol., 83:251-268.

Vogel, J.C., A.S. Talma and T.H.E. Heaton. 1981. Gaseous nitrogen as evidence for denitrification in groundwater. J. Hydrol. 50:191-200.

Wall, D.B. and C.P. Regan. 1991. Water quality and sensitivity of the Prairie du Chien-Jordan Aquifer in western Winona County. Minnesota Pollution Control Agency, Water Quality Division. June, 1991. 63 pp.

Wehmeyer, Lisa K. 1988. Denitrification and nitrate movement in the shallow alluvial aquifer of the West Des Moines River, Palo Alto County, Iowa. Geological Society of America Abstracts with Programs. 20:5:394.

Wilson, G.B., J.N. Andrews and A.H. Bath. 1990. Dissolved gas evidence for denitrification in the Lincolnshire limestone groundwaters, Eastern England. J. Hydrol. 113:51-60.

NITRITE, AMMONIUM AND T. KJELDAHL NITROGEN IN GROUND WATER

Lead Agency: Minnesota Pollution Control Agency
Contact: Dave Wall (612) 297-3847

Nitrogen can occur in water as nitrite (NO_2), nitrate (NO_3), ammonia (NH_3), ammonium (NH_4), and at intermediate oxidation states as part of organic solutes. Some other forms such as cyanide (CN) may occur in water affected by waste disposal. The chemical properties of these species of nitrogen vary greatly. While NH_4, and particularly NO_3, are fairly stable over a wide range of conditions, NO_2 and organic species are unstable in aerated water, and when found in ground water are usually associated with sewage or organic waste contamination.

Most of the ground water nitrogen data collected in the state is in the NO_3-N or NO_2+NO_3-N form. Nitrite, NH_4 and/or total kjeldahl nitrogen data have been collected through four sampling programs: USGS, MPCA ambient, MPCA NPS and U.S. Forest Service. These programs were discussed in a previous section of the report entitled "An Analysis of Nitrate Results from seven Selected Data Sets." A summary of the concentrations measured nitrite, ammonium and total kjeldahl nitrogen is presented in this chapter.

NITRITE (NO_2)

Nitrite is usually present as an intermediary nitrogen species that quickly oxidizes to NO_3, and is therefore not usually detected in ground water. While nitrate concentrations are often actually NO_2+NO_3 due to laboratory methodologies, the NO_2 species is generally considered negligible. Nitrite, when found in ground water, can contribute to methemoglobonemia and a Recommended Allowable Limit (RAL) of 1 mg/l has been set for NO_2-N.

Mean NO_2 analyses from 367 wells sampled as part of three sampling programs (MPCA ambient, MPCA NPS and U.S. Forest Service) were examined for this study (Table D-1). The overall mean and median NO_2-N concentration were 0.02 mg/l and < 0.01 mg/l, respectively. Nitrite-N exceeded 1 mg/l in only one well.

Table D-1. Nitrite-N Concentrations (mg/l) from various sampling programs.

	# Wells	Mean	Median	Maximum	Percent 0-1 mg/l	Percent 1.01-5 mg/l
MPCA Ambient	270	0.026	0.005	1.8	99.6	0.4
MPCA NPS	64	0.015	0.005	0.21	100	0
U.S. Forest Service	33	0.002	0.001	0.015	100	0

AMMONIUM (NH_4)

Ammonium is found at high concentrations in sewage and some industrial wastes. It also originates in the soil from fertilizers, manure and soil organic matter. Ammonium cations are strongly adsorbed to the soil but with time will convert to NO_3 with time in most unsaturated soil conditions. Ammonium is occasionally found in monitoring wells around some manure storage basins, septic systems and municipal and industrial waste application sites, and other point source ground water contamination areas.

There are no drinking water standards for NH_4. However, NH_4 will eventually convert to NO_3 in oxygenated waters. Most laboratories analyze for and report ammonia plus ammonium (NH_3+NH_4-N) concentrations; however, this concentration is sometimes referred to as total ammonia and often as just NH_4. Ammonia (NH_3), a gas, is usually a fairly small percentage of the NH_3+NH_4 concentration in ground water. Ammonia (NH_3) is of particular concern in surface waters where it is toxic to fishes. To distinguish ammonia from NH_3+NH_4-N concentration (or total ammonia), the ammonia (NH_3) species is usually referred to as un-ionized ammonia. The fraction of NH3+NH_4-N that is un-ionized ammonia is dependent on pH and temperature.

Actual Species	Commonly Referred to As	Comments
Ammonia plus Ammonium (NH_3+NH_4-N)	total ammonia, ammonium	Usual concentration reported from laboratory. Most of this concentration is NH_4-N.
Ammonia (NH_3-N)	Ammonia, un-ionized ammonia	Calculated from NH_3+NH_4-N, water temperature and pH. (Minor compared to NH_4, but is species that is toxic to fishes.)
Ammonium (NH_4-N)	Ammonium	Dominant species in NH_3+NH_4-N measurements.

Mean NH_3+NH_4-N concentrations from 608 wells (sampled as part of three sampling programs; USGS, MPCA-NPS, and U.S. Forest Service) were examined for this study (Table D-2). The mean and median concentrations vary by program, but are generally very low (mean < 0.5 mg/l and median < 0.1 mg/l). With the exception of one USGS well that had 60 mg/l, all NH_3+NH_4-N concentrations were less than 10 mg/l, and over 90 percent of all wells had less than 1 mg/l.

Table D-2. NH_3+NH_4-N concentrations from various sampling programs (mg/l).

	# Wells	Mean	Median	Maximum	Percent 0-1 mg/l	Percent 1.01-5 mg/l
U.S. Geological Survey	443	0.47	0.08	60	91.4	7.9
MPCA NPS	69	0.06	0.02	0.56	100	0
U.S. Forest Service	96	0.2	0.06	1.5	96.9	3.1

TOTAL KJELDAHL NITROGEN (TKN)

Total Kjeldahl Nitrogen (TKN), which is a common laboratory analysis, is often considered the sum of NH_3+NH_4-N plus organic-N. However, depending on the laboratory analysis methods, TKN may also represent a fraction of the nitrate present in the water. Organic-N, therefore, will be some fraction of TKN that is equal or less than the NH_3+NH_4-N concentration subtracted from the TKN analysis result. Mean TKN analyses from 1,067 wells taken during four sampling programs were examined for this study (Table 3). Mean and median concentrations for most programs were less than 0.5 mg/l, with a maximum concentration of 6.6 mg/l.

Table D-3. Total Kjeldahl Nitrogen concentrations from various sampling programs (mg/l).

	# Wells	Mean	Median	Maximum	Percent 0-1 mg/l	Percent 1.01-5 mg/l
U.S. Geological Survey	517	0.38	< 0.01	6.3	92.3	7.3
MPCA Ambient	405	0.71	0.33	6.6	81	18
MPCA NPS	69	0.22	0.15	2	98.6	1.4
U.S. Forest Service	76	0.44	0.28	3.7	92.1	7.9

SUMMARY

Nitrite, ammonium, and organic-N have been found at very low concentrations in the state compared to NO_3-N. Mean NO_2-N, NH_4-N and TKN concentrations measured in ground water throughout the state were 0.02, 0.38, and 0.50 mg/l, respectively. Nitrite and NH_4 were at concentrations of concern in a couple of wells.

SURFACE WATER NITROGEN

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INTRODUCTION

The primary focus of this study is on nitrogen (N) in ground water. However, surface water N concentrations are also discussed in this report since:

1. The hydrologic cycle is a continuum where surface water moves into ground water and ground water moves into surface water. The interaction between surface and ground water is important for understanding and protecting water quality in the state;

2. Streams and lakes are a source of drinking water for certain areas of the state;

3. Un-ionized ammonia is toxic to fish and excess N contributes to algal growth and macrophyte growth in certain lakes and reservoirs.

During base flow conditions (non-runoff event) the primary contributors of N to surface water are discharge from ground water (direct or via springs), agricultural tile lines, drainage ditches, and municipal and industrial wastewater treatment facilities. Rainfall or snowmelt can induce N runoff from agricultural fields, feedlots, and fertilized turf. There is also some NO_3 in rainfall that will directly enter lakes and streams. In most surface water conditions, the most stable form of N is nitrate (NO_3). However, conversion of organic N and ammonium (NH_4) to NO_3 is not an immediate process and NH_4 can persist in surface waters long enough to potentially affect aquatic life.

Lakes and streams throughout Minnesota have been sampled for N compounds for many years. This section of the report will discuss 1) monitoring results from routinely sampled streams in Minnesota, 2) how nitrogen compounds in surface waters compare between different ecoregions, and 3) measured N concentrations for the Minnesota River Assessment Project.

ROUTINELY SAMPLED STREAMS

The Minnesota Pollution Control Agency's (MPCA) Routine Water Quality Monitoring Program has been in operation since 1953, with periodic adjustments in sampling stations and analyses to adequately monitor the significant waters of the state. The Routine Water Quality Monitoring Program provides a general diagnosis of the water quality in Minnesota streams and rivers.

Beginning in October 1980 sampling frequency was reduced to nine months per year, with no sampling in November, December, and February. At this time it was determined that a better way of monitoring the state's waters would be to emphasize sampling in a different area of the state each year, while maintaining representative sampling stations state-wide. A yearly rotation of approximately 15 stations was established between southern, northeastern and northwestern areas of the state. For the water year October 1984 through September 1985, 10 stations were added in the southern part of the state; for the 1986 water year 11 stations were added in the northwestern part of the state; and for the 1987 water year 11 stations were added in the northeastern part of the state. Six stations were added to the Routine Network in October 1985 in an effort to collect background data for the ecoregions of the state.

Samples are collected on pre-set days of the month and are sometimes taken following storm or snowmelt events, but water quality analyses most often reflect baseflow conditions. Procedures used by the MPCA for collection of samples are compiled in the Quality Control Manual of the Water Quality Division. All samples were preserved and analyzed in accordance with EPA approved methods. Laboratory analyses were performed by the Minnesota Department of Health.

For this study, all MPCA routinely monitored sites that had been sampled at least 10 times between 1981 and 1990 were included for analysis. Ammonia plus ammonium and NO_2+NO_3 concentrations were reviewed from 110 sites meeting the criteria. The number of analyses obtained for each site ranged between 10 and 120 and averaged 56. The median and 90th percentile concentrations were determined for each sampling site. The median is the middle value (i.e., there is an equal number of values greater than and less than the median). The 90th percentile is representative of peak concentrations (10 percent of values are greater and 90 percent of values are lower than the 90th percentile).

Nitrate plus Nitrite

About 68 percent of all stream sites had median NO_2+NO_3-N concentrations less than 1 mg/l and only one site had a median concentration above 10 mg/l (see Table E-1). There were, however, twenty-one sites in southern Minnesota which had median NO_2+NO_3-N concentrations between 3 and 10 mg/l (Figure E-1). All sites north of the Twin Cities had medians less than 3 mg/l.

Six sites had 90th percentile NO_2+NO_3 concentrations between 10 and 15 mg/l. All sites north of the Twin Cities had 90th percentile concentrations below 3 mg/l, whereas most sites within and south of the Twin Cities had 90th percentiles above 3 mg/l (Figure E-2).

Table E-1 Number of MPCA routine stream sites with median and 90th percentile NO_2+NO_3-N concentrations falling into various ranges.

NO_2+NO_3-N mg/l	Median # sites	90th percentile # of sites
<0.1	25	5
0.1 - .99	50	56
1.0 - 2.99	13	17
3.0 - 9.99	21	23
≥10	1	9

Ammonia plus Ammonium

Median ammonia plus ammonium-N (NH_3+NH_4-N) concentrations were relatively low, with only three sites greater than 1 mg/l (see Table E-2). The few sites with median NH_3+NH_4-N greater than 0.5 mg/l were in the southern half of the state (Figure E-3). At certain peak times NH_3+NH_4-N concentrations were quite high, as evident by 90th percentiles exceeding 1 mg/l at 13 sites. The highest 90th percentile concentration was 7 mg/l. Similar to NO_2+NO_3-N, most of the higher concentration NH_3+NH_4-N sites are located in the southern part of the state (Figure E-4).

Figure E-1

Median Nitrite plus Nitrate
MPCA Routine Stream Sampling Sites (1981-1990)

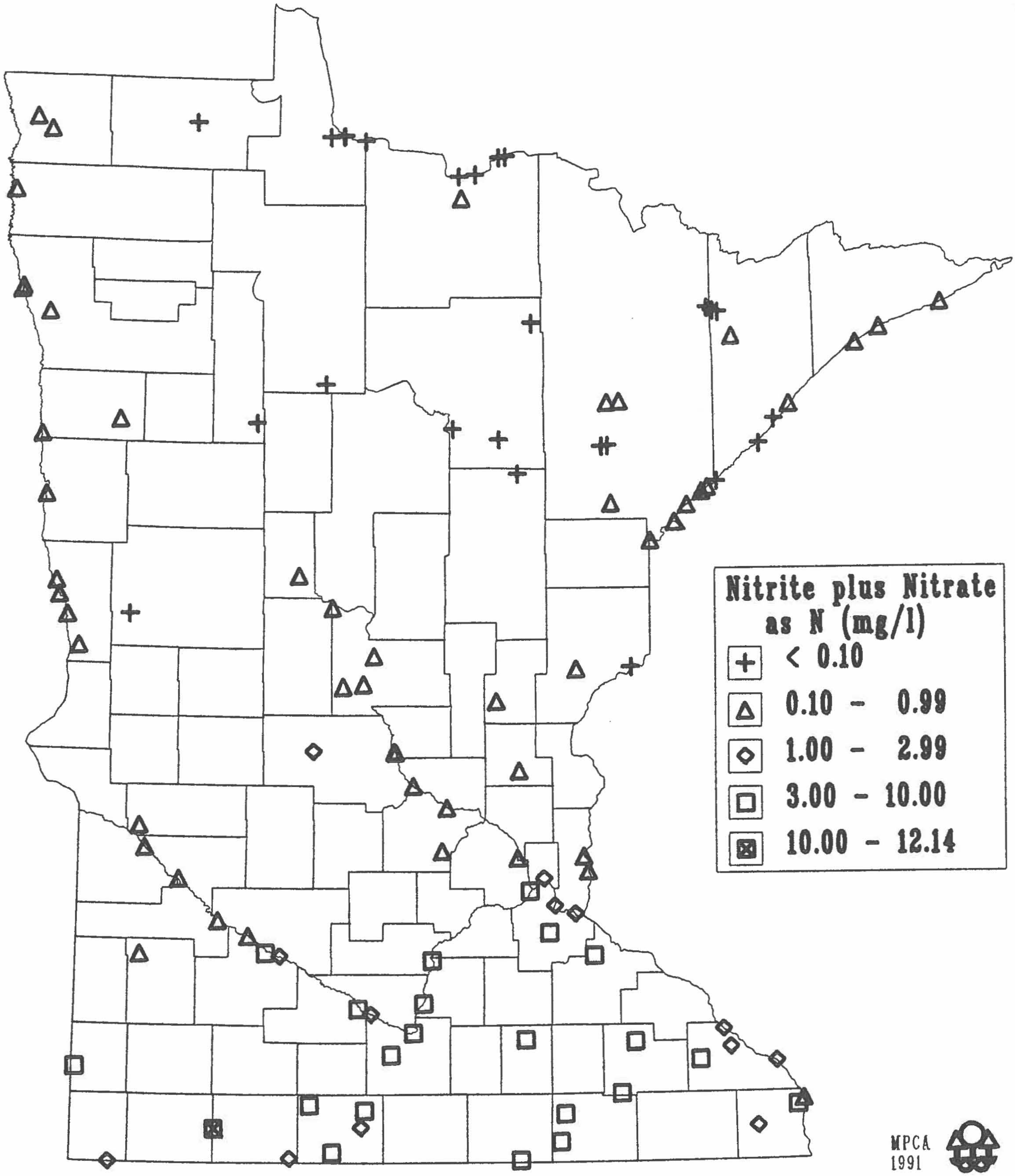


Figure E-2

90th Percentile Nitrite plus Nitrate MPCA Routine Stream Sampling Sites (1981-1990)

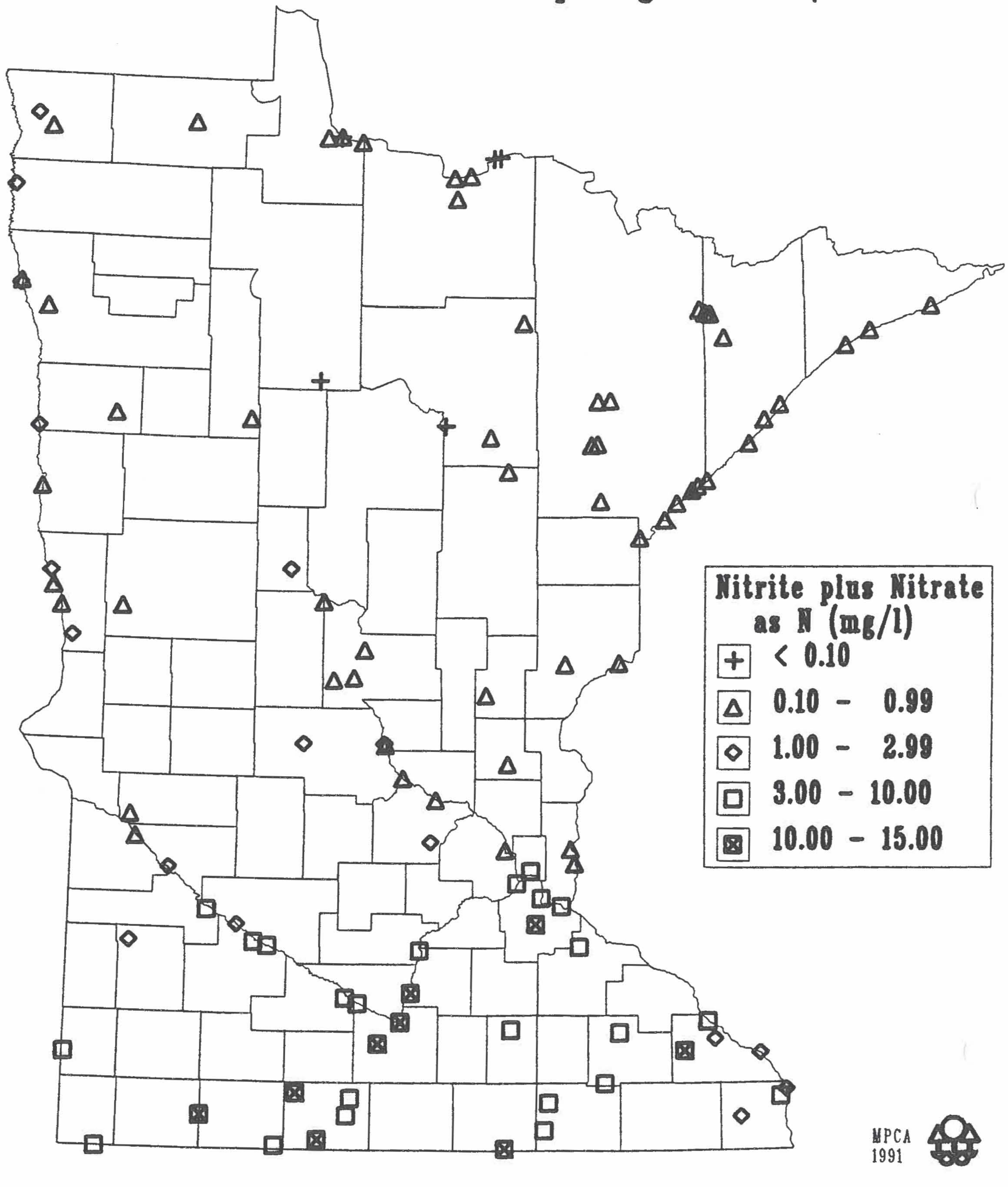


Figure E-3

Median Ammonia plus Ammonium
MPCA Routine Stream Sampling Sites (1981-1990)

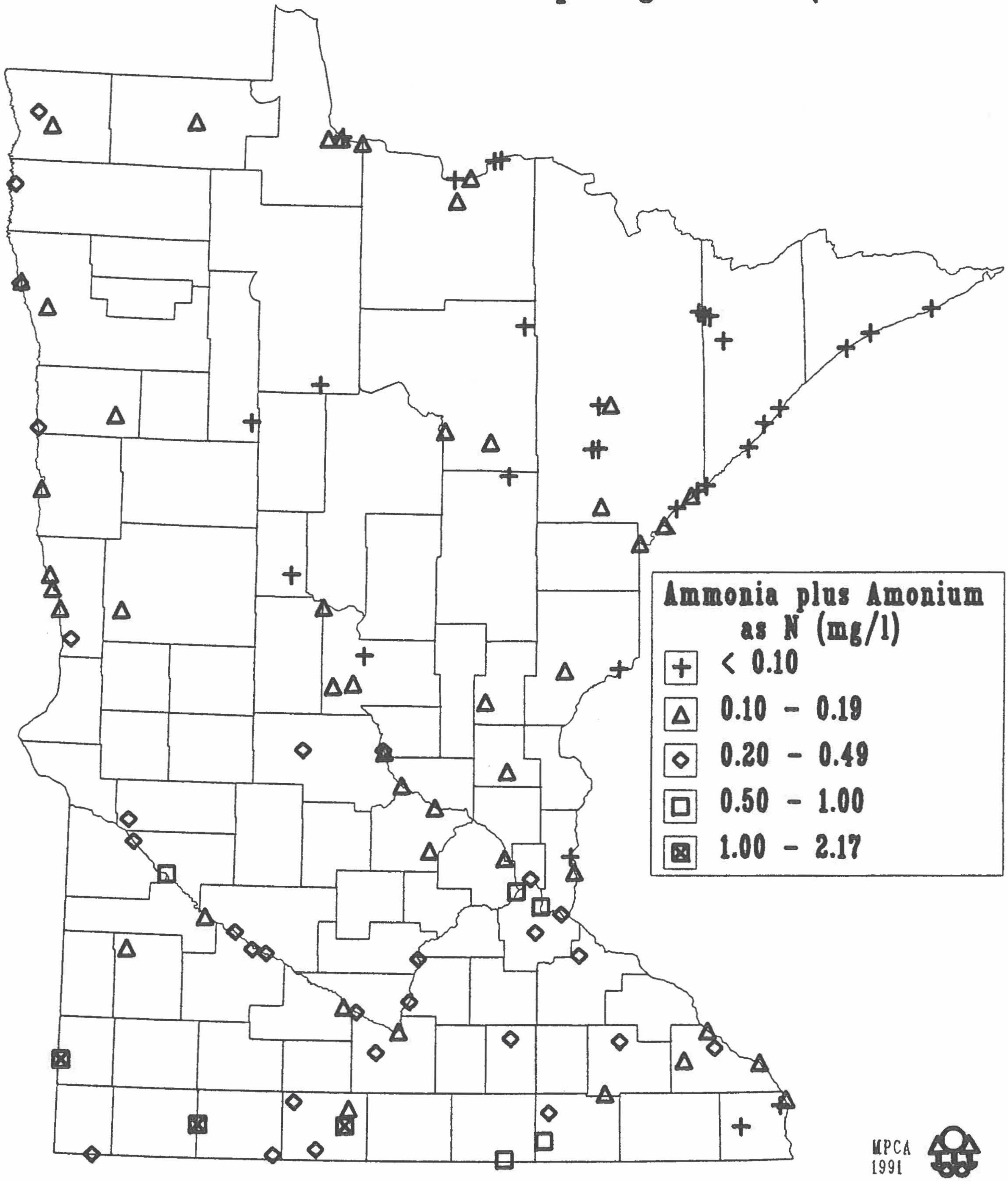


Figure E-4
90th Percentile Ammonia plus Ammonium
MPCA Routine Stream Sampling Sites (1981-1990)

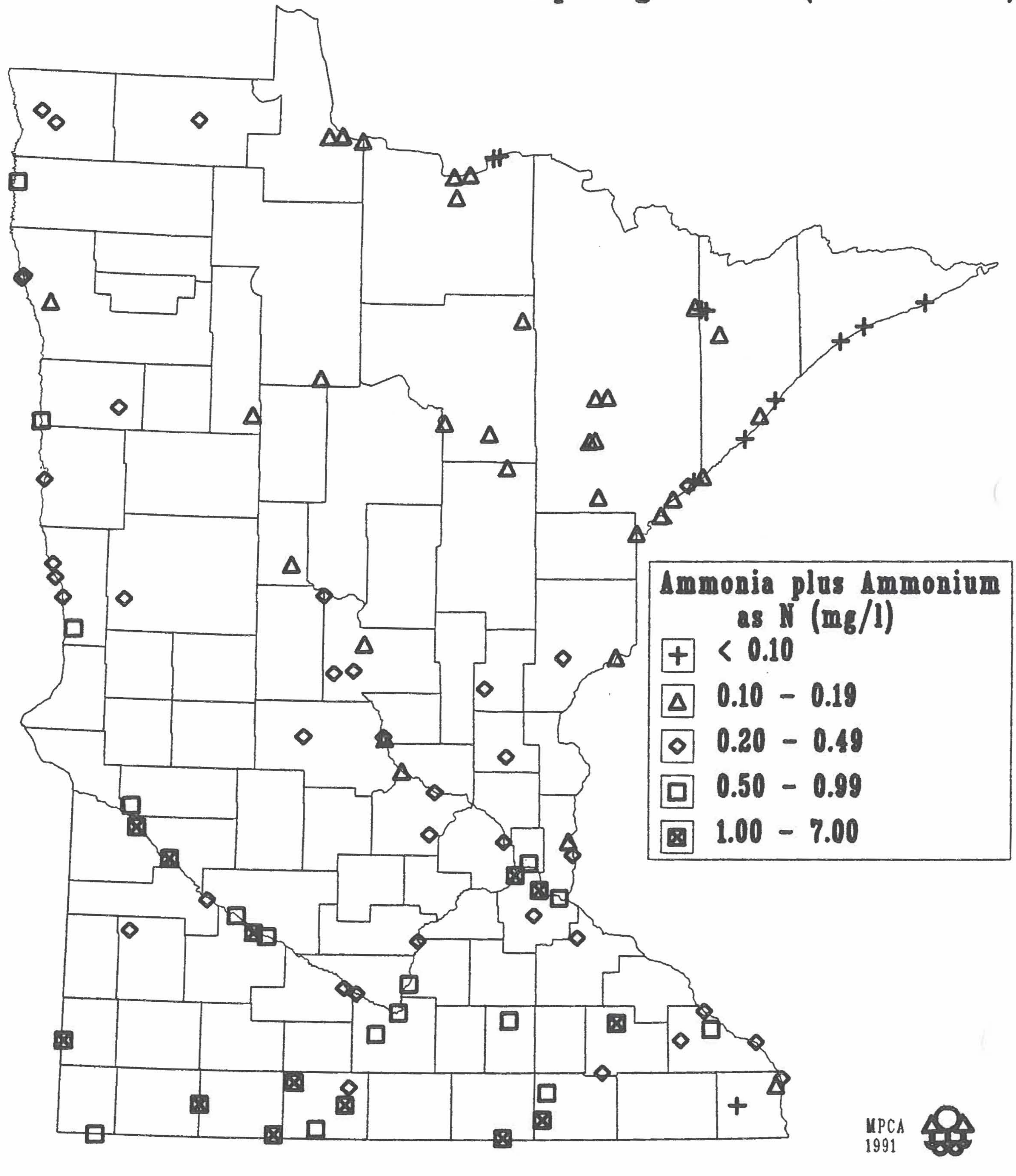


Table E-2. Number of MPCA routine stream sites with median and 90th percentile NH_3+NH_4-N concentrations falling into various ranges.

NH3 + NH4 - N mg/l	Median # sites	90th percentile # of sites
<0.1	30	11
0.1 - 0.19	44	33
0.2 - 0.49	28	28
0.5 - 1.0	5	25
>1.0	3	13

Of particular concern is the concentration of the un-ionized fraction of NH_3+NH_4 (to be referred to as un-ionized NH_3) due to toxicity to fishes. The current standard for ammonia un-ionized as N is 0.016 mg/l for Class 2A streams and 0.040 mg/l for Class 2B, 2BD and 2C streams. Un-ionized NH_3, as a percentage of NH_3+NH_4, increases directly with pH and temperature.

Median un-ionized ammonia concentrations did not exceed standards in any of the 110 routine stations. However, 90th percentile un-ionized ammonia concentrations exceeded standards at eight sites.

NITROGEN CONCENTRATIONS IN MINNESOTA ECOREGIONS

Omernik (1987) delineated 76 ecoregions in the conterminous United States by overlaying land use, land-surface form, potential natural vegetation, and soil characteristics component maps. The seven ecoregions found in Minnesota are presented in Figure E-5. Fandrei, Heiskary, and McCollor (1988) conducted a study with one of the objectives being to define the stream and lake water quality characteristics of the Minnesota ecoregions. The following discussion pertains to the results from this study.

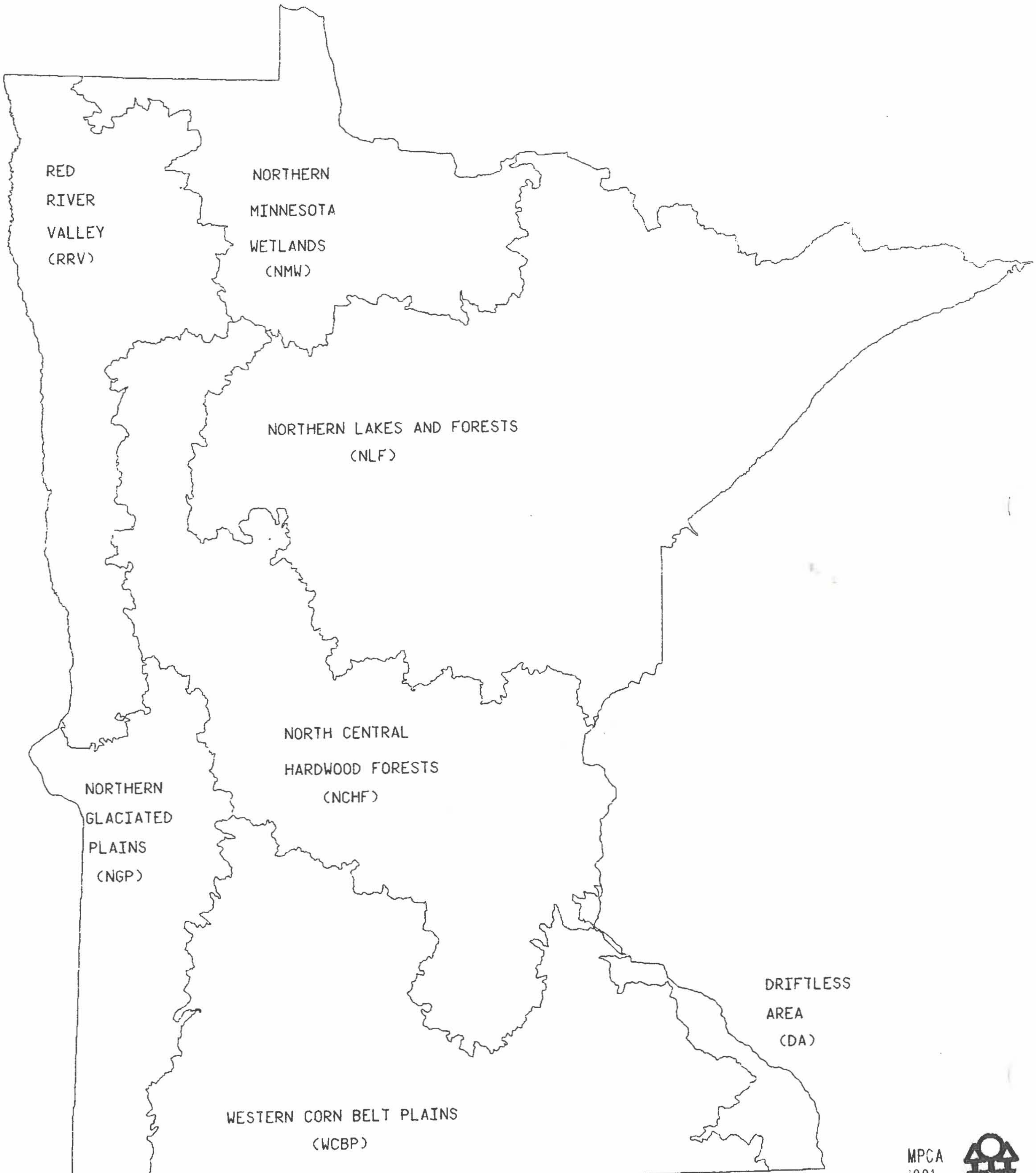
Stream Nitrogen

From the U.S. EPA computer water quality data base STORET, stream monitoring stations were selected by meeting the following criteria:

1. at least four years of data;
2. data collected monthly for at least nine months of each year; and
3. data provides a reasonable representation of the ecoregion, that is, the drainage area contributing to a monitoring station does not include large areas of more than one ecoregion.

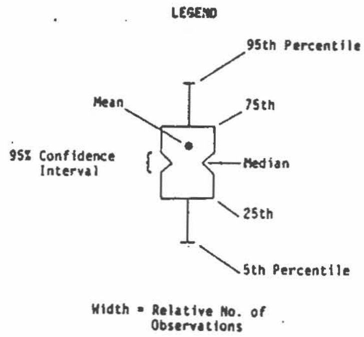
Based on these criteria, 149 stream monitoring stations were identified. Most of the stations were MPCA sampled stations, with some stations sampled by U.S. Geological Survey and the Wisconsin Department of Natural Resources. Many of these sites are the same as the MPCA routinely monitored sites previously discussed. Graphic representation of the nitrogen results are shown in Figures E-6 and E-7. The box plot graphs in Figures E-6 and E-7 provide an estimate of the number of observations, the range in values observed and the central tendency of the data for each ecoregion.

Figure E-5
Ecoregions of Minnesota



MPCA
1991

Figure E-6
Stream nitrite plus nitrate-N in Minnesota's ecoregions (Fandrei et al., 1988)



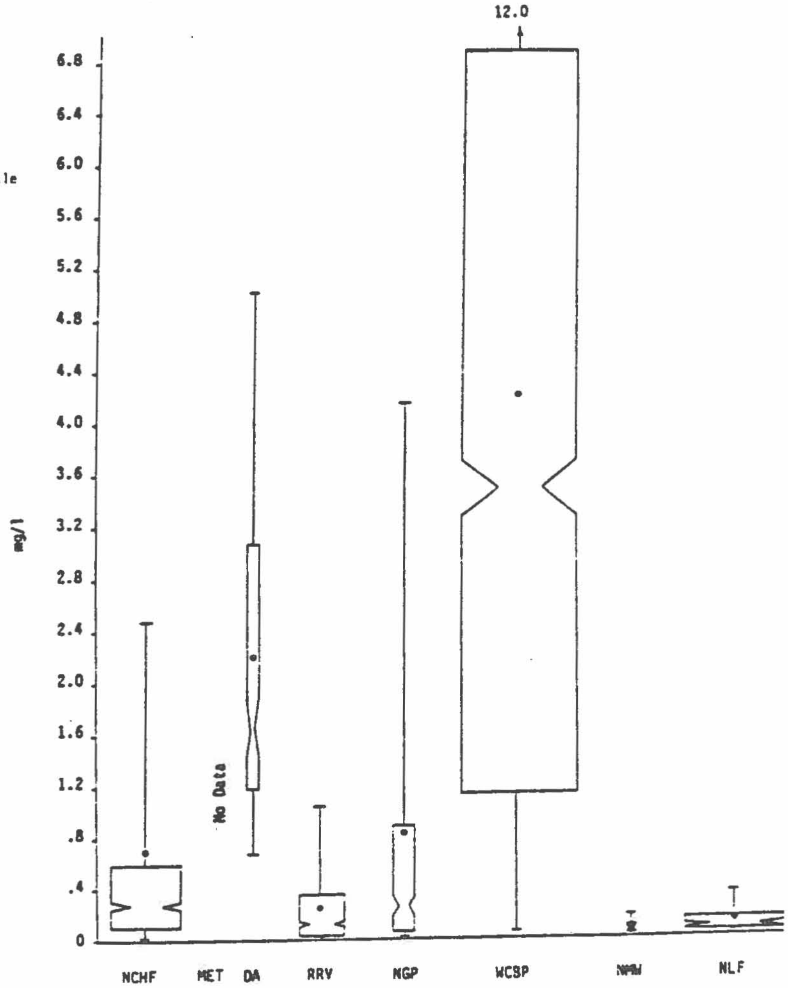
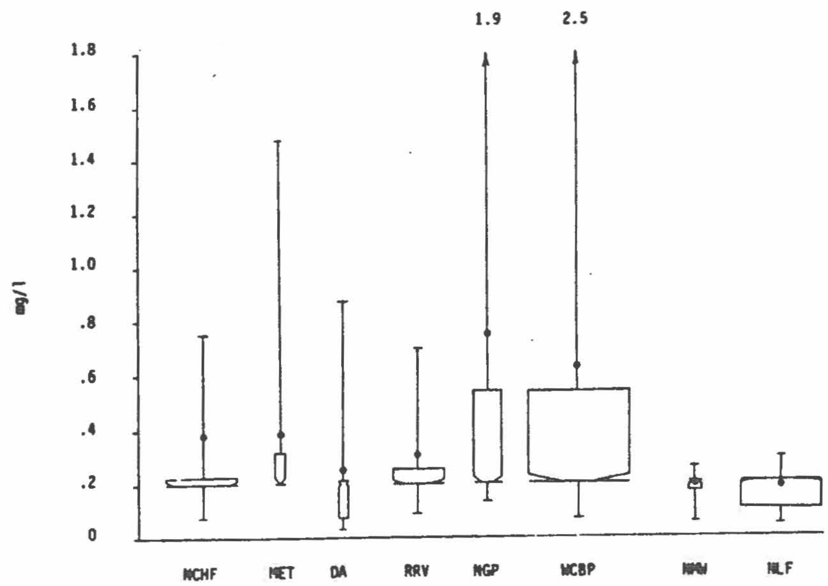


Figure E-7 Stream ammonia plus ammonium-N in Minnesota ecoregions (Fandrei et al., 1988)



Nitrite plus nitrate as N (NO_2+NO_3-N) was highest in the Western Corn Belt Plains and Driftless area ecoregions. Median concentrations were 3.5 and 1.6 mg/l in the two ecoregions, respectively. All other ecoregions had both the mean and median NO_2+NO_3-N concentration less than 1 mg/l. Ecoregions with the lowest stream NO_2+NO_3-N were the Northern Lakes and Forests and the Northern Minnesota Wetlands. For most ecoregions NO_2+NO_3-N concentrations were generally highest in the winter and spring months and lowest in the summer and fall months. Nitrate is not toxic in the aquatic environment.

Major differences in median NH_3+NH_4 concentrations between ecoregions were less apparent (Figure E-8). Mean concentrations in the Western Corn Belt Plains and Northern Glaciated Plains ecoregions were higher than in other ecoregions. The 75th percentiles in those two ecoregions were around 0.5 mg/l NH_3+NH_4-N. For most ecoregions, NH_3+NH_4 concentrations were generally highest in the winter months. See Chapter A for a discussion of the concerns of ammonia in aquatic environments.

Lake Nitrogen

Data retrieved from STORET were utilized to assess lake water quality in the four ecoregions having the majority of Minnesota lakes. All samples were taken between 1980 and 1986 and were collected primarily by the MPCA, but were also sampled by Metropolitan Council, U.S. Forest Service, and Clean Lakes Projects. Data from a total of 1204 lakes were examined. Laboratory analyses for all MPCA collected data were conducted at the Minnesota Department of Health.

The two most common N analyses for lake water are total kjeldahl nitrogen (TKN), which includes organic and NH_3+NH_4-N, and NO_2+NO_3-N. Nitrite plus nitrate-N was found only at trace levels ($<$ 0.1 mg/l) in most lakes (Figure E-8). TKN concentrations were higher than NO_2+NO_3 and varied between ecoregions (Figure E-9). The Northern Lakes and Forests ecoregion exhibited the lowest TKN, typically ranging between 0.4 and 0.6 mg/l. In the North Central Hardwood Forest lakes, TKN concentrations ranged generally between 0.6 and 1.2 mg/l. TKN concentrations were highest in the Western Corn Belt Plains and Northern Glaciated Plains, with medians of 1.9 and 2.0 mg/l, respectively.

The N to phosphorus ratio (TN:TP) is of greater importance for lakes than the total N loading. If a lake has a TN:TP ratio less than 10:1, then the lake may be nitrogen limited and, hence, the N supply may control the amount of algae produced in the lake (references cited in Fandrei, et al., 1988). TN:TP greater than 17:1 indicates a phosphorus limited lake. Ratios for different ecoregions show that lakes tend to be phosphorus limited in the Northern Lakes and Forests, North Central Hardwood Forests, and Western Corn Belt Plains ecoregions. Because of high phosphorus concentrations in lakes of the Northern Glaciated Plains ecoregion, lakes in this ecoregion are often nitrogen limited.

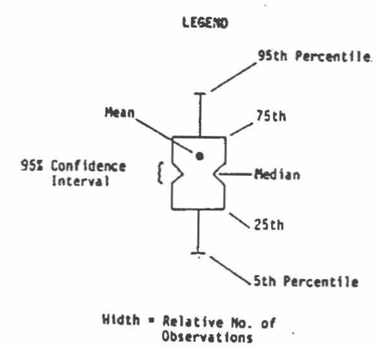


Figure E-8 Lake nitrite plus nitrate-N in four of Minnesota's Ecoregions (Fandrei et al., 1988)

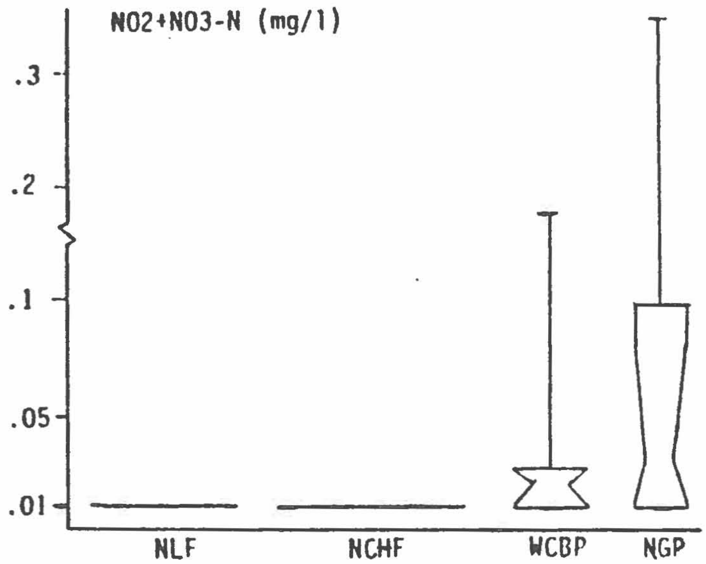
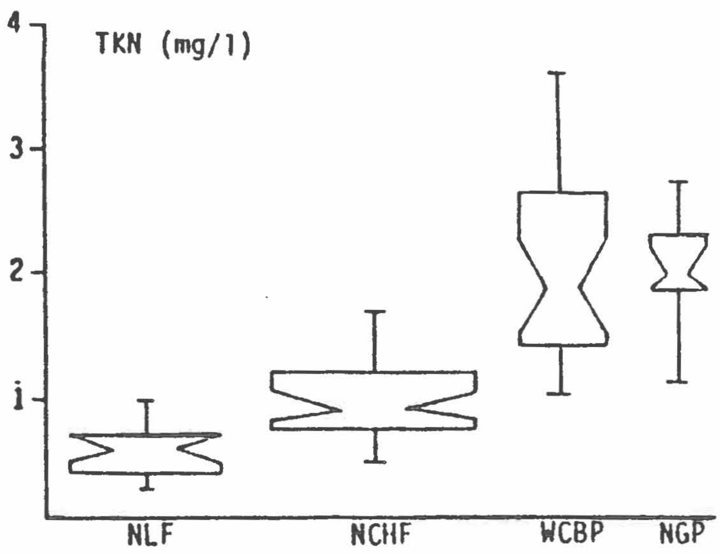


Figure E-9 Lake total kjeldahl nitrogen in four of Minnesota's ecoregions (Fandrei et al., 1988).



MINNESOTA RIVER ASSESSMENT PROJECT - INTERIM RESULTS[1]

The city of Mankato operates a series of wells for its domestic water supply. One of these wells, called the Rainey Well, is located near the confluence of the Blue Earth and Minnesota Rivers. This well has a vertical depth of approximately 60 feet and a number of horizontal laterals running under the Blue Earth River. During the spring of 1990 the NO_3-N concentrations in the Rainey well rose from below 10 mg/l to about 25 mg/l. Pumping of the Rainey well induces flow from the river into the adjacent aquifer and eventually into the well. Nitrate-N levels in the Blue Earth River at Mankato showed increases at about the same time as the Rainey well increases (Table E-3). The Mankato situation represents one example of how surface water nitrogen has affected drinking water supplies.

Table E-3. Nitrate-N concentrations in the Blue Earth River at Mankato for the spring of 1990 and 1991.

	1990	1991
MARCH	0.9 to 9.9 mg/l	5.1 to 17.0 mg/l
APRIL	0.3 to 4.7 mg/l	17.0 to 21.0 mg/l
MAY	9.2 to 25 mg/l	19.0 to 24.0 mg/l
JUNE	6.8 to 25 mg/l	not yet available

The city of Mankato is located near the center of the Minnesota River Assessment Project, a multi-agency diagnostic study of water quality in the Minnesota River watershed. As part of the MNRAP, eight first order subwatersheds averaging approximately 8,000 acres each have been intensively studied during 1991. The eight subwatersheds are in the Blue Earth Basin and have varying surface water pollution potential. Much of the land in these areas is tile-drained and planted to row crops. The monitoring has been conducted on these sites under varying conditions starting with snowmelt runoff during March of 1990. Since 1987 and 1988 were drought years in the basin, nitrogen leaching from the soil profile into tile lines in 1990 and 1991 may be greater than normal.

NO_3-N concentrations in grab and storm event samples taken from subwatersheds in the Blue Earth Basin generally ranged between 5 and 30 mg/l from March to July, 1990.

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[1]Information provided by Tim Larson, Director of the Minnesota River Assessment Project.

SUMMARY

About 80 percent of routinely monitored streams throughout Minnesota have median NO_3-N levels below 3 mg/l. At peak times of the year about eight percent of the streams have NO_3-N levels exceeding 10 mg/l. The high NO_3 streams are located in the southern half of Minnesota, primarily in the Western Corn Belt Plains and Driftless Area ecoregions. Excessive NO_3 levels were found in minor watersheds during 1990 in south central Minnesota, which has many tile lines. Elevated stream NO_3 affected a municipal well in Mankato.

Ammonia plus ammonium concentrations were less than 1 mg/l in most routinely monitored stream sites. At peak times of the year about 12 percent of sites had NH_3+NH_4-N above 1 mg/l. Un-ionized ammonia, which is toxic to fishes, exceeded standards during peak times in eight of the 110 routine monitoring stream sites.

Nitrate was found at trace levels (<0.1 mg/l) in most lakes in Minnesota. NH3+NH4-N + organic-N levels typically range between 0.4 and 2 mg/l in Minnesota lakes, being highest in the Western Corn Belt Plains and Northern Glaciated Plains. Since many lakes in the Northern Glaciated Plains tend to be nitrogen limited, the existing N may be controlling the amount of algae produced in certain lakes of this ecoregion.

CITED REFERENCE

Fandrei, Gary, Steve Heiskary and Sylvia McCollor. 1988. Descriptive characteristics of the seven ecoregions in Minnesota. Minnesota Pollution Control Agency, Water Quality Division. 137 pp.

STATEWIDE COMPARISON OF THE VARIOUS SOURCES OF AVAILABLE NITROGEN

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A comparison has been assembled to inform the reader of the relative magnitude of the various inorganic N sources. These numbers are very general estimates but serve to give the reader a broad perspective of where the major contributors originate. These numbers will vary according to specific conditions throughout the state. The application of these estimates is only appropriate as a statewide overview with the recognition that the magnitude of an individual source is not directly related to the source's impact on water quality.

From a ground water perspective, the most important concern is the amount of inorganic N within our ecosystem rather than the organic or total amounts. With time the organic-N will eventually be converted to inorganic forms. As discussed in the terminology section in Chapter A, inorganic forms (nitrate or NO_3 and ammonium or NH_4) are required in plant nutrition; these forms also pose environmental concern. With this in mind, Table F-1 and Figure F-1 were created to illustrate inorganic contributions from the major sources. Annual amounts of inorganic N available across the state are estimated between 1.4 to 1.8 million tons. Distributed equally across the entire state's land surface[1], this would be a total contribution of approximately 63 lb/A/yr.

The general utilization pathways differ according to the source. Fertilizers, manures, legumes, rainfall deposition, and municipal and industrial wastes (land applied), are surface-applied and readily available for plant uptake. Decomposition of soil organic matter also provides N within the plant root zone. Amount of N loss to ground or surface waters is dependent on numerous factors. Since most of the N from these sources is assimilated by plant life, the amount of N eventually getting into the ground water system is typically very small when compared to the net inputs.

The two remaining major sources have significantly different utilization pathways. Very little N derived from septic systems or lost through leaky manure storage facilities is available for plant life. The ultimate fate of N is either denitrification or eventually movement into the underlying aquifer.

A thorough discussion of these utilization pathways and factors affecting the sources is presented elsewhere in this report. Estimates presented here are statewide and will vary in importance in each county, soil type, agricultural region, urban region, etc.

Contributions from soil organic matter are the largest and also the most subject to interpretation. Key assumptions made here are: 1) the "average" soil organic matter across the state is 2.5%; and 2) the inorganic contribution is 10 lb/A for each percentage of organic matter. Organic fractions vary tremendously in not only amounts but also with depth. Tillage, tile drainage of wet soils, and other of man's activities will alter mineralization rates. Quantitative mineralization values have been documented on a limited number of research plots. These values are very site specific and it is not appropriate to extrapolate to a statewide basis. Despite the caveats, these gross approximations demonstrate the importance of mineralized N.

1.Minnesota's area is 84,068 square miles or 53.8 million acres. Subtracting the lake area (3.3 million acres), the land area is 50.5 million acres. Lake area data from Dave Ford, Department of Natural Resources-Division of Waters (Personal Communication).

Contributions from atmospheric depositions, although too small to be of significance in crop production, are surprisingly substantial in terms of net N loading across the entire state.

Table F-1. Sources and estimated contributions of inorganic N within Minnesota.

SOURCE	TONS	PERCENT
Soil organic matter[2]	672,544	42
Atmospheric deposition[3]	134,000	8.4
N fertilizers[4]	579,109	36
Manures[5]	98,196	6
Legumes[6]	98,980	6
Septic systems[7]	3,750	0.2
Municipal waste[8]	6,500	0.4
Industrial waste[9]	1,000	0.06
Annual Total Inputs	1,594,079	100

2. Assuming 25 lb/A/yr across the entire state and all soil types.

3. Assuming 5 lb/A/yr across the entire state. See Chapter L for assumptions and discussion.

4. See Chapter G, "Fertilizer rate effects on ground water quality and yields", for assumptions and calculations.

5. See Chapter G, "Effects of manure on ground water quality", for assumptions and calculations.

6. See Chapter G, "Effects of legumes on ground water quality", for assumptions and calculations.

7. Chapter I, "Septic tanks", for more information. Assumed 400,000 systems with each system serving 3 people. Discharge is 45 gal/day/person with an effluent concentration of 50 ml/L. An additional 200 tons/yr N is generated from the septage.

8. See Chapter J "Municipal and industrial waste" for additional information. Basic assumptions are 3,175,000 people @ 8 lb/person/year and 35% N removal through the treatment process.

9. See Chapter J, "Municipal and industrial waste" for more information. Crude estimate of industrial discharge based on MPCA file records.

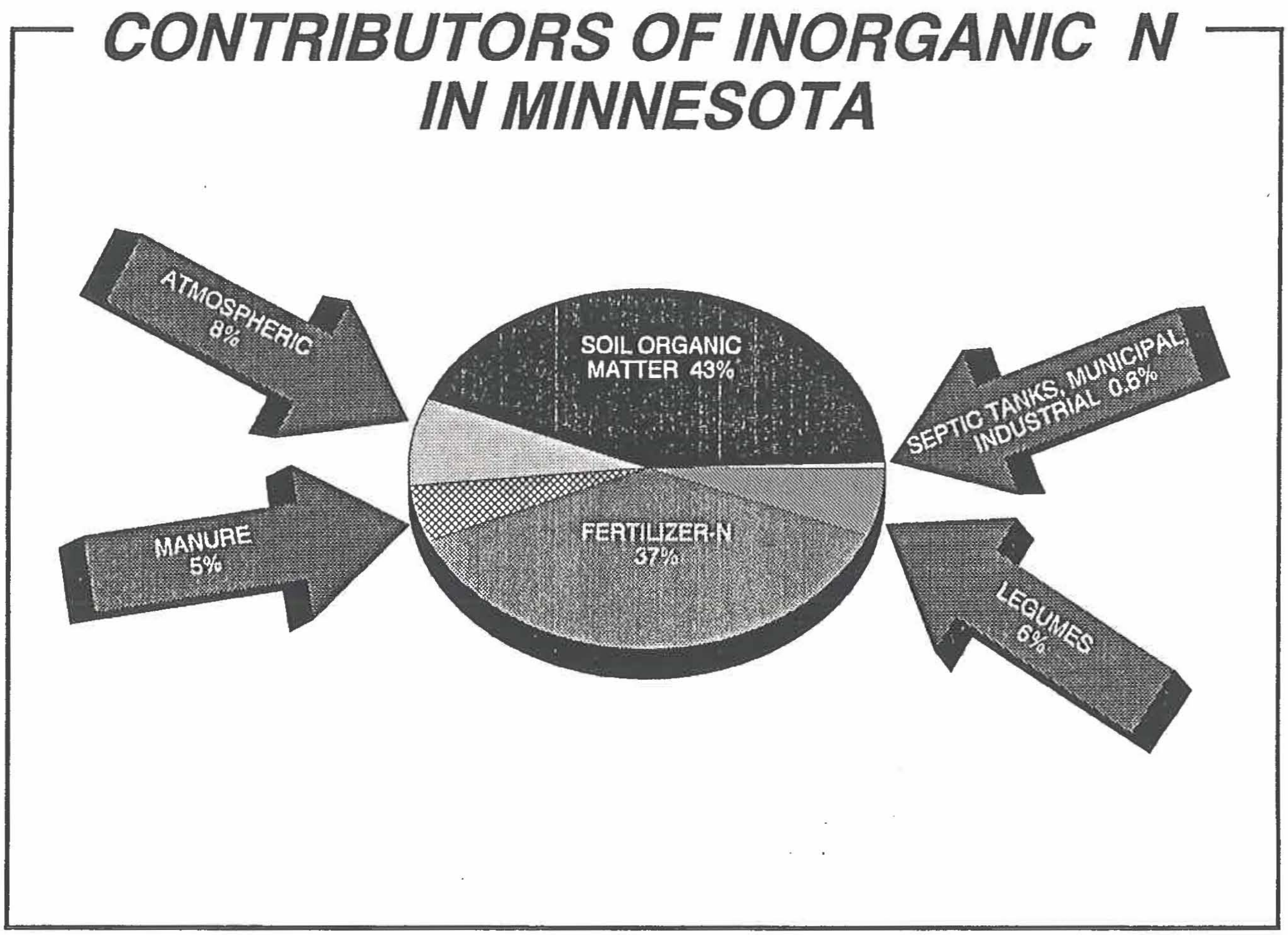

Figure F-1. Sources and estimated contributions from the most important inorganic sources of N in Minnesota. Magnitude of the source should not be equated with the likelihood of ground water contamination. See Table F-1 for calculations and assumption highlights.

CROP PRODUCTION

LEAD AGENCY: MINNESOTA DEPARTMENT OF AGRICULTURE
CONTACT: BRUCE MONTGOMERY (612) 297-7178

This section will review the effects of nitrogen (N) inputs resulting from non-point agricultural activities. Point source problems associated with feedlots and agricultural storage facilities will be discussed elsewhere within this publication. There are a number of excellent, extremely comprehensive reviews of the fate of N within agricultural systems (i.e. Stevenson, 1982; Follett, 1989). It is not the intent of this publication to review all aspects of the agricultural N cycle. The goal of this section is to review and summarize the large amount of N research as it pertains specifically to Minnesota and its diverse conditions. Very briefly, the topics which will be reviewed are: current N loading or utilization within the state and also on a county-level were sufficient data exists; tools and measurements used to assist in making N recommendations; effects of past and current agricultural practices on ground water quality; and proposed "Best Management Practices" for minimizing ground water degradation and their environmental ramifications. In most cases, the research selected will be from Minnesota and, where appropriate, its contiguous states.

INTRODUCTION TO N MANAGEMENT FOR MINNESOTA AGRICULTURAL SYSTEMS

The ultimate goal of N management is to maximize N use efficiency. Reasons for maximized utilization vary but generally are a composite of both economic and environmental issues. Ideally, effective N management should be structured in a step-wise fashion. The grower needs to first understand the limitations and potentials for the specific soil and climatic conditions, then select a yield goal that is reasonable. Importance of establishing a yield goal is absolutely critical and will be discussed later. After the goal is set, the amount of N to satisfy crop needs must be estimated. Soil, plant, and manure testing programs are all helpful tools in establishing credits and other criteria to supplement fertilizer N applications. Upon application rate selection, the next step in the management program is developing a strategy for protecting the N from any of the natural routes other than through plant uptake, thus maximizing N use efficiency. Strategy selection must be customized to fit a particular growers schedule, equipment and availability as well as the surrounding ecosystem. Following the N source discussion (fertilizers, manures, and legumes), strategies such as timing, sources, irrigation management and nitrification inhibitors will be reviewed in terms of yield and effects on ground and surface waters.

DETERMINING N FERTILIZER NEEDS

Selecting a Yield Goal

From an environmental perspective, setting a yield goal may be the most important single decision that the grower will make the entire year. Yield goal selection will ultimately dictate N fertilizer rates. Among the array of management options, N rate has the most direct impact on NO_3 contribution to ground water. Once a rate is selected, there are a large number of management factors which will aid in the efficient use by the crop but all these factors are secondary in comparison to the selection of the correct rate. Soil physical and chemical properties, such as texture, moisture holding capacity, and native soil productivity, along with climatic conditions (heat units and rainfall) and grower management effects are all vital in the yield goal decision. Peterson and Frye (1989) describe efficient N management as a step-wise process with yield goal as the number one step in both importance and the actual process.

Because of the large number of complexities involved, site specific yield goals are highly desirable (Peterson and Frye). If good records are not available or in the event that past obtained yields may not be a good indicator of optimized yields (due to limiting factors such as disease, weeds, or hidden nutrient problem), farm or county averages should be utilized.

With regards to yield goals decisions, a recent Nebraska effort is worthy of discussion. A three-phase N management program within the Central Platte Natural Resource District was recently initiated. Phases are distinguished between three categories of ground water NO_3 levels. In Phase II (irrigation wells between 12.6-20.0 mg L/1) growers are required to provide an annual N management history. Data from the 200,000 acres has been an excellent source of information on how farmers select yield goals and make the corresponding recommendations. Schepers et al. (1991b) reported that farmers missed their yield goals by an average of 9% in 1988 compared to 28% when surveyed during 1980-84. Overly optimistic yield goals in 1988 translated into an over application of 18 lb/A and explained 42% of the excess application (the average application was 43 lb/A too high based on University of Nebraska results).

Bock and Hergert (1991) reported on the long-term efficacy of applying insurance N in Midwest corn production. These authors concluded that there was little economic incentive for using insurance N and that setting yield goals based on a "running average" provided economically viable and environmentally sound agricultural production. The Nitrogen Fertilizer Task Force (MDA, 1990) defined the "running average" under several different crop rotation scenarios. Under continuous corn, yield goals should be based on the past 5-year average, excluding the worst year.

Methods for Determining N Recommendations

Efficient N management will help minimize the contribution from agriculture and the need to develop better methods for estimating fertilizer N rates has taken on a new urgency. Accurate N fertilizer recommendations, once a yield goal is established, is absolutely essential. Additional assessment on N needs are then further enhanced by soil and plant information.

N Recommendations without a Soil Nitrate Test

Currently the University of Minnesota N recommendations are based on yield goal, the previous crop type, and the organic matter amount in the soil. Minnesota, unlike its contiguous neighbors, does not endorse a statewide[1] soil NO_3 test. Rates are based on the equation:

$$N_{fertilizer} = (k * \text{Yield Goal}) - N_{credit},$$

where "k" is the amount of N needed to produce a bushel of grain. N_{credit} accounts for contributions from the sum of N from organic matter, legumes, manure, and irrigation water. Residual N would be included as a N credit in western Minnesota and currently not included in the eastern portion of the state (see Figure G-1).

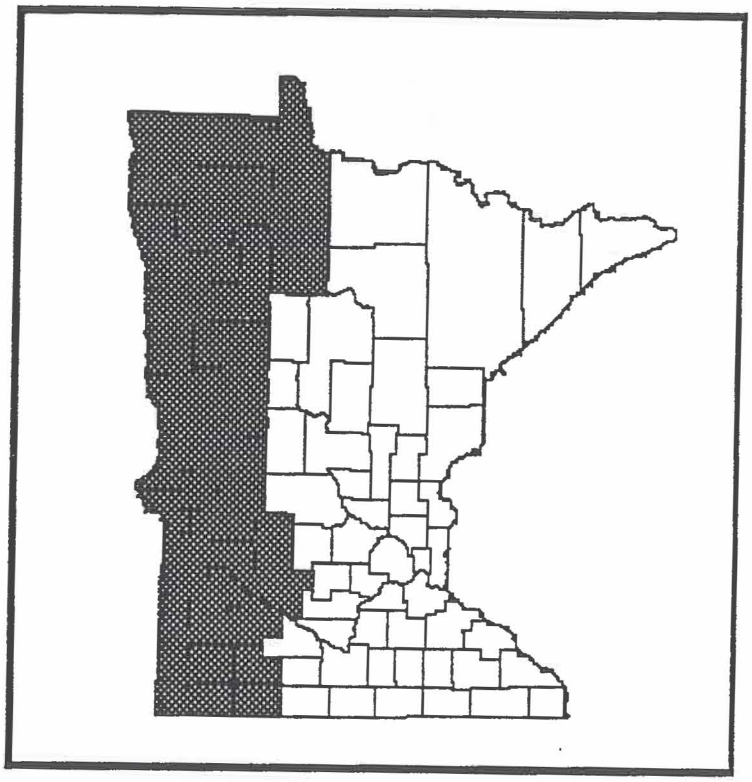

Figure G-1. Areas of Minnesota where University of Minnesota recommends the soil NO_3 test.

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1. The Minnesota Extension Service, as of 5/91, is endorsing a spring soil nitrate test for identifying fields where no additional N is needed (Rehm and Schmitt, 1991). The test is not intended for fields coming out of alfalfa. Fields testing > 175 lb A^{-1} require no additional N although a starter N may be beneficial. Fields testing < 175 lb A^{-1} will use the traditional recommendation based on previous crop, yield goal, soil organic matter, and manure credits.

Soil Nitrate Tests

The preplant soil NO_3 test is recommended in the western portion of the state and is an excellent management tool in this drier region. Soil sampling to a minimum depth of 24" is recommended. Recommendations are altered if additional information from the 24 to 48" depths are included. Western Minnesota and the Dakotas have found great success in soil NO_3 testing for many years. Because of the lower amounts of non-cropping season recharge, the likelihood of maintaining NO_3 in the root zone for the following years crop is much greater. Testing is encouraged after soil temperatures at the 6" depth have dropped below 50°F in the fall or test in the spring. Although soil samples to 24" are common, deeper NO_3 tests such as in sugar beet production are recommended when knowledge of the root zone levels is critical.

In the more humid areas, it had been assumed that residual NO_3 would be lost to leaching or denitrified prior to the next growing season. Recent Wisconsin information (Bundy and Malone, 1988) has shown that in some years, significant amounts of residual NO_3 can remain in well-drained silt loams. During years with normal off-season precipitation, about 60% of the fall residual NO_3 remained in the root zone in the following spring. Corn yields were maximized where residual soil nitrates in the top 3' exceeded 135 lb/A. As a result of this research, the University of Wisconsin has recently developed a preplant soil profile test (Bundy and Malone, 1988). Soil samples to a depth of 3' (in 12" increments) are collected in spring before planting. Recommendations are primarily designed for corn-on-corn applications. Nitrogen recommendations are based on soil organic matter, length of growing season, soil yield potential, and soil texture.

Several pre-sidedress soil NO_3 tests have emerged from the humid eastern states. Magdoff et al. (1984) described a PSNT (presidedress nitrate test) which measures NO_3 in the top 12" of soil just before sidedressing time for corn. There are two important underlying assumptions in utilizing this type of test: (1) the technique, because of the delayed soil sampling period, will integrate the numerous soil and climatic factors influencing the presence of available N at a time just prior to when the physiological need for N is critical; and (2) the amount of NO_3 at the sampling time is directly related to the N supplying capability of the soil during the entire growing season. This test has been under development in Vermont and Pennsylvania for a number of years. The test is limited to fields where no N or only starter N has been applied before soil sampling. Fox et al. (1989) summarized the PSNT after examining data from 87 experiments over a 4-year period in central and southern Pennsylvania.

The PSNT, when sampling 4 to 5 weeks after corn emergence, was found to be a good indicator of whether a response to sidedress N would be attain. Currently Pennsylvania is recommending little or no N fertilizer for sites testing above 25 mg NO_3-N/kg soil (equivalent to 100 lb/A of residual NO_3-N in the top 12"). The correlation between soil NO_3 concentrations and the soils ability to supply additional N during the remainder of the growing season is not defined enough for making actual fertilizer recommendation. Fox concluded that the test appears to be valuable for identifying non-responsive sites rather than predicting fertilizer needs but will aid in minimizing the number of growers who apply insurance N, particularly when the N credits from manures and legumes in rotations are uncertain. The test has been a much better predictor than the soil incubation procedure (mineralization index) which was previously developed (Fox et al., 1989). The PSNT, which can be viewed as an in situ incubation test, was found to work where laboratory incubation tests failed.

A Maryland study (Meisinger et al., 1991), which incorporated a variety of treatment combinations including manure and legume additions, concluded that the PSNT accurately reflected differences in N availability. Soil NO_3-N greater than 22 mg NO_3-N/kg soil (equivalent to 88 lb/A of residual NO_3-N in the top 12") were associated with relative yields of 95% or greater.

Iowa is now recommending a PSNT for the surface 12" soil layer when corn plants are 6 to 12" (Blackmer et al., 1989). This procedure is a modification of the Magdoff method. Unlike the Vermont PSNT, the Iowa PSNT is useable for fields which have received less than 125 lb/A of preplant anhydrous N. Iowa's new NO_3 test has **not** been adequately evaluated for: (1) sandy soils; (2) no-till management; (3) fields receiving greater than 50 lb N/A as injected manure; and (4) irrigated fields. The overall concept of the late spring soil test is that it offers a compromise between the need to sample as late as possible (to reflect weather effects on gains or losses of NO_3) and the need to sample early enough to be able to correct for any deficiency by the addition of N fertilizer. A linear-response-and-plateau model showed that NO_3 concentrations could explain 82% of the variability in relative yields. Model output indicated that 21 mg/L NO_3-N in the surface (top 12") would be adequate to attain maximum yield. The authors suggest that a range of 20 to 25 mg/L be considered optimal. The 1990 fertilizer recommendations were based upon NO_3-N concentrations, yield goal, and adjustment factors for soil associations. Iowa recommendations were altered for the 1991 cropping season (Lane, 1991). Iowan researchers now recommend that 10 pounds of N/acre be applied for each part per million (mg/L) the soil test value falls below the new optimum concentration (21 to 26 mg/L) and that no more than 160 lb/A sidedress applied N to any field.

The University of Minnesota recently compared their recommendations to the preplant test (0-3′ depth) and PSNT (Schmitt et al., 1990a). Recommendations were made at 15 Minnesota sites in 1989 and 15 in 1990. The preplant test worked well under Minnesota conditions. The PSNT test did about the same as Minnesota recommendations where no actual soil samples were collected. Bock et al. (1991) concluded that in the humid corn producing areas of the United States, the PSNT and the preplant NO_3 test were particularly promising for identifying nonresponsive sites following manure application and dry years, respectively.

Plant Tissue Nitrate Tests

A companion study examining corn stalk NO_3 test was completed by Fox et al. (1989) while studying the efficiency of the PSNT in Pennsylvania. The lower 4" of the corn stalks 22 to 37 days after emergence were analyzed for NO_3 concentrations. The poor correlation between stalk concentrations and either relative yield or the soil N supplying capacity was attributed to interactions from solar radiation and soil moisture.

Traditionally, the most commonly used tissue analysis tests to evaluate the N status of corn are: (1) the N concentration in the leaf opposite and below the primary ear at silking; and (2) the N concentration in the grain at harvest. Cerrato (1989) determined that either of these tests were capable of explaining only small percentages of the variability in yields. These findings suggest that these two tissue tests are not reliable indicators of N status in high fertility soils and encourage the application of excessive N (Blackmer, 1989).

Iowa researchers (Binford et al., 1990; Blackmer et al., 1991) evaluated the post-mortem stalk NO_3 test to characterize the degree of excess N during corn production. Samples of the lower portion of corn stalks were collected within two weeks of black layer formation. The relationships between relative grain yields and stalk NO_3 concentrations indicate distinct breaks between plants that were N stressed, adequate, or received excessive amounts of N. The test may provide some valuable "end of the season" type information and yield a feedback mechanism that can be used to adjust future fertilizer practices. Preliminary results indicate that excessive N rates are frequently applied (El-Hout and Blackmer, 1990).

A new technique is currently available for measuring leaf chlorophyll to assess N requirements (Turner and Jund, 1991). This is a quick, non-destructive technique which uses spectral ratio reflectance to quantify leaf color of intact leaves. This type of technology was reported as early as 1963 (Inada, 1963); yet assessments on the chlorophyll meter as a N management tool are all quite recent. Schepers et al. (1991a) and Edmisten et al. (1991) both concluded that chlorophyll meters yield data similar to traditional leaf N analysis and that this technology can be used to enhance fertilizer efficiency by improving synchronization between N availability and crop needs. Correlations between meter readings and yields are improved significantly when normalized for a specific corn hybrid (Masterson et al., 1991) and when typical plant populations are selected (Blackmer et al., 1991).

New Technology for Measuring Soil N Status

There are several obvious limitations with any of the pre-sidedress NO_3 tests. The short period of time between soil sample collection, analysis, and the sidedress application can be a major problem since it is physically difficult to get anhydrous ammonia properly applied after the V10 growth stage (Ritchie and Hanway, 1984). Several new products are now available to aid the grower in determining their own NO_3 status. This technology is extremely compatible with the Iowa and Vermont PSNT. N-Trak Soil Test Kit[2] allows the user to collect soil samples, dry them overnight and then determine the fertility status. This kit, endorsed by the Iowa State Extension Service, could reduce analysis time if the growers' alternative was to mail the soils to a testing lab. There are some concerns regarding the grower's ability to perform the analytical procedure with the required accuracy. Horiba[3] manufactures a NO_3 electrode meter which is relatively inexpensive and has similar characteristics to electrode equipment found in many soil testing labs. Like the N-Trak system, growers can run their own samples with very short turnaround time. These "quick test" methods may enhance farmer awareness concerning N management and soil testing concepts. The general ability of growers performing any of these tests is unknown and these tools may be most useful in the hands of a trained consultant or extension agent.

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2. N-Trak is a Hach Company trademark. Product names are included here for the convenience of the reader and do not constitute endorsement of such products by the authors or their respective employers.

3. Specific name is the Cardy nitrate meter. Available through Spectrum Technologies, Plainfield, Il.

Variable-rate fertilizer application technology is now available on a commercial scale. Computer driven fertilizer spreaders are fed a combination of field specific digitized soil maps as well as a grid based set of soil testing information. Rates and formulations are altered "on the go". The systems are saving between $3 to $15 per acre in fertilizer costs in Illinois (Pocock, 1991). Environmental benefits have not yet been determined but speculated to be of significant importance. The feasibility of this approach is currently being evaluated on some diverse soil complexes within the Anoka Sand Plains[4] and other Minnesota sites. Approximately 30 commercial units are currently operating in the state serving over 100,000 acres in 1991[5]. Precise location referencing is expected to become very exact as global positioning systems are adopted by the industry.

Another "on the go" concept is called the Soil Doctor[6]. This system takes soil nitrate nitrogen readings by nitrate sensors mounted on the cultivator, then adjusts liquid N rates while traveling across a field (Houtsma, 1990). Limited research results can be found in Murdock (1991).

SUMMARY OF YIELD GOAL SELECTION AND DETERMINATION OF N NEEDS

Selection of yield goal and the subsequent N application rate has a profound effect on ground water quality. Preliminary research has indicated that growers tend to set unrealistic goals, commonly missing them by 10 to 30%, and as a result application rates are much higher than needed to maximize yields.

The Nitrogen Fertilizer Task Force is strongly recommending the 'running average' concept for yield goal selection. This approach will provide a sound basis for a field-specific recommendation that is environmentally sensitive and agronomically sound. Yield goals should be based on the past 5-year average, excluding the worst year. See "Recommendations of the Nitrogen Fertilizer Task Force" for details.

Tools such as soil testing, and to a lesser degree plant tissue sampling, play a valuable role in determining application rates once a yield goal is established. Minnesota is in somewhat of a transition period in terms of a statewide soil test. Minnesota Agricultural Experiment Station is currently assessing testing programs of some of the surrounding states and also looking at other new parameters for estimating N availability.

Soil testing in the appropriate portions of the state is highly recommended. See "Recommendations of the Nitrogen Fertilizer Task Force" for details.

Technology for farming soil types rather than fields is quickly becoming a reality and may play an important role in the future of agriculture. Currently the value of such high level management maybe overshadowed by gross over-application due to unrealistic yield goals and not taking proper credit for other N sources.

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4. Personal communication with Dr. Gary Malzer and Pierre Robert, University of Minnesota, Department of Soil Science.

5. Personal communication with Dean Fairchild, Agri-Information Services, White Bear Lake, MN.

6. Trademark of Crop Technology, Inc., Houston, Texas.

METHODOLOGIES FOR ASSESSING NITRATE LEACHING LOSSES

Ideally, the effects of various agricultural activities should be measured by the resource that we are ultimately trying to protect--the ground water itself. But serious complications accumulate when attempting to relate current management strategies to ground water quality. Changes in water quality attributed to any nonpoint source in certain hydrogeologic systems may take years, even decades, to occur. The dynamic nature of the N chemistry in soil also confounds the problem and this type of assessment would integrate all the various transformations within the N cycle. Travel times and other complexities with unsaturated and saturated flow commonly prohibit directly relating ground water NO_3 concentrations and losses to specific management practices. Ground water quality monitoring should be viewed as a reasonable integration of numerous management practices over a period of time (Schepers et al., 1991b). Accordingly, ground water monitoring must be viewed as a portion of the NO_3 assessment process. The precautions associated with aquifer monitoring, due to the spatial and temporal nature of NO_3, were discussed earlier (Chapter B).

Problems and advantages of statewide and other large scale ground water monitoring programs were also previously discussed. Researchers have successfully utilized isolated watersheds for investigating the amount of NO_3 loading entering ground water systems (Hallberg, 1989; Pionke and Urban, 1985; Schuman et al., 1975; Jackson et al., 1979; Burwell et al., 1976). Studies using this technique are limited due to difficulties in locating the proper conditions and high costs associated with the required monitoring equipment. If the drainage can be monitored and measured for a defined area, the entire watershed can be viewed as a macrolysimeter. The likelihood of locating an entire watershed under one management practice is minimal. This technique is one of the most realistic methods for quantifying NO_3 loading since it accounts for all possible N transformations. Time lag factors are generally estimated with tracer materials such as chloride or bromine which have mobility characteristics similar to NO_3.

More frequently, researchers will make direct measurements or infer leaching losses directly below the rooting zone to avoid complications associated with ground water monitoring. Methods for assessing leaching losses vary tremendously and knowledge of the methods is essential when comparing dissimilar methods and results. Quantifying NO_3 leaching losses is extremely difficult even under research conditions. Actual measurement of these losses requires both the concentration of the percolation and the amount of percolation taking place. Again, these types of measurements vary tremendously, both spatially and temporally.

Lysimeters have been used successfully in a number of Minnesota research studies. The term "lysimeters" generally refers to isolated blocks of either disturbed or undisturbed soil. These units give researchers the opportunity to accurately measure the amount of drainage through the soil profile as well as the concentration of the leachate. Lysimeter sizes vary from 55 gallon drums (Brown et al., 1985) to large tile-drained plots isolated by curtained walls and a restrictive natural barrier to serve as the floor (Gast et al., 1978). Lysimeters, particularly those which are self-contained with undisturbed soil profiles, are difficult to construct and extremely costly to build and maintain

(Brown et al., 1974). Alternatives to lysimeters which still offer direct leachate measurements have been devised and reported elsewhere (Duke and Haise, 1973; Montgomery et al., 1987). Time lag factors are significantly reduced although still pose some problems in data interpretation. Currently lysimeter studies are ongoing in Minnesota at the Rosholt Irrigation Farm (Westport), the Southern Experiment Station (Waseca), and the Southwest Experiment Station (Lamberton).

Researchers often resort to utilizing indirect methodologies for estimating NO_3 leaching losses. Nitrogen balance techniques are commonly utilized. Additive errors associated with technique are very large and leaching losses are only gross approximations. Nitrogen balance work is frequently enhanced with "tagged N" (enriched or depleted ^{15}N). Although very effective, extremely high costs of the fertilizer material and analysis costs limits this technique to plot-size studies.

In situ soil solution extractors such as tension plates (Cole, 1958) and suction cups (Wagner, 1962) have been incorporated in many experiments to provide NO_3 concentrations of the leachate. Percolation estimates are made based on independent drainage or flux calculations. High variability and the chemical relationship of the extracted solution to actual leachate has been seriously questioned (Hansen and Harris, 1975; Broadbent and Carlton, 1980; Barbee and Brown, 1986).

It is not the intent of this report to review the various strategies for assessing NO_3 leaching losses but it is imperative that the reader understands that each technique has its own distinct advantages and disadvantages. The method selected will define the result, therefore where possible, studies using direct measurements will be utilized. Errors associated with many of the indirect methods are much too large to differentiate the seemingly small but critical changes in the leaching loads resulting from management alterations.

EFFECTS OF MANURE ON GROUND WATER QUALITY

Introduction

Proper manure utilization is essential from both economic and environmental perspectives. Manure contributions to the overall N balance in Minnesota is substantial. Total N (both inorganic and organic forms) from manure was earlier estimated to be equivalent to 25 to 30% of the commercial fertilizer value (MDA, 1990). Based upon a simplistic animal inventory source[7], the total N generated in 1987 was 196,400 tons. In Chapter H (Feedlots), a more complete animal inventory based upon 1990 census numbers was assembled and total N produced was estimated at 270,000 tons. The more recent inventory accounted for calves, replacement cattle and multiple batches of poultry.

For purposes within this chapter, the 1987 inventory will be used. Based upon the assumption given below, manure provides approximately 98,200 tons of plant available N. Manure accounted for 13% of the "plant available N" based on the total contributions from fertilizer, legume and manure credits (See Figure G-2). Dairy cows beef cattle, hogs, and turkeys[8] supply the majority of the manure and contribute 38, 14, 28, and 15%, respectively, of the total production. Chickens (3%), sheep (1%), and horses (1%) make up the remainder of the domestic contributions. Production from wildlife has not been estimated.

Estimates of county level manure loading are presented in Figure G-3. The following assumptions were made:

1) Manure estimates were based on domestic animal numbers from census data provided from the 1987 Census of Agriculture.

2) Nutrient analysis and manure amounts generated per animal were supplied from "Livestock Waste Facilities Handbook", 1985.

3) Assumed that 50% of the N was lost to the atmosphere during the storage, handling, and distribution processes.

4) Density calculations were based on equal distribution across all acres classified as "cropland" in the 1987 Census of Agriculture. This would include acreage such as pasture and Conservation Reserve Program (CRP) lands. This would not include areas in forests, lakes, and urban acreage.

5) County specific turkey populations are not available and therefore Figure G-3 does not reflect turkey contributions.

It is extremely important to note that the two high input regions generally coincide with the most sensitive hydrogeologic sections of the state: the sand-plains and the karst regions.

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7. Animal inventory numbers were from the 1987 Census of Agriculture (AC87-A-23) Department of Commerce. This data was available in spreadsheet format from EPA (1990).

8. Minnesota turkey production numbers were supplied by Dr. Mike Schmitt, Minnesota Extension Service (personal communication).

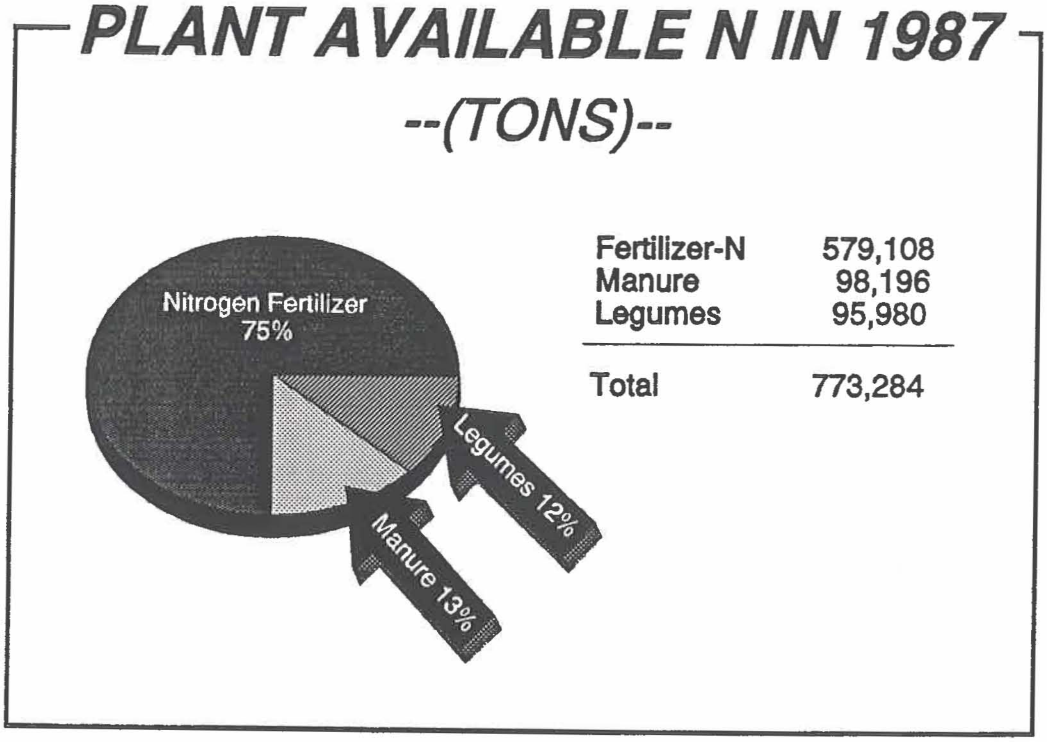

Figure G-2. Estimates of "plant available" N amounts in Minnesota during 1987 from fertilizer, manure, and legumes. See text for details on the assumptions used in the figure assembly.

Annual manure production in Minnesota based only on N is valued at over \$39,000,000[9]. This value would be extremely conservative since manure supplies many other valuable nutrients as well as improvement to soil structure, tilth, and moisture holding capacity. Yet manure continues to be viewed as a liability and is commonly mishandled (Legg et al., 1989). Even as late as the 1970's, researchers were investigating the effects of extremely high manure inputs as a method of disposal. Effects were generally considered maximized when yield reductions were observed. Evans et al. (1977) reviewed a number of these studies as well as conducted some high input/long term research. Degradation of surface or ground water were generally not of concern, although deep soil sampling did substantiate NO_3 movement beneath the root zone.

Manure additions seriously complicate attempts to make accurate N budgets. Currently, one of the most common problems in manure management is that many growers fail to take the proper N credit and, as a result, excessive N is applied. Schepers and Fox (1989) summarized some of the most salient uncertainties in estimating manure inputs: vague estimates of the amount applied; extreme N variability of the manure; variable gaseous losses; and the

9. Based upon total nitrogen production of 196,400 tons/year@ \$0.20 per pound and assuming that 50% is lost to the atmosphere during the storage and distribution processes.

uncertainty of the proportion available for plant uptake. Legg et al. (1989) categorized many farmers as "risk averse" meaning that these individuals would prefer a strategy with lower expected value and lower risk to one with a higher expected value and higher risk. Farmers perceive fertilizer-N as a risk-reducing input and because of the uncertainties or disbelief of the nutrient value of the manure or legume, risk averse farmers would then tend to over-apply to maximize profits.

Kaap (1986) summarized N inputs within the Big Springs in northeast Iowa and concluded that N in excess of 80 lb/A/year was being applied and the major reason for the discrepancy was the fact that farmers were not taking adequate credits for manure and alfalfa. Sixty percent of the farmers did not take any credits for these sources of N. Nitrogen inputs were estimated to exceed crop

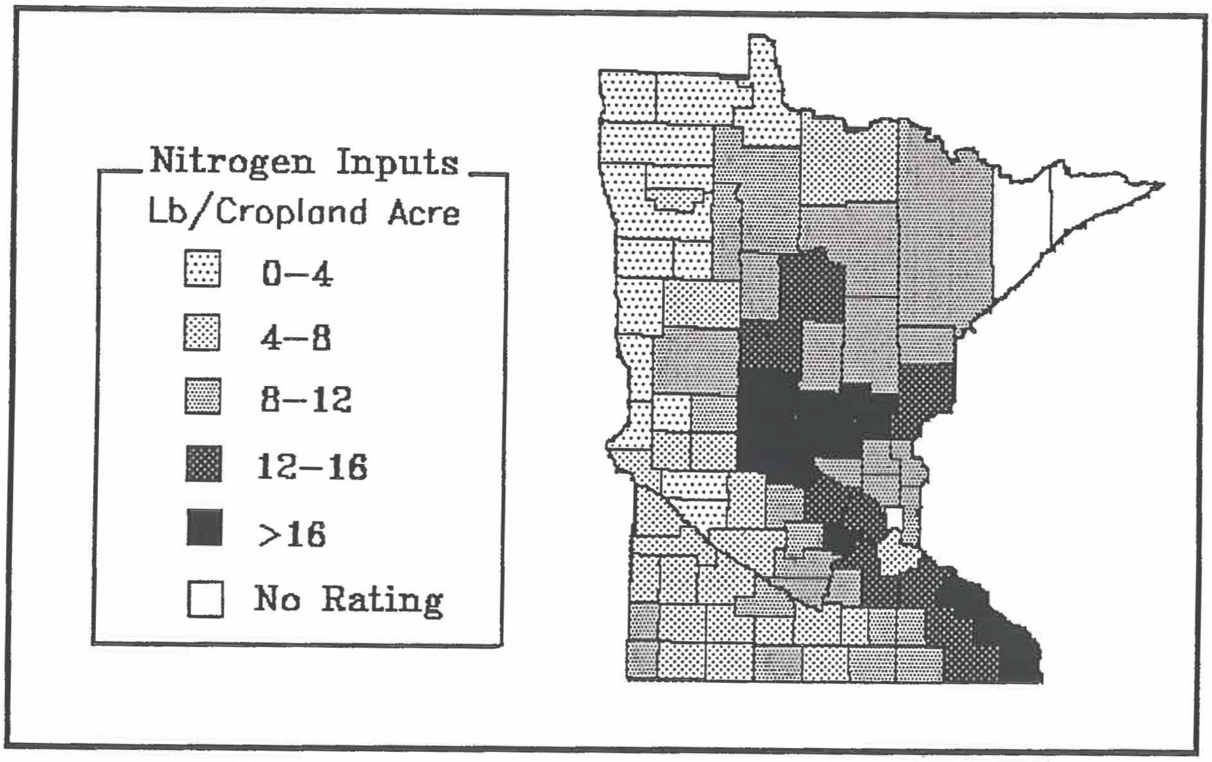

Figure G-3. Estimate of plant available N contributions from manure based on the following assumptions: 1987 Census of Agriculture animal populations; assumed 50% N loss due to storage, handling, and distribution; and densities are based on equal distribution across all cropland acres. See text for more details. This map does not include turkey contributions.

needs in Fillmore County and southeast Minnesota by greater than 50 lb/corn acre (Legg et al., 1989). Amounts in excess varied by the type of producer. Amounts ranged from no excess under continuous corn with only commercial fertilizer to 130 lb/acre under dairy operations. In a related study, Legg et al. (1990) surveyed 36 farms and determined that manure accounted for 30% of the "applied"

N and excess amounts ranged from 40 to 65%. Producers were in excellent agreement for crediting alfalfa and soybeans but grossly over applied manure (See Figure G-4).

A misconception commonly shared by the general public is that because manure (and legumes) are organic, any environmental concerns can be dismissed. This misconception is illustrated by a recent survey of a majority of dairy farmers in Beltrami County (Anonymous, 1990). Road salts were ranked at a higher level of environmental concern than manure. On the other end of the spectrum, some people fear consuming food products grown in manured systems. In either case, it is very important to note that most agronomic crops can only utilize inorganic forms of N (NO_3 and NH_4). Legumes are the exception in that they can utilize atmospheric N. The advantage of organic sources of N is that the

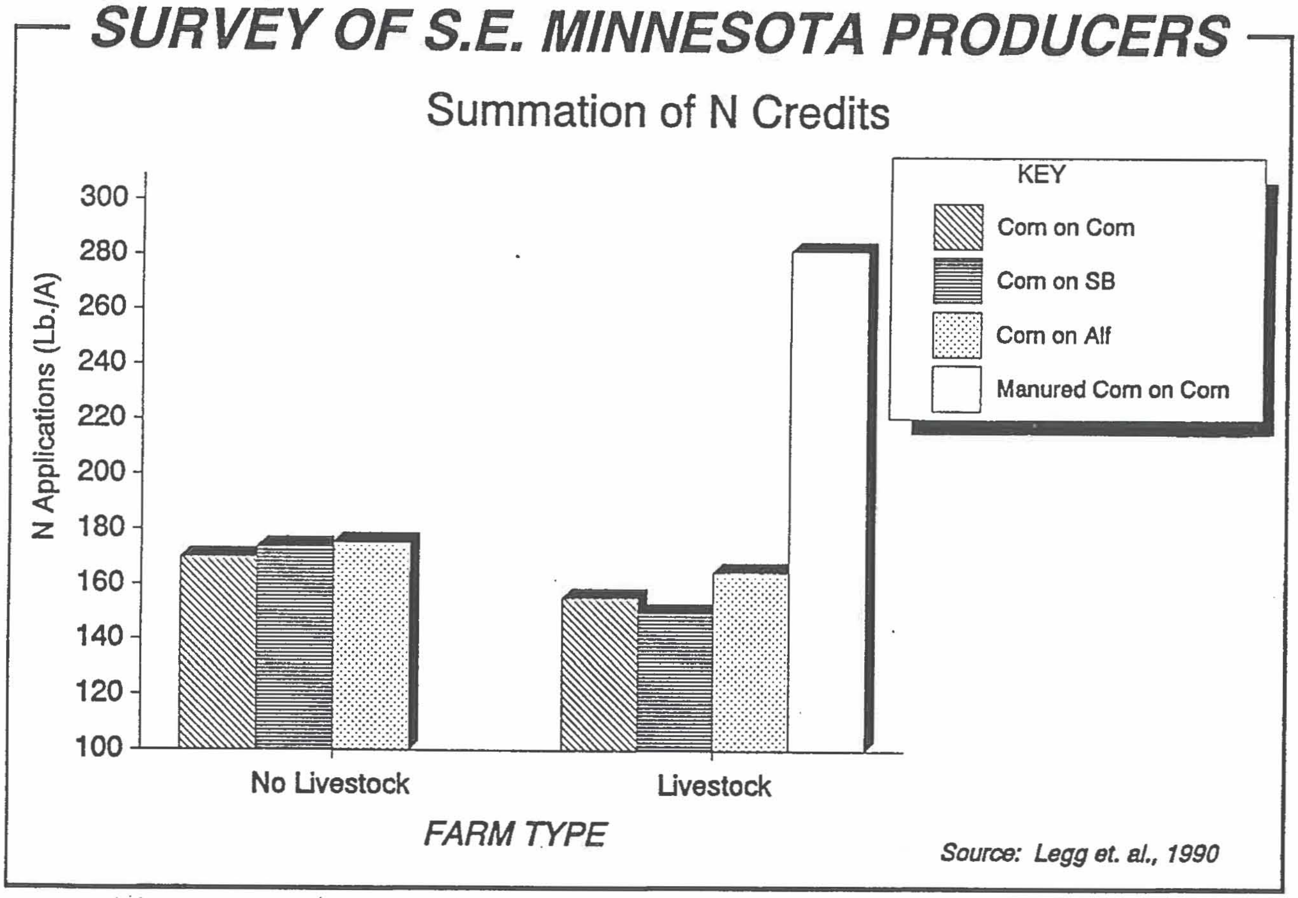

Figure G-4. Nitrogen credits applied to corn from various rotations with and without manure applications in southeast Mn. (Source Legg et al., 1990).

mineralization process (the conversion of organic N to NH_4^+) is controlled by many chemical and biological processes resulting in a gradual but somewhat unpredictable release of inorganic N. Since the conversion is dependent on biological processes and slow in nature, the likelihood of an accumulation of NO_3 at any one time is reduced in comparison to commercial forms. Yet if excessive rates of organic N are applied, environmental degradation problems are likely. Point source contamination problems can also occur where manures are improperly stored (See Chapter H, Feedlots).

A sound manure management program involves awareness of the nutrient value, manure analysis, proper crediting, equipment calibration and uniform application rates, and planned applications (Anonymous, 1989). Sufficient information is available for Minnesota producers to optimize full benefits from manure resources (Schmitt, 1989; Sutton et al., 1985). It must be noted that there are several fundamental methods for managing manure. Maximum efficiency for all nutrients would be achieved based on the nutrient at the highest concentration level which generally would be phosphorus (P). Manure would then be applied at agronomically sound P rates and other critical nutrients would be supplemented with commercial fertilizer. This method would be suitable when sufficient acres of land are available (Anonymous, 1989). More commonly, application rates are applied to fulfill the N demands and is a much more suitable approach when land area is limited. Excess phosphorus can be a surface water threat when there is an erosion hazard. Wisconsin data (Anonymous, 1989) suggests than manure applications must be reduced when P levels reach 150 lb/A in the plow layer. The economic value of the manure is reduced when any one nutrient is applied in surplus.

Another problem associated with manure management is that growers tend to over apply specific fields. Commonly, this is directly related to the distance from the feedlot or barn. In the Beltrami survey (Anonymous, 1990), 40% of the farmers reported applying manure to less than 25% of the cropland. Yet only 22% of the respondents said that distance was a factor and these applications were based on crop needs. Another observation, although not necessarily posing an environmental problem, is that growers tend to apply manure to crops which may not maximize its' full value. Sixty percent of the Beltrami dairy farmers applied manure to alfalfa. More information in a follow-up survey is currently being processed.[10]

Characteristics of Manure and Effects of Handling Methods

Dominant N forms found in manure are organic N, NH4-N, and readily hydrolyzable urea which rapidly converts to NH_4. Ratios of NH_4/organic N and the total N content will vary with animal type, feed sources, and within urine and solid phases but typically 40 to 60% of the total N content is present in the inorganic form. Nutrient values for a variety of livestock has been reported by the Midwest Plan Service (1985). The organic fraction is stable while the fate of NH_4 from the time of excretion to applying as a resource is extremely variable. These gaseous losses do not pose a direct water quality threat but inconsistencies can easily create confusion when attempting to take proper credit and represent economic loss. Sutton et al. (1985) summarized N losses associated with methods of storage and handling. Losses can range between 15-30% in a daily scrape and haul to as high as 70-80% in long-term storage lagoons. Due to the great variability induced by handling phases, manure analysis should be done on a routine basis. In a case study of livestock producers in southeast Minnesota, losses associated with storage and application rates were extremely variable (Legg et al., 1989). Schmitt[11] investigated the N, P, and potassium (K) contents of 26 liquid manure systems in southeast

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10. Personal communication, Jeff Hrubes, Beltrami Soil and Water Conservation District.

11. Personnel communication with Dr. Mike Schmitt, University of Minnesota-Extension Service, Department of Soil Science. Funding provided by the Minnesota Department of Agriculture-Sustainable Ag Program.

Minnesota. Available N contents (lbs/1000 gallons) averaged 29 (range of 10 to 47) for dairy and 49 (range of 24 to 87) for hog handling systems.

Application methods also affect volatilization losses. Ammonia volatilization from surface applied slurry can account for more than 50% of the NH_4 in the manure (Lockyer et al., 1989; Thompsen et al., 1987) and substantial losses commonly occur within hours of the application (Pain et al., 1989; Sommer and Olesen, 1991). Few farmers (less than 25%) incorporated manure in the Beltrami survey (Anonymous, 1990). These losses can be substantially higher if manure is applied on warm, breezy days. Incorporation within 72 hours is highly recommended when possible. Sommer and Olesen (1991) determined the effect of dry matter content of cattle slurry on NH_3 losses. Moisture contents of slurry were altered between 0.9-22% and subjected to wind tunnel tests. Accumulated NH_3 losses during a 6 day period ranged from 19 to 100% from slurries having a dry matter content of 0.9 and 15.6%, respectively. Effects of moisture content on losses were small when the dry matter content was higher than 12% or lower than 4%.

Injection of manure reduces losses significantly (Klausner and Guest, 1981). The fertilizer equivalent from the injection system was three times higher than top-dressed applications. Sutton et al. (1985) reported losses under injection systems to be within 0 to 2%. There are some disadvantages associated with injection. Various forms of crop injury can occur if placed too close to concentrated manure bands (Sawyer and Hoeft, 1990; Schmitt and Hoeft, 1986). Due to the high moisture environment along with abundant NO_3 and carbon within the injection band, denitrification losses can be substantial (Comfort et al., 1988) but are not as large as volitization losses incurred with traditional surface applications (Schmitt, 1989). Sweep knife injection systems have been shown to significantly reduce both volatilization and denitrification losses.

Once manure is soil-applied, another important consideration is the quantity and timing of available N to the crop. The general rule is that 33 to 50% of the organic N will be converted each year after manure application (Schmitt, 1989) although a large number of factors affect the rate of conversion of organic N to inorganic forms. Motavalli et al. (1989) reported that first-year availability from injected dairy manure ranged from 12 to 63% with an overall average of 32%. These authors concluded that more reliable indices for predicting availability are needed and proposed a simple simulation model. Sims (1986) found that soil moisture and temperature were important factors in understanding N availability in poultry manure. Tillage methods influenced the mineralization rates after the first year of application (Joshi et al, 1991). Decay series tables have been developed to predict the proportion of manure N from a given application that will become available in succeeding years (Schepers and Fox, 1989).

Economic costs may be too great and/or paybacks too slow for farmers to purchase the proper equipment to obtain the full nutrient benefits from manure. Legg et al. (1990) summarized that "if farmers consider risk in their objectives, and are risk averse, they may choose not to invest in storage facilities and spreading equipment even when the expected cost savings from the venture is positive". The economic value of manure is dependent upon fertilizer prices and have been comparatively low in the last ten years. Even if farmers had the proper holding facilities, many would probably elect to fall-apply manure. Ideally, manure should be applied in the spring especially in environmentally sensitive areas. Yet time and labor is limited due to other field activities.

The value difference gained in higher nutrient efficiency is overshadowed by high opportunity costs (Legg et al., 1990). The Beltrami survey strongly indicated that late fall was an important manure distribution period (Anonymous, 1990).

Environmental Effects from Animal Manures

There is a limited number of long-term studies addressing the effects of manure management on ground water quality. Similar to commercial N fertilizers, existing data indicate that NO_3 leaching will occur when manure application rates exceed crop needs (Evans et al., 1977; Sutton et al., 1986; Randall et al., 1990; and Roth and Fox, 1990). Additional references relating high application rates to soil NO_3 levels can be found in a review by Smith and Peterson (1982). Losses may not be directly proportional to application rates when grossly excessive amounts are added due to higher denitrification losses resulting from the elevated amounts of labile carbon, NO_3, and water content. As a result, less NO_3 leaching may occur under heavily-manured fields in comparison to excessively-fertilized fields with commercial sources at equivalent rates (Schepers and Fox, 1989; Sutton et al., 1986). Randall et al. (1990) applied very high rates of dairy manure (400 and 690 tons/A, dry weight basis) to a Webster clay loam. A disproportionately higher amount of NO_3 was found under the lower manure rate. Under the management imposed (surface applied on a weekly basis and incorporated), denitrification had a pronounce effect.

Under realistic N rates, the potential for NO_3 leaching losses could be actually greater under manured fields than fields fertilized with commercial fertilizer. This is due to the continual mineralization of the manure after the crop needs diminish. Nitrate remaining in the profile during late fall through early spring is extremely susceptible to leaching due to the low evapotranspiration losses/high percolation rates (Roth and Fox, 1990). Carryover from commercial sources of N, when applied at proper rates and times, should be minimal (Hahne et al., 1977; Herron et al., 1968; and Linville and Smith, 1971).

This simplistic analogy is not supported by some Minnesota research. Joshi et al. (1991) studied the effects of manure and commercial N on corn yield and water quality as a subset of a larger tillage study in Goodhue County. Anhydrous at 175 lb/A of N, manure at an equivalent rate, and the equivalent rate applied biennially were monitored in silt loam soils with suction cup samplers at 5′. Overall trends for 1989 and 1990 were similar and NO_3-N concentrations were ranked accordingly: anhydrous>annual manure>biennial manure. Approximate 1990 NO_3-N concentrations were 60, 35, and <10 mg/L, respectively. Randall et al. (1990) used soil solutions from the 5′ depth to assess the risk associated with commercial N rates of 0, 75, 150, and 225 lb/A and hog manure at rates of 6,000 and 9,000 gallons/A (equivalent to 315 and 490 lb/A of N). Realistic applications of hog manure did result in some leaching losses. This data supports the concept of potential leaching losses, regardless of source, once the crop needs are satisfied.

Summary of Effects of Manure on Ground Water Quality

Potential N generated from Minnesota's domestic animals is an important component of the N cycle and the amount available for crop production is equivalent to 10 to 15% of that supplied by fertilizer N.

Information is sparse regarding how Minnesota farmers store, credit, and apply their manure. Limited state and national information strongly indicate that farmers seldom take proper credit, if any, for manure.

Potential problems associated with not taking the proper credits are compounded by the fact that the heaviest loading is occurring in the states' most sensitive hydrogeologic regions.

Ground water contamination will occur if rates, regardless of the source, exceed the crop needs. Like commercial applications, there are risks associated with manure usage. Due to the slow release, the amount available for leaching at any point in time is limited. Yet continual mineralization will occur after the crop needs are satisfied and there is the potential for "off-season" leaching losses. Few studies have examined these long-term effects.

Storage and handling have an extremely profound effect on the amount of N in the manure by the time it is distributed onto the soil. Gaseous losses from uncollected urine, during the storage process, and during the actual field application can be collectively as high as 15 to 80%. These volatilization losses do not pose a water quality concern but the lack of understanding of these losses will cause considerable confusion in crediting the portion of the N which does eventually get applied to the field.

The Nitrogen Fertilizer Task Force highly recommends manure analysis.

The general consensus among researchers is that farmers need to be better educated in manure management. An overall effort to educate farmers must include manure and legumes as well as commercial fertilizers if a ground water protection program is to be successful.

EFFECTS OF LEGUMES ON GROUND WATER QUALITY

Introduction

Legumes form a symbiotic relationship with a genus of bacteria (Rhizobium) which converts N_2 (atmospheric dinitrogen) to a useable form for the plant. This process is commonly termed 'symbiotic N fixation'. Rhizobium, while supplying the N the plant needs, obtains soluble carbohydrates from the host (the legume) for energy. Scientists are re-examining the importance of legumes and manures in agricultural systems for a number of reasons: 1) depletion of the ozone layer may be accelerating and the loss of nitrous oxide gas from soils may be a significant contributor; 2) high energy consumption to produce N fertilizers; and 3) the increasing evidence of ground water contamination by agricultural activities (Peterson and Russelle, 1991). Legumes are important crops in Corn Belt states (Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, Ohio, and Wisconsin) and these states account for 40% of the nation's alfalfa production. Peterson and Russelle (1991) estimated that the annual total amount of N fixed in Minnesota was in excess of 200,000 tons. Fixation amounts vary considerably depending on the age of the stand and other factors; estimates range from 60 to 360 lb/A. Although most of this fixed N is removed in the harvesting, these authors estimate that the annual fixed N_2 input directly to the soil was 92 lb per alfalfa acre.

Alfalfa and soybeans are Minnesota's most common legume crops. During 1985-89, these crops occupied an average of 1.9 and 4.9 million acres, respectively (Minnesota Agriculture Statistics Service, 1990). Based on the assumptions below, legumes accounted for 96,000 tons of available N[12] in 1987. Legumes accounted for 12% of the "plant available N" based on the total contributions from fertilizer, legume and manure credits (See Figure G-2). Nitrogen contributions from alfalfa and soybeans on a county level are illustrated in Figure G-5. The following assumptions were made:

1) Soybean and alfalfa acres were obtained from the 1987 Census of Agriculture.

2) Contributions from alfalfa and soybean were assumed to be 75 and 30 lb/A, respectively.

3) Contributions from other legume hays such as red clover were not included. Census data only distinguishes the categories of 'alfalfa' and 'other hays'.

4) Density calculations were based on equal distribution across all acres classified as "cropland" in the 1987 Census of Agriculture. This would include acreage such as pasture and Conservation Reserve Program (CRP) lands. This would not include areas in forests, lakes, and urban acreage.

5) Assumed that 33% of the alfalfa acres were plowed down each year.

12. Available N refers to the portion of the total N pool which is readily absorbed and assimilated by growing plants.

Contributions from the sum of alfalfa and soybeans range from 0 to 12 lb/cropland acre/yr. Heaviest loadings are in the southern third of the state due to high soybean acres and in the north central region due to a dominance in alfalfa production. Keep in mind that this map represents a density loading across crop acres. What appears to a significant amount of N being supplied by alfalfa in the northeast portion of the state is in actuality quite small in terms of total pounds: total crop acres is small and heavily dominated by this legume. Clover and other hay crops[13] account for another 1.1 million acres. The relative importance of clover in some counties could alter the loading map significantly.

Complexities associated with N contributions from legumes to succeeding crops are poorly understood although great progress has been made in recent years. University of Minnesota-Extension Service is currently crediting established alfalfa, red clover, and soybeans at rates of 75-150, 75, and 20-40 lb/A, respectively (O'Leary et al., 1989; Rehm et al., 1991). A number of recent Minnesota studies have verified these contributions (Lory et al., 1991; Bongard et al., 1991). A three year study by Wagar et al. (1989) demonstrated that alfalfa and manure supplied enough N for two cropping seasons of corn. Yields were not increased by the application of additional N until the third year. A long-term Wisconsin study has clearly identified second and third-year contributions from alfalfa[14]. Other studies have suggested that contributions attributed from legumes are inflated (Hesterman et al., 1987; Bruulsema and Christie, 1987; Harris and Hesterman, 1990; Hesterman et al., 1986). Reasons for the apparent inconsistencies may be due to complications from "rotation effects", not accounting for exuded N from the roots or organic contributions below 12", or management factors affecting N availability. Additional discussion on complexities associated with crediting legumes can be found in Schepers and Fox (1989). Peterson and Russelle (1991) summarized the seemingly poor progress in understanding the complex nature of N release by stating "..even after decades of research, accurate prediction of soil N mineralization eludes researchers".

Similar to manure, many studies are now revealing that growers seldom take the proper credits following a legume crop. Peterson and Russelle (1991) calculated that the amount of N fertilizer used in the Corn Belt could be reduced by 8 to 14% without any reduction in yield simply by taking the proper credits for legumes and manures. These authors cite these additional studies which strongly indicate poor legume management:

Case 1: An Economic Research Service survey of 1,700 farms showed that N fertilizer rates for corn following a legume averaged only 9 lb/A less than continuous corn (Daberkow et at., 1988).

Case 2: In Iowa, 58% of the surveyed fields in corn following alfalfa had also received applications of manure the previous year (El-Hout and Blackmer, 1990). In a subsequent study (Morris et al., 1991), the application of 25 lb N/A following alfalfa optimized corn production 3 out of the 4 study years. Yet surveys of local farmers indicate that the average N application rate was 120 lb/A.

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13. Minnesota Agricultural Statistics Service does not differentiate clover from non-legume hay crops. Therefore the N credits have not been calculated for their acreage.

14. Personal communication, Dr. John Moncrief, Mn. Ext. Ser.

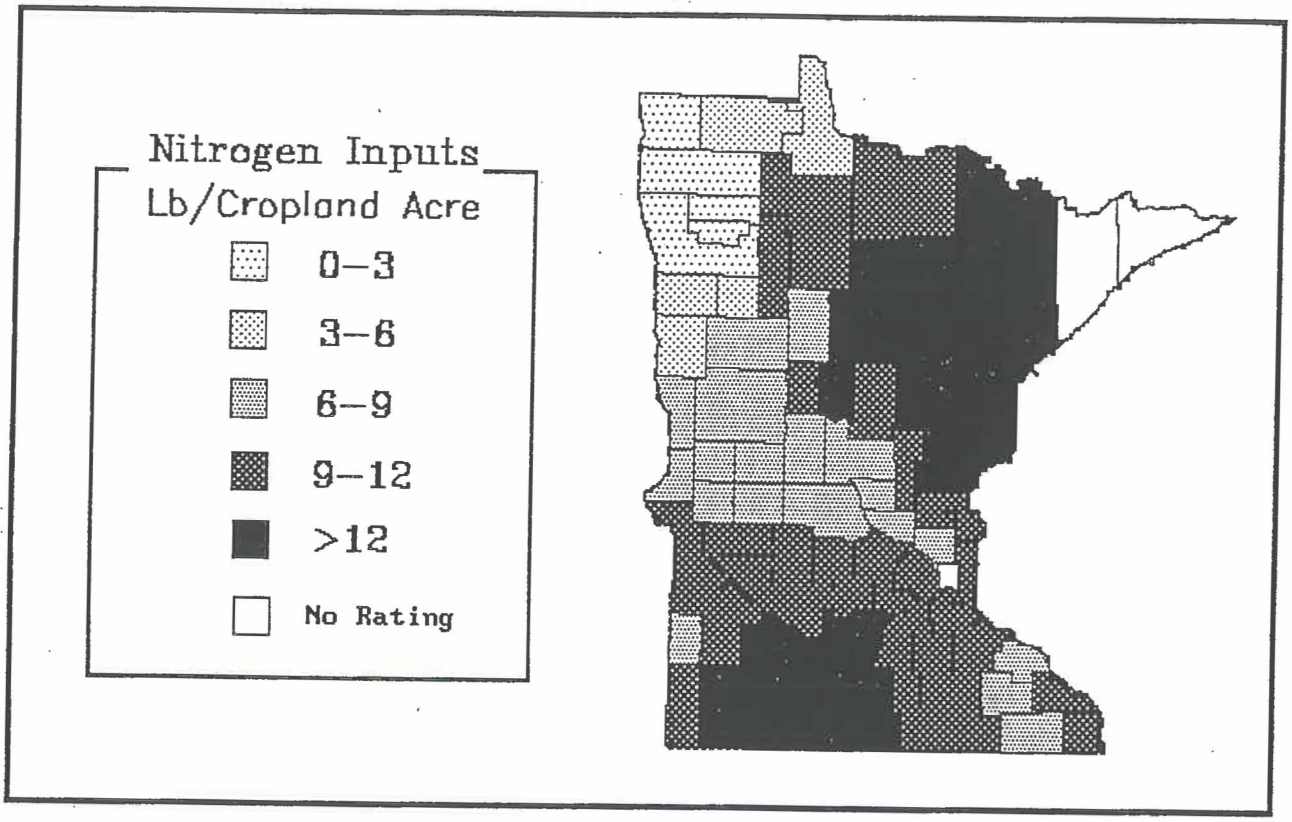

Figure G-5. Estimate of plant available N contributions from soybeans and alfalfa in 1987. Densities are calculated on the amount of cropland acres in each county. See text for additional assumptions and calculation methods.

The practice of applying manure to legumes can have some beneficial results. This practice offers the producer the opportunity to supply phosphorus and potassium to the crop. It also provides a mechanism for disposal of manure when not needed for the N credits elsewhere in the farming operation. Legumes will utilize soil NO_3 in preference to fixation. This practice is not the most efficient use of the N resource and would be considered a disposal option. Alfalfa fields may be the only sites available for manure application at certain times of the year (Peterson and Russelle, 1991). Manuring an alfalfa crop prior to plowdown poses some serious environmental consequents.

Environmental Effects of Living Legumes and Following Plowdown

Although the practice of incorporating a legume into a rotation to increase the N status has been utilized for a very long time, there is some data suggesting that some legumes may be beneficial in reducing the NO_3 concentrations of the soil water while the crop is living. Alfalfa, because of its' deep, extensive root system, is excellent at scavenging residual or mineralized N. This crop has the potential to serve an important role in capturing residual NO_3 which is commonly elevated after corn or potato production (Muir et al., 1976). Alfalfa will utilize residual soil NO_3 in preference to fixing atmospheric N. Researchers have utilized alfalfa's ability for intercepting NO_3 in abandoned feedlots (Schuman and Elliot, 1978; Mielke and Ellis, 1976). The ability for soybean to scavenge NO_3 is not well documented.

Owens (1990) monitored leachate under five years of corn production and NO_3 concentrations often ranged from 15 to 40 mg/L depending on fertilizer rates. When an alfalfa-orchard grass mix was established, the concentrations during the next two years dropped substantially. Nitrate concentrations were commonly under 5 mg/L. Low concentrations and losses (<5 lb/A/yr) under growing alfalfa were also reported by Bergstrom (1987). Drainage characteristics under a mix of alfalfa and orchardgrass were found to be very low in NO_3 concentrations and leachate quantities (Chichester, 1977). Even with annual N applications of 160 lb/A/yr, NO_3-N concentrations seldom exceeded 3 mg/L and losses averaged 9 lb/A/yr over the 3-year study. In a Canadian study, NO_3 losses under alfalfa tended to be 25 to 50% less that those found under either rotational or continuous corn (Bolton et al., 1970). Nitrate-N losses under irrigated alfalfa in St. Paul were reported at 23 lb/A and the major losses occurred in April and May (Peterson and Russelle, 1991).

Additional literature supporting the concept of actively growing alfalfa as an environmental benefit is reviewed by Peterson and Russelle (1991). Not all of the research is in complete agreement. Russelle and Hargrove (1989) reviewed a number of reports which documented that living legumes can adversely impact ground water. Nitrate leaching losses and/or excessive soil NO_3 accumulations were found.

There is a unified concern that NO_3 levels will be elevated after the stand is killed due to the high amount of N released during the decomposition stages. Effects of irrigated alfalfa production under Idaho silt loams were studied over a 2-year period (Robbins and Carter, 1980). Nitrate concentrations and leachate losses under active alfalfa stands were 3-15 mg/L and 40 lb/A/yr, respectively. Long-term results after alfalfa termination were spectacular. Losses under unfertilized crops following the alfalfa were commonly 54 to 86 lb/A/yr. These data strongly suggest that great care must be taken in determining N application rates and selecting crops to be planted on lands following alfalfa. Bergstrom (1987) also observed high NO_3 concentrations and losses after either non-fertilized alfalfa or previously fertilized grasses where plowed up. Nitrate concentrations under irrigated corn following an established alfalfa stand in Wadena county appeared to be significantly elevated by alfalfa decomposition and method of tillage (Moncrief et al., 1991a). In a field plot in Winona County, Randall et al. (1991) observed long-term impacts from alfalfa. No N fertilizer has been added to this field since manured alfalfa was plowed down in 1985. In 1990, soil solution NO_3-N concentrations at five feet averaged 13 mg/L. The authors concluded that "the role of alfalfa and manure contributions to available N for succeeding corn crops needs to be carefully examined and understood before improved N management is a reality on these soils (fine-textured, high organic matter soils of southeastern Mn)".

Zadak et al. (1989) initiated research in southeast Minnesota to examine residual soil NO_3 following alfalfa with tillage and corn hybrids as the variables. Net gains in soil NO_3 from spring to fall, even after producing a 130 bu/A, were approximately 140 lb/A. Nitrogen rate, tillage, and corn hybrid also affected NO_3 levels.

Effects of soybeans on ground water has not yet been clearly identified. Effects are commonly masked by high residual carryover from previous crops (Baker and Johnson, 1981). Owens (1990) concluded in a 1990 study that "with the increasing awareness of the need to preserve ground water quality and with a renewed interest in crop rotations, legumes may have an important role in agriculture and its effects on the environment". Yet it is clear that legumes, like any other N source, can eventually have a detrimental effect when not properly managed. Effects are magnified by not taking full N credits for a legume and also by the common practice of applying manure to alfalfa stands just prior to plowdown (Lory et al., 1991).

Summary of the Effects of Legumes on Ground Water Quality

Legumes are an important source of N in some areas of Minnesota. Legumes contribute 10 to 15% of the N supplied by the sum of manure, fertilizer, and legumes. Heaviest legume loading (in proportion to the cropland acreage) is from soybeans in the southern one-third of the state and the north central portion due to alfalfa contributions.

Existing literature is in full agreement that plowing down or other methods of killing the alfalfa increases the potential for NO_3 leaching losses. As the roots and remaining above-ground residue decompose, the mineralization rate will commonly exceed the following crop N needs, resulting in a potential NO_3 contamination problem.

Care must be taken in selecting high N use crops and avoid applying any other sources of N after terminating the alfalfa. Credits for any legume crop must be recognized in an overall management plan. The practice of applying manure before plowing down a legume must be eliminated.

Problems associated with legumes in N management appear to be simplistic in comparison to those associated with manure. Proper crediting appears to be straight forward and the only estimates the grower has to make is population estimates in alfalfa. Proper record keeping and uniformity problems across the field are minimal.

Effects of other crops such as soybean and clover have either limited or no research to assess their direct impact on ground water.

FERTILIZER RATE EFFECTS ON GROUND WATER QUALITY AND YIELDS

Introduction

Although N sales in Minnesota have increased dramatically in the past 30 years, data from 1985-90 indicate that annual sales are leveling off (See Figure G-6). Nitrogen usage in Minnesota has increased from 104,000 tons utilized in 1965 to 647,000 tons in 1990 representing an increase of 500% (Berry and Hargett, 1988; Berry and Hargett, 1990). Statewide N inputs on corn has increased from 35 to 127 lb/A over the last 30 years (MDA, 1990). Rates have increased by a factor of 3 to 5 fold while yields have increased by a factor of 2. Similar increases were reported from Iowa (Hallberg, 1987) where rates on corn have increased from 45 lb/A (1965) to 143 lb/A (1984). Trends across the remainder of the Corn Belt are similar.

A 1990 survey (National Agricultural Statistics Service, 1991) reports statewide applications at 109, 87, 80, and 2 lb/A for corn, potatoes[15], wheat, and soybeans, respectively. Estimated N rates for the remaining significant crops were calculated[16] and illustrated in Figure G-7.

Corn production in Minnesota has averaged approximately 6 million acres since 1985 (Minnesota Agriculture Statistics). Dominance of this crop is the main reason why Minnesota is one of the nation's largest consumers of commercial fertilizer. In 1990, 553,000 tons of N[17] were sold in the state and this value represents 6.3% of the nation's sales (Hargett and Berry, 1990). Minnesota ranks fourth in national level sales behind Iowa (835,000 tons), Illinois (760,000), and Nebraska (671,000) (See Figure G-8).

Crop area percentages in 1990, as estimated by NAAS (1991) and MN Agricultural Statistics (1991), were 37, 24, 23, and 12% for corn (all types), small grains, soybeans, and hay crops, respectively (Figure G-9). Sugar beets, potatoes, and other miscellaneous crops[18] accounted for the remaining acres. Approximately

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15. Due to the limited area incorporated within the NASS pilot project, the estimated rates on potatoes is probably a better reflection of dryland production in northwest Mn. Application rates under the estimated 35,000 acres of irrigated potatoes is 2 to 3 times higher.

16. N rates for those crops not estimated by NASS (1991), the following procedur was utilized: Based on the average state yield, the N required to grow a specific crop was estimated based on UM soil testing recommendations. A residual soil nitrate value of 40 lb A^{-1} was assumed across the entire state for all crops. The N rate was then multiplied by each crop's total acres. Following this procedure, the cumulative amount of N fertilizer estimated for the state was 578,380 tons which compares favorably to actual sales of 553,000 tons reported by TVA (Hargett and Berry, 1990).

17. Tonnages reported here include only single-nutrient nitrogen materials and a natural organics. Principal multiple-nutrient grades are not included; 1990 nitrogen tonnages from multiple grades (such as 18-46-0 and 10-34-0) would add another 94,000 tons to Minnesota sales (MDA files).

18. Includes edible beans, sunflower, and wild rice.

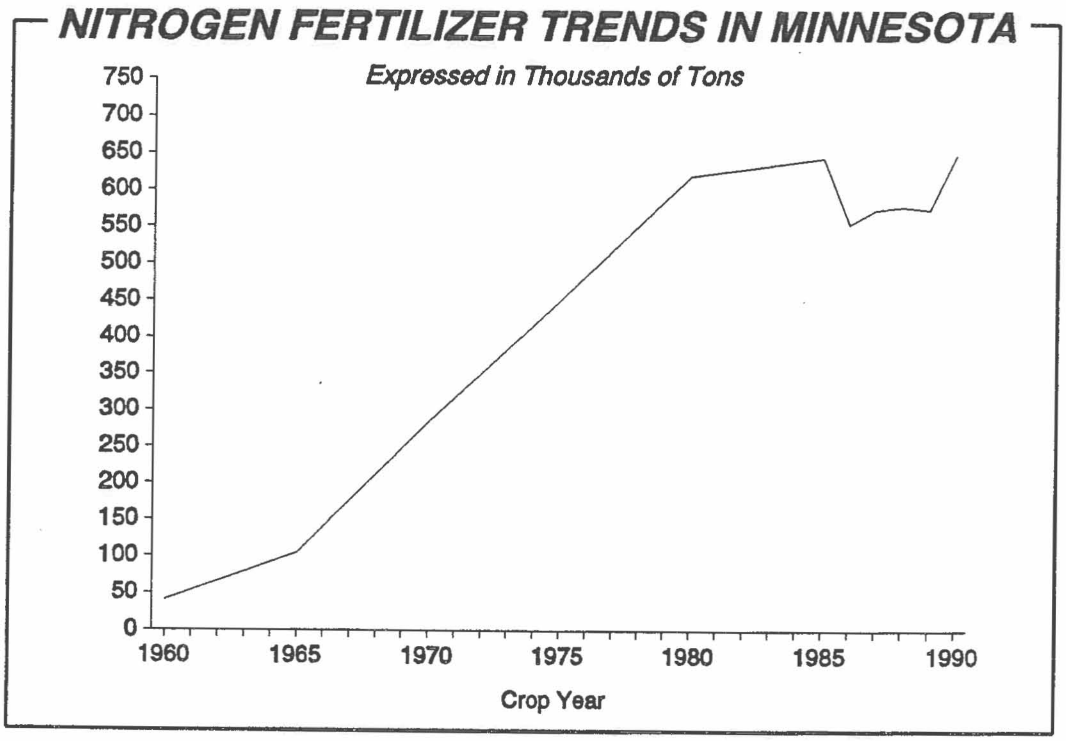

Figure G-6. Nitrogen fertilizer sales in Minnesota from 1960 through 1990. (Source: Berry and Hargett, 1988; Berry and Hargett, 1990).

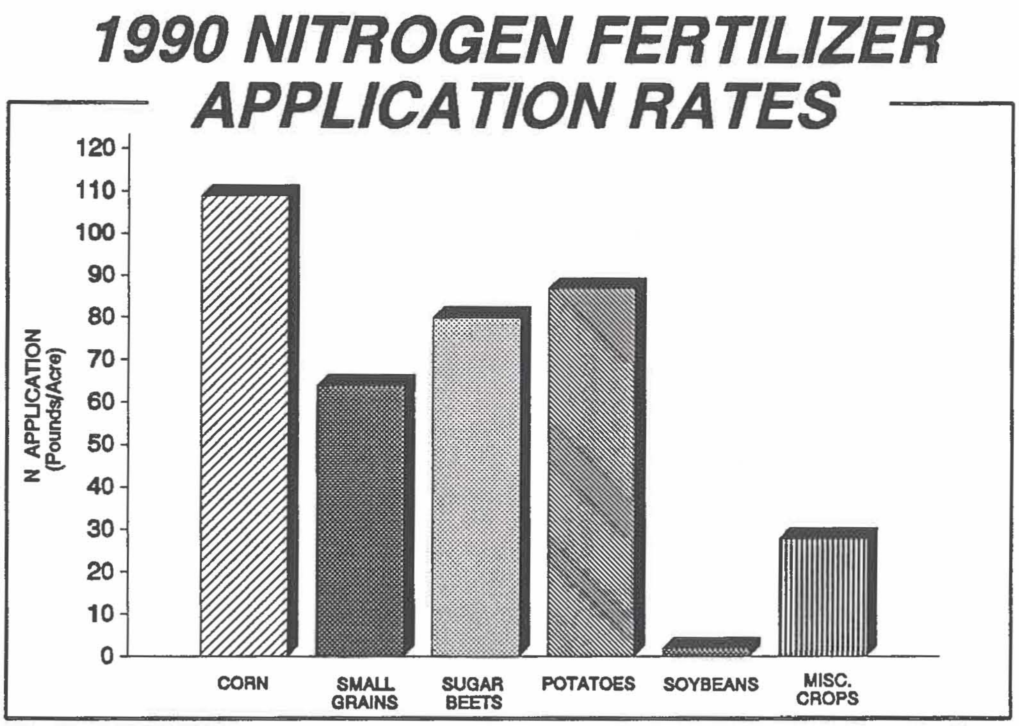

Figure G-7. Estimated N rates during the 1990 season for Minnesota's major crops. Nitrogen use estimates for corn, small grains, wheat, and soybeans from NASS, 1991.

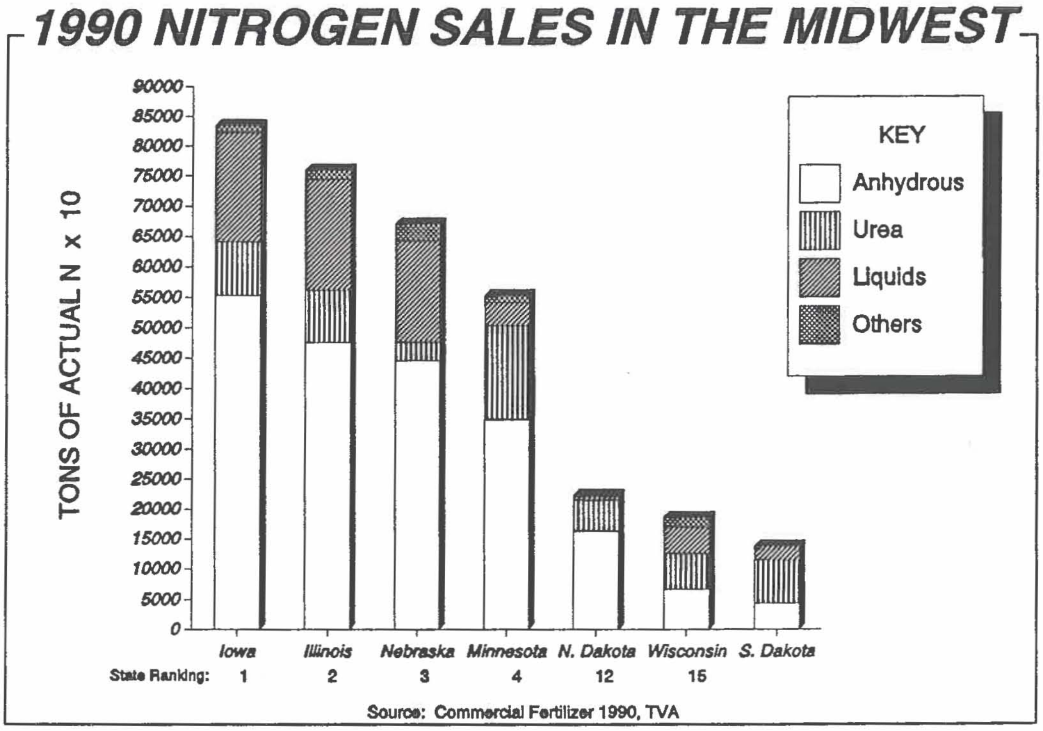

Figure G-8. Single-nutrient nitrogen sales in selected Midwestern states in 1990. Source: Hargett and Berry, 1990.

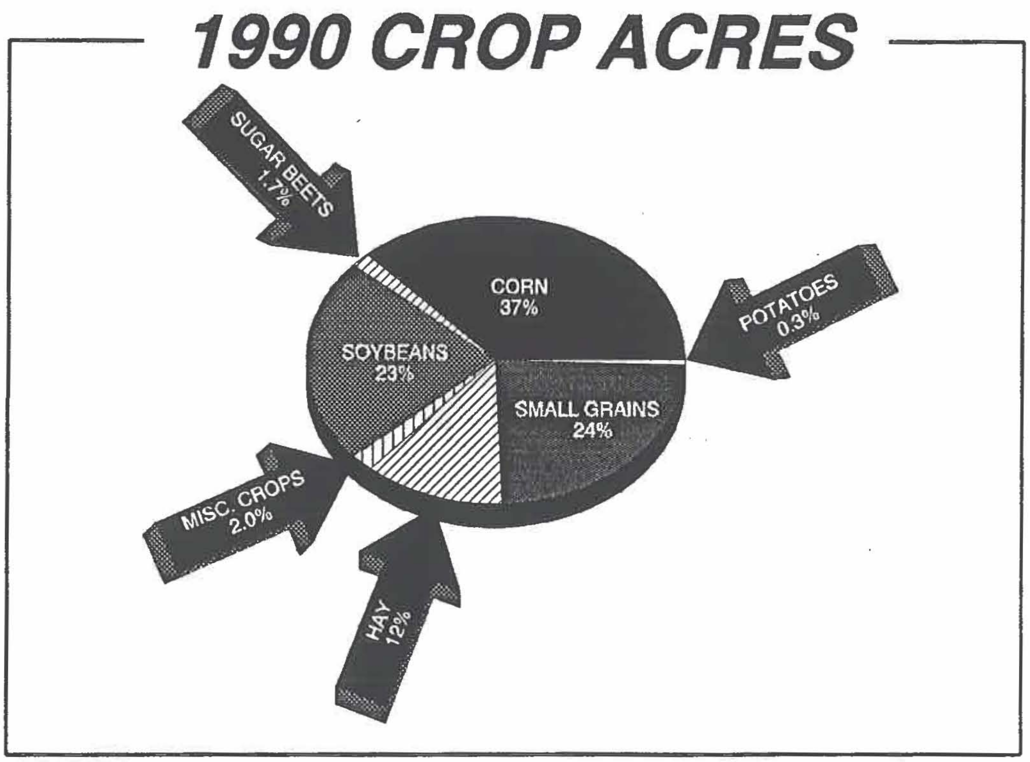

Figure G-9. Crop types and distribution in Minnesota during 1990.

69%[19] of the state's commercial N fertilizer was applied in 1990 to corn for grain, silage or sweet corn production (Figure G-10). Small grains (wheat, barley, oats, and rye) account for another 26%. Sugar beets, soybean, miscellaneous crops[20], and potatoes received 2.3, 2.0, 1.1, and 0.6% of the total, respectively.

Estimates of county level N fertilizer usage are presented in Figure G-11. Data is based on 1987 sales (EPA, 1990). The following assumptions were made:

1) Fertilizers were utilized within the county where the sale occurred.

2) Only fertilizers sold in 1987 were land-applied.

3) Fertilizers were utilized 100% for agricultural purposes. Some portion of these materials undoubtedly were applied to flower gardens and lawns.

4) Density calculations were based on equal distribution across all acres classified as "cropland" in the 1987 Census of Agriculture. This would include acreage such as pasture and Conservation Reserve Program (CRP) lands. This would not include areas in forests, lakes, and urban acreage.

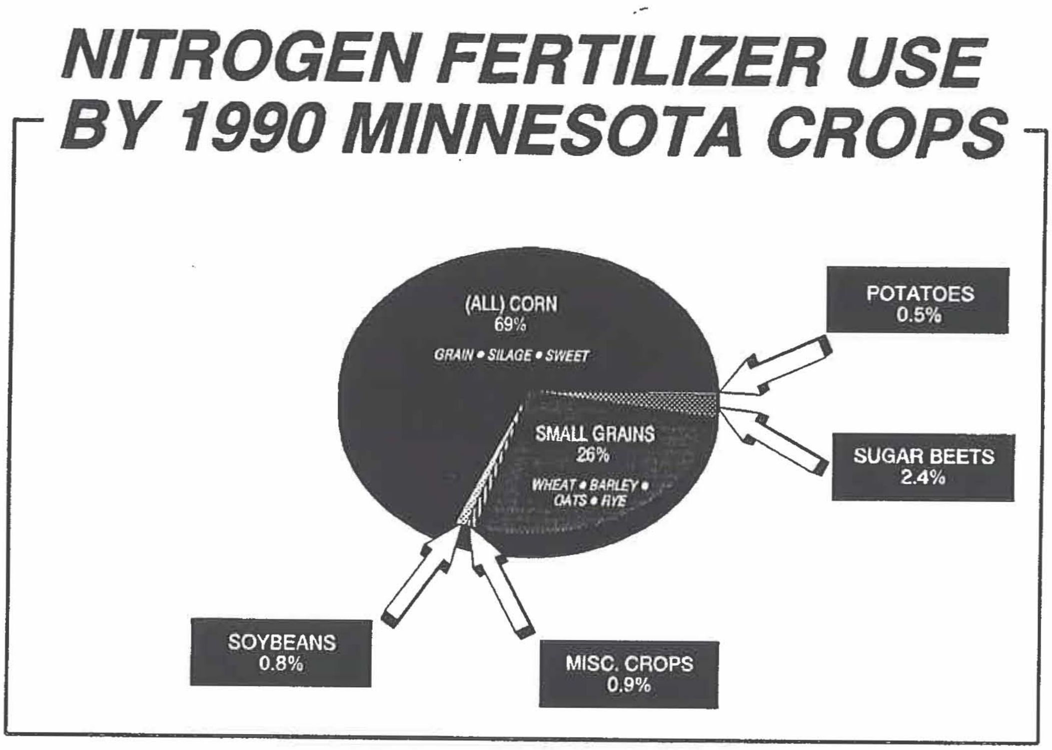

Figure G-10. Estimated N fertilizer usage by MN crops grown in 1990.

19. National Agricultural Statistics Service (NASS, 1990) estimated acres and applied N rates for grain corn, potatoes, and soybeans. Acreage of other crops (1990) were supplied by Minnesota Agricultural Statistics. Based on average yields from this report, estimates for applied N were made based on UM Soil Testing recommendations. Assumed that 40 lb A^{-1} of residual N was available.

20. Sunflowers, wild rice, and edible beans.

Effects from the increased N usage will be divided and reviewed within two broad categories. Large, broad scale monitoring results will first be examined. These tend to focus on an individual watershed or incorporate a number of watersheds. The other category is small, research orientated studies. Some of the advantages and disadvantages have been previously discussed.

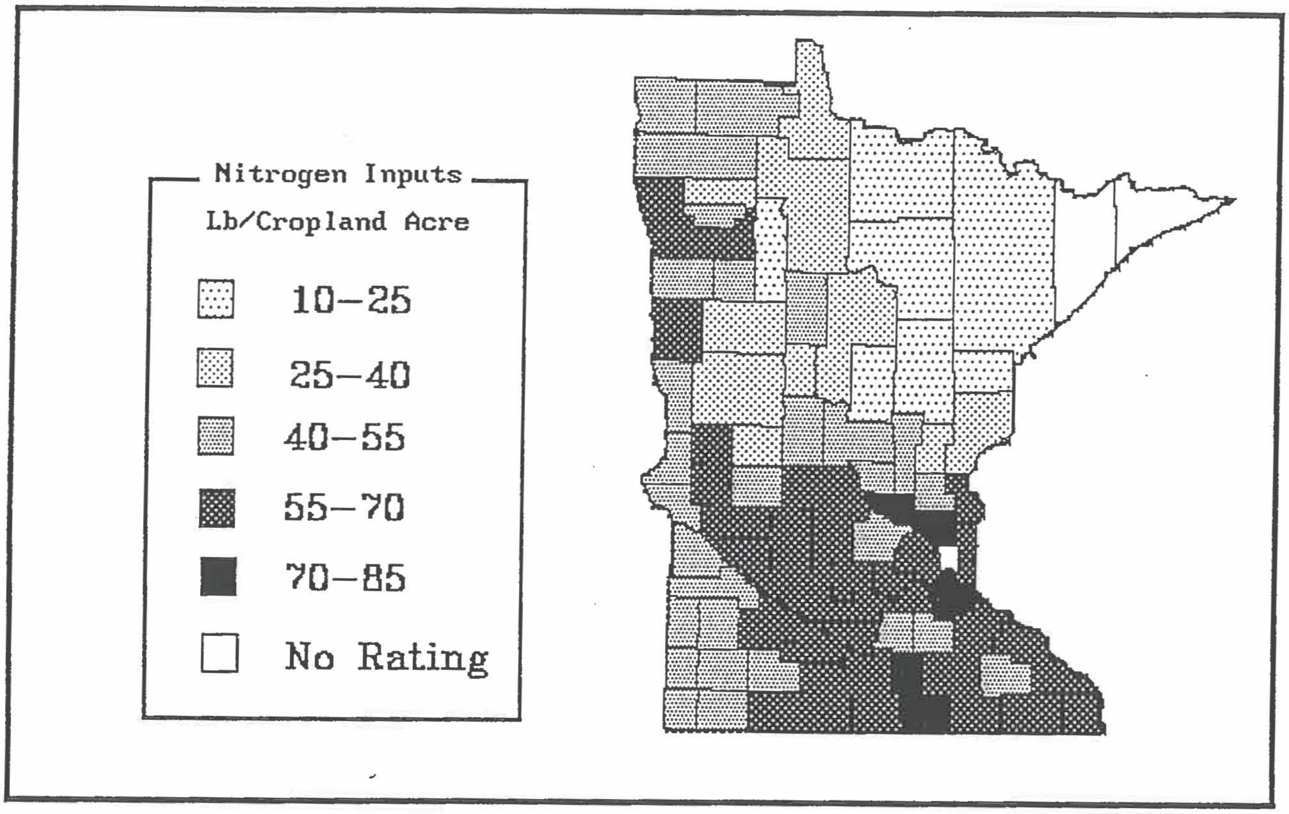

Figure G-11. Nitrogen contributions from N fertilizer based on 1987 county level sales (EPA, 1990). One of the key assumptions here is that the N is equally distributed across all cropland acres. See text for additional assumptions and calculations.

Cumulative Contributions from Fertilizers, Manures, and Legumes

Total inorganic contributions from fertilizers, manures, and legumes on a county level are illustrated in Figure G-12. This map is particularly important since it is the total amount of available N in the soil profile, regardless of its source, that is the single most important factor affecting leaching losses. It is important to keep in mind the assumptions made for each of the individual maps (G-3, G-5, and G-11).

Inorganic N contributions on a cropland acre basis in 1987 were estimated at 53, 9, and 9 lb/A for fertilizer N, manures, and legumes, respectively. Contributions in northern Minnesota were commonly below 50 lb/cropland acre, counties on the western edge were routinely between 55-70 lb/cropland acre, and the southeast and south central counties were generally higher than 70 lb/cropland acre. No individual county exceeded 95 lb/cropland acre.

It is extremely difficult to conclude how these values relate to ground water NO_3 concentrations and leaching losses. Sufficient data (ie; residual soil NO_3, N uptake by crops, organic contributions, areas traditionally receiving manure, etc.) does not exist to do a complete N balance to estimate how much, if any, excessive N is being applied. However, the associated maps do serve as an aid in directing which counties should receive the most education and that educational efforts and research be focused on the appropriate potential sources.

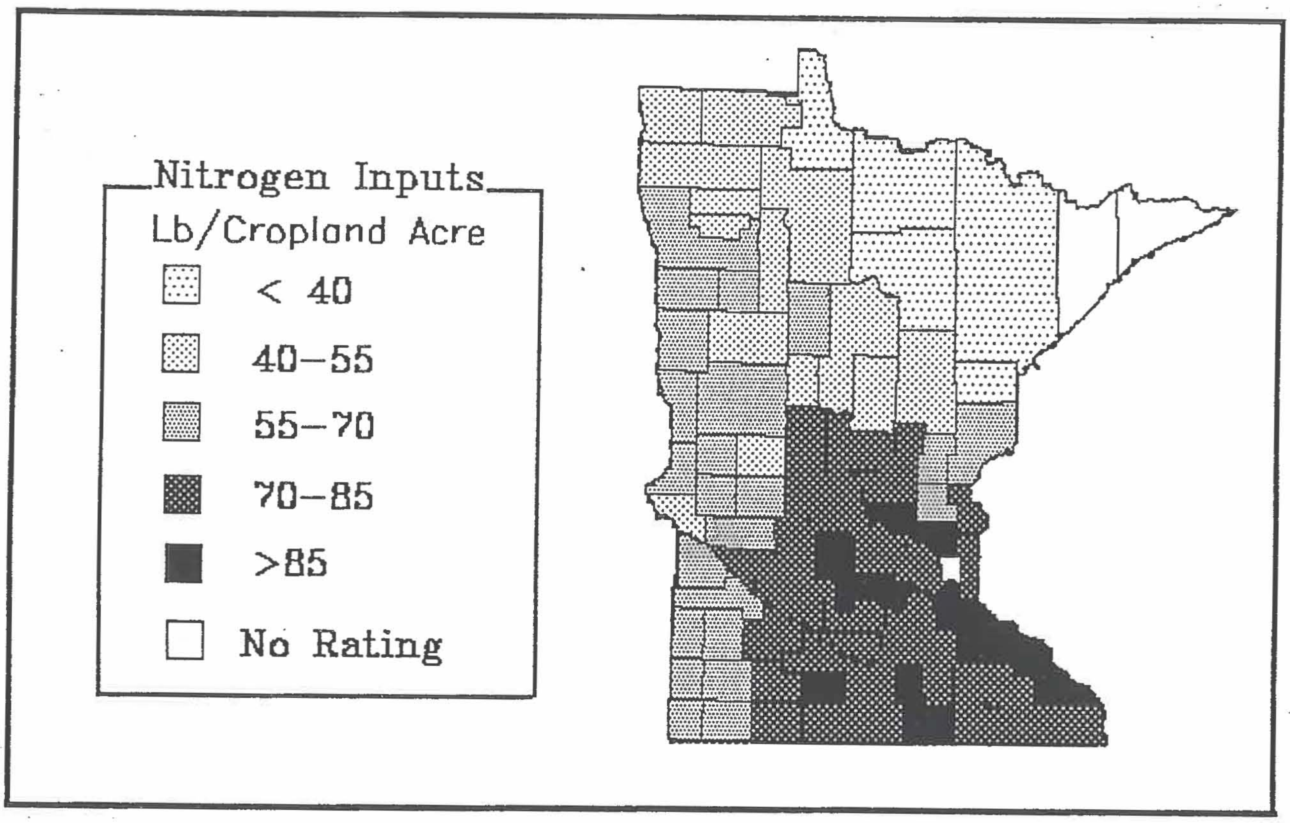

Figure G-12. Estimate of plant available N contributions from legumes, manures, and fertilizers in 1987. See Figures G-3, G-5, and G-11 and the associated assumptions.

Broad Scale Leaching Loss Studies

A small number of Midwest studies have provided valuable long-term, large-scale information demonstrating the relationship between agricultural N inputs and ground water quality. The Big Spring Basin in northeast Iowa is extremely unique in that the 103 $mile^2$ basin is well defined hydrogeologically. Discharge and water quality measurements are obtained at a spring in which the entire basin drains. Land use is almost completely agricultural, eliminating complications from other N sources. Primary sources of N are manure and commercial fertilizer. Farming practices and inputs are inventoried (Kaap, 1986) and water quality data has been collected since the 1930's. Hallberg (1989) reported that the increase in NO_3 concentrations have directly paralleled the increase usage of fertilizer N. Concentrations found in the 1930's were less than 1 mg/L. During the 1950's and 1960's, NO_3-N concentrations averaged 3 mg/L and by the 1980's the concentrations had increased to 9 mg/L. Fertilizer N increased 3-fold in the past 30 years as a function of increased corn acreage. Estimates from the entire basin, along with small scale studies within the basin, indicate that losses equivalent to 33 to 55% of the average application rate are lost to the ground water system through leaching. Annual NO_3-N leaching losses of 45 to 70 lb/A/yr have been estimated. Hallberg has reviewed other Iowa studies and similar trends were found. Kapp estimated that as much as 80 lb/A/yr is added in excess of crop needs and attributed that much of this excess was due to the fact that the farmers were not taking the proper credits from legumes and manures.

Spalding et al. (1978) reported that NO_3-N concentrations in Merrick County, Nebraska, increased from 3 to 12 mg/L during the early 1950's to 1974. Increases of 1 mg/L/yr were also reported in Buffalo and Hall Counties by Ferguson (1990). Most of these trends have been observed within irrigated, sandy soils although one study reported similar increases in fine-textured, deep (vadose>90') soils (Spalding et al., 1988). Soil NO_3 were being transported at rates of 5 to 6' per year through these silt loam soils when flood irrigated. In another Nebraska study (Schepers et al.,1991b), 3,500 farms were surveyed. A strong correlation existed between application rate (deviation from the recommended rate) and NO_3-N concentrations in the irrigation wells.

Small Scale Leaching Loss Studies

Of all the various N management research, the overall effects due to N rates is the most voluminous. Much of the work tends to be short term and performed on small plot research sites. Long-term effects on ground water are seldom clearly defined. Many existing studies limit the scope to only the cropping season and commonly miss the full treatment effect since the majority of the annual leaching losses can occur during the non-cropping season (Montgomery et al., 1988; Gast et al., 1978). Many research citings have utilized inferred methods of loss estimates and it is difficult to compare leaching loads or concentrations from one study to another due to methodology inconsistencies. Despite many of the short comings of the present literature, best management strategies can be formulated. Due to the numerous publications dealing the N rates, the literature reviewed within this chapter was screened using the following criteria: studies which utilized reasonable, agronomically sound N rates and management schemes; NO_3 leaching loss measurements or other reasonable criteria for assessing effects of N rates on ground water, soil water solutions, or residual soil NO_3 were made; and limited the geographical scope to Minnesota and its neighboring states.

Effects of N Rates under Dryland Agriculture

Gast et al. (1978) studied the effect of rates of urea, ranging from 18 to 400 lb/A as N, on NO_3 concentrations in tile-drain effluent over a 3-year period in southwest Minnesota. Plots, isolated with plastic sheeting to a depth of 6′, were installed in a Webster clay loam. Rates exceeding 100 lb/A influenced concentrations after the first full year. Effects accumulated as soil NO_3 steadily increased during the next 2 years which were accelerated due to poor yields as a result of dry growing conditions. Annual N losses (all years) averaged 13, 16, 28, and 54 lb/A and drainage NO_3-N concentrations averaged 17, 21, 31, and 53 mg/L for the 18, 100, 200, and 400 lb/A rates, respectively. After seven years at the same site, Randall et al. (1986) reported concentrations averaging 16, 47, 106, and 172 mg/L for the same respective rates. Soil NO_3 accumulated under rates as low as 100 lb/A. These authors concluded that the drainage waters from these high organic matter soils in southern Minnesota will seldom contain NO_3 below the drinking water standard regardless if any N fertilizer is applied. No additional N was applied after 1979, yet the concentrations in the tile lines still showed residual treatment effects in 1984 averaging 12, 15, 18, and 33 mg/L from the 18, 100, 200, and 400 lb/A rates. These finding were considerably different than the results found in low organic matter, coarse-textured soils where no treatment effects were carried over from the previous year (Clay et al., 1990).

Results from a similar experiment at Waseca were also reported by the same authors (Randall et al., 1986). After 3 years of annual application rates of 0, 100, 200, and 300 lb/A, NO_3-N concentrations of the tile drainage averaged 13, 41, 58, and 85 mg L^{-}. Results from Lamberton and Waseca have shown the potential for soil NO_3 accumulation and poor N utilization under dryland agriculture when moisture is the limiting factor.

Randall et al. (1990, 1991) determined the relationship of N rates and the resulting corn yields and soil solution concentrations at 3 farm sites in southeast Minnesota. Rates maximizing yields ranged from 50 to 150 lb/A. One site was apparently affected by past management of alfalfa and manure. Nitrate-N concentrations at 5′ obtained at crop maturity provided a good basis of comparing the environmental risks associated with the rates. Background levels (check plots with 0 N applied) ranged from 1 to 19 mg/L. Yet it was evident that a strong correlation existed between N rates and solution concentrations. Concentrations at all sites appeared to increase rapidly after reaching maximized yields.

Nitrate accumulations in clay loam profiles at Morris and Waseca were monitored for 10 to 15 years (MacGregor et al., 1974). Application rates of 0, 40, and 240 lb/A were applied annually to corn. Even under the excessive rate (240 lb/A) on the tile-drained LeSueur clay loam, no increases in the 4 to 32′ range were noted although the 0 to 4′ zone did increase to 500 lb/A over the study period. It was speculated that N removal via the tile drains was the major loss mechanism. The Forman clay loam (not tile-drained) also increased to about 500 lb/A in the top 4′ but unlike the LeSueur profile, this profile showed a net gain of approximately 1100 lb/A. Soil NO_3 showed no net gains in either soil profile at the 40 lb/A N rate.

Schuman et al. (1975) studied the environmental impacts of 150 and 400 lb/A rates on two small Iowa watersheds during a 3-year period. Minimal soil NO_3

accumulated at the recommended rate (150 lb/A) although ground water levels did increase from 2 to 4.5 mg/L. Ground water concentrations below the 400 lb/A (2.5 times greater than the recommended rate) increased from 4 to 13 mg/L. Another watershed study was conducted in Iowa over a 4-year period (Baker et al., 1975). Corn, fertilized with 100 lb/A, was rotated with non-fertilized oats or soybeans. Nitrate losses were highly dependent on outflow which varied from 0 to 11" per year. Nitrate-N concentrations averaged 21 mg/L and losses ranged from 0 to 83 lb/A.

Effects of N Rates under Irrigated Agriculture

Walters and Malzer (1990 a; 1990 b) utilized 27 drainage lysimeters and tagged ^{15}N fertilizer to evaluate in a factorial design the effects of rates, incorporation methods, and nitrification inhibitors on the fate of applied N in a 3-year experiment on sandy loam soils in Pope County. Nitrate-N losses averaged 13, 27, and 61 lb/A for the 0, 80, and 160 lb/A annual rates, respectively. Doubling the N rate increased corn yields by 17% (24 bu/A) and increased the fertilizer derived NO_3-N leaching load by a factor of 3.4 times.

This study clearly demonstrated the difficulties in attempting to identify the fertilizer contribution within the N leaching loss component. Fertilizer-derived leaching losses were three times greater when determined by a N balance technique than by isotope-ratio analysis. Despite conditions (irrigated coarse-textured soils) conducive for leaching, the bulk of leaching from a single application was not observed until two to three years later.

Effects of N rates (0, 100, and 200 lb/A) on NO_3 solution concentrations, leaching losses, and root zone accumulations were studied under optimally irrigated corn in southeast North Dakota (Montgomery, 1984). Soil solutions were collected directly below the root zone of a Maddock sandy loam during a 5-year period with in situ trough extractors and ceramic suction cups. Leaching losses averaged 2, 5, and 26 lb/A/yr and growing season (flow-weighted) NO_3 concentrations averaged 4, 8, and 36 mg/L for the respective N rates. The 100 lb/A rate had no significant effect on leaching losses or concentrations in comparison to the check but improved corn yields by 195%. Only 2% of the applied N (100 lb rate) was lost to leaching. This data strongly supported the concept that under proper management, near maximum yields can be obtained without posing an environmental threat.

Prunty and Montgomery (1991) reported the results from a 4-year lysimeter study under similar conditions described above. This data supported earlier work (Montgomery, 1984) indicating that optimal corn production (90 to 95% of maximum yields) could be achieved without creating ground water recharge in appreciable excess of 10 mg/L. Annual leaching losses were 19.6 and 20.9 lb/A for the 85 and 130 lb/A rates, respectively (Montgomery et al., 1990). It is interesting to note that for this particular combination of soils, climate and management that treatment effects due to N rates were not observed until 325 to 500 days after the initial application. These findings appear harmonious with Walters and Malzer. Flow-weighted NO_3 concentrations during this time period (325-500 days after application) were 8.6 and 12.3 mg/L for the 85 and 130 lb/A rates, respectively. Although both the concentration and flux were statistically different, the overall net effect was minor due to the low percolation values during that time interval.

Rates (160 and 240 lb/A) and timing of application were evaluated on a Sverdrup sandy loam in central Minnesota (Gerwing et al., 1979). Soil solutions were collected at 5 and 8' with suction cup extractors. Initial concentrations for both rates were approximately 12 mg/L. Concentrations for single applications peaked at 55 and 80 mg/L, respectively, under the 160 and 240 lb/A rates. Concentrations appeared to return to initial levels during the late fall. Irrigation amounts appeared to be excessive (17" plus an additional 14" of rainfall) and may explain the elevated concentrations. Splitting the applications significantly decreased the NO_3 levels.

Research strongly suggests that proper N selection and management are important tools for maximizing N use efficiency. Irrigation can be an effective tool for minimizing leaching losses in many situations. Irrigation is commonly beneficial in increasing N uptake during critical uptake periods (V-12 to R-1 growth stages). Efficiency of water stressed plants in taking up N is substantially less. Therefore under irrigated conditions, it is not uncommon to have a significant reduction in soil nitrates at the end of the cropping season. Oberle and Kenney (1990) in a Wisconsin study determined that at N rates normally required to maximize corn yields, crop fertilizer N recovery was greater on irrigated loamy sand soils than rainfed silt loam soils. Hahne et al. (1977) found that residual soil NO_3 were significantly reduced in a variety of soil textures under irrigated conditions. Albus[21], in a two year study, found that residual soil NO_3 were not correlated to N rates (100, 150, and 200 lb/A of available N) when careful irrigation/N management was employed. For the respective N rates, yields averaged 136, 170, and 187 bu/A and N plant uptake averaged 94, 125, and 155 lb/A. Similar findings can be found elsewhere (Liang et al, 1991; Montgomery, 1984).

Summary of the Effects of N Rates on Ground Water Quality

Leaching losses are highly dependent upon the amount of N left in the soil profile at the end of the cropping season. Under most cropping/climatic conditions of Minnesota, the majority of the leaching losses take place during the non-cropping season. It is imperative that the amount of available N within the soil profile is minimal at harvest time.

Leaching losses can be greatly minimized by not exceeding the crop's physiological need for N. This need or "threshold value" is highly dependent upon soil moisture availability. Where water is not the limiting factor, corn will commonly consume 150 to 200 lb/A of available N. Commercial fertilizer is required to make up the difference between the plant's needs and other sources of N (residual N, mineralized N, etc.). Long-term, field specific yield information will aid the grower in selecting the correct N rate.

Leaching losses are commonly curvilinear after the threshold value for a given crop is exceeded. Leaching losses and carry-over N can be significantly minimized by not exceeding this value. In corn production, it appears that the balance between N use efficiency and yield falls somewhere between 90 to 95% of the maximum yield.

21. Personal Communication, Walt Albus, Oakes Irrigation Field Trials, Oakes, N.

The amount of available N, regardless of its source (commercial fertilizer, manure, legume, or residual soil NO_3), is clearly the single most important factor affecting leaching losses.

Within the specialty crops such as potatoes, it is not clear within the current literature what level of yield reduction would be required to keep leaching losses at an acceptable level.

Nitrate leaching losses will never be completely eliminated, even with complete elimination of all nitrogenous fertilizers. Concentrations of tile-drainage waters from the high organic matter soils typical of southern Minnesota ranged from 13 to 16 mg/L where no additional N was applied.

EFFECTS OF NITROGEN TIMING FOR REDUCING NITRATE LEACHING LOSSES

Introduction

Ideally the closer a producer can match N applications to the dynamic physiological needs of the crop, better yields and higher the N use efficiency will be obtained leaving minimal amounts of N to be leached. In actuality, recommended timing strategies vary throughout the state's variable climatic and soil conditions. Fall-applied N, for example, is a common strategy on many of the fine-textured soils. From 1985 through 1990, 30 to 45% of the state's fertilizer N has been sold in the fall (EPA, 1990). For the sake of simplicity, this portion of the discussion will be categorized in two broad soil textural groups.

Efficacy in Fine-Textured Soils

Randall et al. (1991) monitored the effects of timing on yield and tile-drained NO_3 leaching losses through Webster clay loams at Waseca. Primary experimental treatments were N applications (135 lb/A) applied in fall (with and without N-Serve), spring, and at sidedress (40% preplant and 60% sidedress). In 1990, flow-weighted NO_3-N concentrations were 30, 35, 27, and 28 mg/L and annual NO_3-N losses averaged 69, 109, 60, and 73 lb/a, respectively. Additional information during years with adequate drainage losses needs to be collected to fully understand the full treatment effects and biases due to lysimeter variability. Although this data indicates that about 50% more NO_3 is leached under fall fertilization, the losses are actually quite similar when the losses are "flow-normalized" or expressed as loss per unit of percolation.

In a similar investigation, Randall et al. (1991) also studied the effects of timing on silt loam soils of southeast Minnesota from 1987-1990. Corn yields and soil solutions (via suction cup extractors at 5 and 7') were monitored in response to a number of combinations of timing and addition of nitrification inhibitors. No yield responses due to splitting the applications were observed at any of the three sites yet it appeared that the lowest solution concentrations resulted from the single preplant application. Split applications may be positionally unavailable under dry soil moisture conditions in these fine-textured soils. At the Olmstead site, fall (with and without N-Serve) and spring (without N-Serve) were compared. Although over the 4-year yield average indicated no significant difference in spring versus fall applications, a yield increase in spring applied N of approximately 10 bu/A (statistically nonsignificant) in 1990 which had a wet spring.

Corn yields, as affected by N rates, sources, and timing, over a 29 year period have been reported from the Southwest Experiment Station (Fuchs and Nelson, 1990a). Although the statistical significance was not reported, it appears that timing of application is critical on these fine-textured soils only when the N supply is limiting production. At the 40 lb/A rate, yields were 85, 94, and 97 bu/A for the fall, spring, and sidedress applications, respectively. When N rates were increased to 80 or 160 lb/A, fall-applied yields were not significantly decreased. In the same location, Fuchs and Nelson (1990 b) investigated the efficacy of applying urea (120 lb/A rate as N) at mid-December, mid-February, and spring preplant. Respective corn yields during the 3-year study were 93, 99 and 101 bu/A. The fate of the some of the N from the fall/winter applications is uncertain but these researchers speculated that some volatilization and surface runoff occurred. Chalk et al. (1975) did not detect any yield differences between spring and fall-applied anhydrous ammonia applied to Wisconsin silt loam soils.

Efficacy in Coarse-Textured Soils

Effects of fall, spring, and sidedress (V6 growth stage) applications were studied on coarse-textured, nonirrigated soils at the University of Wisconsin-River Falls (Schmitt et al., 1990b). Although there were significant interactions between N rates and inhibitors (See Inhibitor Section), the overall conclusions were that fall applications were inferior to spring or sidedress applications. Corn yields (1989) obtained from the respective treatments were 130, 147, and 153 bu/A. Soil samples collected from the top 12" during the growing season indicated that NO_3 was limited in the fall applied plots suggesting that some losses did occur. Bauder and Montgomery (1979) determined that considerable overwinter leaching occurred on well-drained sandy loam soils of North Dakota and concluded that NO_3-based fertilizers should not be fall-applied. Overwinter NO_3-N accumulation from nitrification and mineralization (across sources) averaged 51 lb/A when N rates of 100 lb/A were applied.

Gerwing et al. (1979) determined that split applications were superior to a single preplant application in irrigated sandy loam soils of central Minnesota. Split applications at the recommended rate (160 lb/A) had only a minimal effect on the shallow aquifer (15′) but the one time application increased the concentrations from 7 to 10 mg/L. Yields were similar but the split application resulting in a better utilization of the applied material. In a follow up study utilizing the same plots and strategies as Gerwing, Buzicky (1982) demonstrated dramatic differences due to N timing. Yields from 0, 160 (preplant), and 160 lb/A (4-way split) plots were 44, 87, and 124 bu/A. Nitrogen recoveries of the applied fertilizer, as determined by ^{15}N techniques, were 38 and 60% from the single and split applications, respectively. Soil solutions, via suction cups at 2.5 and 5′, clearly indicated that the effects from the split application basically mimicked the check plots while the concentrations below the single application basically doubled during the cropping season. Based on ^{15}N analysis, 51% of the NO_3 at 5′ originated from the applied material in a single application, yet only trace amounts were detected under the split technique.

Under sweet corn production in loamy sands of central Wisconsin, Jung et al. (1972) identified the 5th through the 8th week after planting as the most effective time period for obtaining an optimum yield response.

Under Nebraska irrigated conditions, Watts and Martin (1981) showed that leaching losses were not affected by timing (preplant, sidedress, or fertigation) until seasonal drainage exceeded 5". Zubriski et al. (1983) observed an interaction between irrigation scheduling, rate, and timing of application on corn yields. Splitting the applications had little effect when a combination of conservative irrigation and rates were utilized. Splitting the application increased yields by 6% under optimum irrigation levels.

<u>Summary of the Effects of N Timing on Ground Water Quality</u>

Generally speaking, the practice of fall N applications on fine-textured soils does not necessarily pose a significant threat to ground water. Yield and leaching responses will be strongly influenced by climatic conditions. Denitrification is probably the major loss mechanism in these soils under wet conditions. Soils in southeast Minnesota are generally silt loams but due to the nature of the underlying limestone bedrock, fall applications are **not** recommended. See "Recommendations of the Nitrogen Task Force" (MDA, 1990) for additional details.

Sidedress applications in fine-textured soils can result in N which is positionally unavailable for maximized N use efficiency. In corn production it is recommended that N be applied as a preplant or in an early sidedress application.

Timing of N applications in coarse-textured soils is critical. Sidedressing, multiple applications, and fertigation are instrumental management tools in reducing NO_3 leaching losses. Their effectiveness is dependent on the amount of percolation during the growing season.

EFFECTS OF INHIBITORS ON GROUND WATER QUALITY

Introduction

The use of nitrification inhibitors (NI) has been incorporated into many existing N management plans. Nitrapyrin, commercially sold by DowElanco as N-SERVE[22], is the most common inhibitor sold in United States. N-SERVE is typically used in conjunction with anhydrous ammonia and to a lesser extent impregnated onto urea or into 28% liquids. N-SERVE is cleared for usage on corn, grain sorghum, wheat and cotton. Guardian[23] and Terrazole[24] are several of the other products which perform a similar role in N stabilization (Peterson and Frye, 1989).

Although there are numerous routes in which N can become unavailable to the crop, leaching and denitrification are two of the most prominent. Both of these loss mechanisms require that N to be in the NO_3 form. Inhibitors retard the nitrification process by the suppression of the Nitrosomonas bacteria which is responsible for the conversion of NH_4 to NO_2 (Hausenbuiller, 1972). Further oxidation of NO_2 to NO_3 is quickly converted by the Nitrobacter bacteria. Inhibitors would be most beneficial under the following conditions: coarse-textured soils, particularly under irrigation, where leaching would be the major loss mechanism; and under fine-textured soils when subject to very wet conditions. Denitrification would be dominant under the latter circumstance and could potentially occur in southern Minnesota, especially in the spring.

There is no doubt that these commercial NIs suppress the select genus of bacteria essential in the nitrification process. What is in question is the effectiveness in actually reducing N losses and the indirect yield response due to the extra N retained that otherwise would have been lost. Evaluation of inhibitors has been difficult and commonly inconsistent. The probability of getting a yield increase in a variety of soil types and timing conditions has been presented for Wisconsin conditions (See Table G-1).

Table G-1. Relative probability of increasing corn yields by using nitrification inhibitors. Taken from "Nutrient and Pesticide Best Management Practices for Wisconsin Farms" (Anonymous, 1989).

Soil Type	**** Time of Nitrogen Application **** Fall	Spring Preplant	Spring Sidedress
Sands and loamy sands	NR[25]	Good	Poor
Sandy loams and loams	Fair	Good	Poor
Silt loams and clay loams			
Well Drained	Fair	Fair	Poor
Somewhat Poorly Drained	Good	Good	Poor

22. Trademark name for nitrapyrin [2-chloro-6-(tricloro-methyl)pyridine].

23. Dicyanamide (DCD) is the technical nomenclature and is marketed by the Conkl Company, Inc., Shakopee, Mn.

24. Developed by Olin Corp., Little Rock, Ark and marketed by Uniroyal Chemical Company. Chemical formula is (5-Ethoxy-3-trichoro-methyl-1,2,4-thiadiazole). Also marketed by the tradename "Dwell".

25. Fall applications not recommended on these soils.

Numerous research efforts have been dedicated to NI effects since being introduced in the 1970's. Researchers have used a number of methods to do the evaluation. Studies involving crop response, analysis of residual soil N, bacteria counts, and direct monitoring of leaching or denitrification losses have been made. Effectiveness of NI's are very dependent on soils, climatic conditions, and numerous cultural practices. Estimates on yields are the easiest to make and are the most common within the literature. Many growers who previously used a NI commonly concluded that the product simply did not work because there was not an associated yield increase. What must be kept in mind is the fact that an event for N loss must occur before its benefits can be observed. Even if a N loss event occurred (leaching or denitrification), the effects of the NI could be masked if excess N is present. In other words, significant losses could occur yet enough N maybe present to satisfy crop needs. Generally yield increases have occurred when NI's were used under yield-limiting N rates (Hergert and Wiese, 1980).

Effects on Nitrification Inhibitors in Irrigated Agriculture

Effects of NI's have been highly variable under Minnesota's diverse soils and climatic conditions. The Westport lysimeter complex on the Rosholt farm in Pope County has provided a major contribution of the state's existing data on nitrification inhibitors. Soil type within the lysimeters is a Esterville sandy loam, typifying the irrigated soils in the surrounding area. Timmons (1984) evaluated the effectiveness of inhibitors using lysimeters and soil column experiments. Under irrigated corn over a 3-year period, NO_3-N leaching losses were reduced 10 lb/A or a 7% reduction in comparison to no inhibitor usage. The column study showed that N-SERVE or Terrazole were equally effective and most beneficial under conditions of excessive leaching.

In a later study, Walters and Malzer (1990b) examined interactions between N rates, inhibitor use, and fertilizer placement. Under conservative (80 lb/A) and excessive (160 lb/A) N rates, leaching losses were reduced 9 and 3%, respectively, when using N-SERVE. No significant differences in yield were found (Walters and Malzer, 1990a). The authors concluded that although inhibitors did reduce the potential for leaching, the overriding factors are selection of proper N rates and conservative water management. Effects of N rates and DCD were studied in a follow up study (Clay et al., 1989 and 1990). DCD was found to have no effect on N uptake and yield but late season NO_3-N concentrations were reduced 55%. Although not significant, it appeared that elevated concentrations peaked out the following fall due to the delayed leaching patterns resulting from the DCD.

Effects of Nitrification Inhibitors in Dryland Agriculture

Effects of N timing, rate, and use of inhibitors were studied on coarse-textured, non-irrigated soils at the University of Wisconsin-River Falls (Schmitt et al., 1990). The importance of either N-SERVE or DCD increased as available N became more limiting. Inhibitor use on fall-applied N was not nearly as effective as applying non-treated urea as a preplant or sidedress.

No yield differences, even in a wet spring such as 1990, were detected in a comparison of spring preplant (no N-SERVE) to fall-applied (with or without N-SERVE) in silt loam soils of southeast Minnesota (Randall et al., 1991). Losses under the 135 lb/A N rate (fall-applied), with and without N-SERVE, were 109 and 69 lb/A, respectively. Interestingly, leaching under soybeans rotated after the 1989 corn treatments lost 20% more NO_3 the next year where N-SERVE was applied.

Long-term effects of N-Serve on a 6-year Ohio lysimeter study (Owens, 1987) were studied under no-till corn receiving 300 lb/A/yr. Nitrate-N concentrations and annual leaching losses under treated urea were 23 mg/L and 105 lb/A, respectively. The same parameters under non-treated urea were 31 mg/L and 145 lb/A.

Bronson et al. (1991) examined the effectiveness of DCD in fall-applied N on winter wheat to split or single fall applications without DCD. Yields and ^{15}N recovery were not influenced by DCD the first year but immobilization of ^{15}N was 21, 20 and 15% with the fall N plus DCD, split applied N, and fall N, respectively. Differences were small but there was evidence that DCD and split applications did reduce the leaching component. Higher amounts of the ^{15}N were recovered in the second year's crop with DCD. No additional yield response in wheat was observed in northwest Minnesota with the addition of DCD in 1988-89 (Lamb et al., 1990). Residual NO_3 tended to be lower when rates exceeded 80 lb/A with DCD.

Addition of NIs to stabilize manure has been evaluated. Pryor (1988) summarized a number of German studies which found that DCD significantly altered the nitrification process in manure and commercial N fertilizers. In contrast, Comfort et al. (1988) did not find a response to N-SERVE when applied with injected liquid manure.

Summary of the Efficacy of Nitrification Inhibitors

Effects of nitrification inhibitors have been highly variable under Minnesota's diverse soils and climatic conditions. Nitrogen losses are commonly inferred by crop response. Other than the Westport site, few long-term studies actually quantifying reductions in leaching losses have been performed.

Under irrigated, coarse-textured soils, researchers have found that NI's have reduced the potential for leaching. Yet factors such as selection of proper rates and efficient irrigation management overshadow the differences that NI's can make.

Effects of NI's are most likely to be observed in yield performances when N is limiting.

Under conditions where high percolation of soil water (generally limited to coarse-textured soils) or soils prone to extended saturated conditions (generally fine-textured soils), the use of nitrification inhibitors should be encouraged.

Nitrification inhibitors can, under specific conditions, increase leaching losses by keeping the N positionally unavailable during the N uptake period.

Specific recommendations for the use of NI's are given for each region of the state in the "Recommendations of the Nitrogen Task Force".

EFFECTS OF TILLAGE ON GROUND WATER QUALITY

Introduction

In recent years, the effects of tillage has generated a large amount of research thrust. Moldboard plowing, the traditional method of tillage which still dominates over 65% of United States acreage (Thomas et al., 1989), has been identified as the principle source of many of the nation's eroded acres. Most of the earlier research focused on sediment movement and yield differences associated with variety of tillage operations.

Effects on water quality are not completely clear at this time. Plowing practices have greatly reduced direct channeling from the soil surface to the deeper portion of the vadose zone. These channels, or macropores, are naturally formed through soil formation and by soil dwelling organisms such as earthworms, other insects, and small mammals. As tillage practices evolve to no-till or other form of conservation tillage[26], the effects of these preferential flow paths becomes increasingly important. Through the macropores, there is rapid movement of water to specific depths while there is much slower water and chemical movement through the micropores. Evaporation losses are lower due to the mulch accumulations resulting in wetter soils under reduced tillage. Infiltration is also commonly increased as a result of more surface protection by residue which can prevent soil surface sealing. Residue can also serve as small damming structures which lengthen the residence time of surface water (Baker, 1987). As a net result of macropore flow, more infiltration, and the wetter soils, percolation through the soil profile is commonly increased.

Effectiveness of no-till or reduced tillage on surface runoff has been highly variable (Andraski et al., 1985). Factors enhancing infiltration can be offset by higher bulk densities in the upper portion of the profile commonly associated with reduced tillage. Higher bulk densities reduce porosity and hydraulic conductivity of the plow layer. The soil surface plays a critical role as a hydrologic interface. Other factors, such as surface roughness, pore size distribution, and stability play an important role in understanding tillage effects and must be considered a very dynamic process (Onstad and Voorhees, 1987).

Currently, Minnesota agriculture is dominated (75%) by moldboard tillage (Thomas et al., 1989). About 24% would be considered within the conservation tillage parameters with chisel plow being the most prevalent. Randall and Bandel (1987) describe the various tillage options in detail. Less than 1% of Minnesota's tillable land is under no-till.

Tillage Effects on Ground Water

Eight tile-drained lysimeters containing a Webster clay loam have been monitored over a 9-year study period to quantify the effects of tillage on NO_3 losses under non-irrigated corn in southern Minnesota (Randall and Anderson, 1991). No-till and moldboard plow were the tillage treatments and the resulting corresponding yields (9-yr average) were 122 and 137 bu/A, respectively. All plots were fertilized at the 180 lb/A/year rate. Annual percolation was

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26. Conservation tillage is defined as a tillage practice which results in a minimum of 30% surface coverage by residue (Mannering et al., 1987).

increased by 0.8" under no-till but NO_3-N concentrations were slightly lower (13.6 vs. 13.1 mg/L) than under the moldboard tillage treatment. As a result, no significant differences in NO_3 losses were found. Annual NO_3-N losses were 27.6 and 28.8 lb/A under moldboard and no-till, respectively. Higher total N amounts were found under no-till. Similar tendencies were found in Iowa: concentrations were lower under no-till but drainage amounts were higher resulting it no net difference in total amount of NO_3 loss (Kanwar et al., 1988). Differences were not noted until the third consecutive year of treatments.

In another related study in southern Minnesota, Randall (1990) compared the long-term effects of four tillage systems cropped in continuous corn. Tillage had a profound influence on NO_3 accumulation and distribution in the soil profile. Nitrate accumulations (top 5′) after harvest were 751, 546, 345, and 198 lb/A for the moldboard, chisel, disk and no-till systems, respectively.

Late fall NO_3 concentrations under grower operated conditions in Olmsted, Goodhue, and Winona Counties were also found to be commonly lower under no-till (Randall et al., 1991). Conversely, Bischoff et al. (1990) found that solution concentrations in the upper portion of the profile where higher under moldboard plow but lower in the deeper portion of the profile. Differences may be due to the better macropore development open to the surface on under no-till conditions. Overall drainage losses were not different.

Under irrigated conditions at the Westport lysimeter complex, flow weighted NO_3-N concentrations under no-till and roto-till methods were 12 and 19 mg/L, respectively (Vivekanandan et al., 1991). Tillage effects on yield and ground water may have been magnified due to the previous crop of soybeans.

Through the use of lysimeters and a small Ohio watershed, Dick et al. (1986) determined that leaching volumes were 2.3 times higher under no-till than under conventional tillage on silt loam soils. Lower evaporation losses and the development of continuous macropores throughout the soil profile from earthworms were important factors in explaining the higher drainage through no-till soils. Chemical losses were not reported. Total water loss due to either percolation or surface runoff was 1.5 times higher under no-till. Effects of tillage were not immediate; this study indicated that it took a minimum of two years to observe a response. Length of time required to see a hydrologic change will be dependent upon such variables as soil type, mulch effects, and organism population growth.

Magette et al. (1989) summarized a Maryland modeling effort with CREAMS to examine NO_3 losses as a function of cropping systems and soil type. These data seem to indicate that tillage was not nearly as an important factor for this particular setting as crop rotation, selection of yield goals, and rates. The importance of tillage on N losses maybe dependent upon generalized rainfall amounts. Randall and Bandel (1987) summarized that conservation tillage maybe useful under drier conditions associated with portions of the Great Plains. Reduction of leaching losses could be attributed to better water use efficiency and ultimately affect N use.

Tillage Effects on Surface Waters

There is agreement in the literature that conservation tillage does reduce the volume of surface runoff. Gilliam and Hoyt (1987) summarized several studies concluding that 20 to 25% reductions were reasonable while Dick et al. (1986) measured differences as high as 50% reductions under no-till. Influences on N concentrations are somewhat complex. Due to the sediment reduction associated with most conservation tillage operations, it would appear logical that losses of tightly bound nutrients such as NH_4^+ and PO_4-P would be reduced. Yet most studies summarized by Baker and Laflen (1983) have found that dissolved and total N concentrations are actually higher under conservation tillage. Higher concentrations result from several factors: 1) most fertilizer N is surface applied without incorporation and; 2) soil N tends to be higher at the surface in conservation tillage systems.

The net effect of conservation tillage is generally less organic N and more soluble N losses with a overall net increase of total N. The absolute differences between any of the tillage systems is small and the expected average annual increase is probably only 2 to 4 lb/A (Gilliam and Hoyt, 1987).

Summary on Tillage Effects on Ground Water Quality

Conservation tillage will continue to grow in Minnesota due to conservation compliance, energy and time savings. No-till, the conservative tillage considered as conservation tillage, will probably not make up many acres in Minnesota.

Nitrogen management decisions such as rates and timings will have a much larger impact on water quality than method of tillage.

Percolation is higher under conservation tillage due to wetter soil profiles caused by mulching effect of the residue, more macropores, and possible reduction of surface runoff.

Nitrate concentrations are commonly less under reduced tillage but due to the increased percolation losses, the net leaching loads are commonly the same as conventional tillage practices.

The volume of surface runoff can be reduced as much as 20 to 50% in comparison to conventional tillage practices but N losses due to surface runoff under any type of tillage are generally minor in comparison to other avenues of loss.

EFFECTS OF IRRIGATION ON THE FATE OF APPLIED NITROGEN

Introduction

Although the interactions of irrigation, N, and yields are well documented, information regarding how various irrigation management practices affect ground water is sparse. Many wide scale (such as in Nebraska and California) and numerous localized ground water problems have been specifically linked to irrigation developments (Adriano et al., 1972: Branson et al., 1975: Ferguson, 1990; Muir et al., 1976; Spalding et al., 1988). Many of these problem areas have been associated with flood irrigation, poorly managed sprinkler applications, and/or specialty crop production.

Irrigated acreage in Minnesota during 1985 was approximately 300,000 acres (Young and Woods, 1987). System types commonly used in the state (center pivot, traveling gun, solid set, and set moves) are capable of uniform water applications and not plagued with many of the hazards commonly associated with flood irrigation. Only wild rice production, which accounts for around 20,000 acres in northern Minnesota, is reliant upon flooding. Corn (53%), soybeans (14%), wild rice (7%) and alfalfa (7%) account for a bulk of the total irrigated acres. Potato acreage (6.4%) is particularly significant because of the traditionally high fertilizer and water inputs. Remaining acres are dominated by canning crops, dry beans and sugar beets.

Most of Minnesota's irrigation development is over the surficial outwash aquifers (Wright, 1989). The overlying soils, commonly coarse textured, respond extremely well under irrigation with 100% yield increases would not be considered uncommon. Rehm et al. (1991) reported that irrigated corn yields are commonly five times higher than under dryland conditions on the coarse sands of north central portion of the state. A bulk of Minnesota irrigation is in the general area called the "Central Sands" stretching east to west across the middle-third of the state. Future irrigation development on Minnesota's fine-textured soils does not appear to be economically feasible (Johnson et al., 1987).

General Status of Water Quality under Minnesota Irrigation

Wide-scale effects from irrigation on the quality of the states' water resources are not well documented. A limited number of Minnesota studies have found elevated NO_3 levels under irrigated coarse-textured areas. Anderson (1989) monitored 56 wells over a 2-year period in the surficial sand-plain aquifers of west-central Minnesota. Paired comparisons of similar soils under non-irrigated and irrigated conditions were made. Management strategies such as crop type and fertilizer rates were not documented. Mean ground water NO_3-N concentrations were 6 and 17 mg/L for the non-irrigated and irrigated, respectively, and differences were significant at the 95% confidence interval.

Myette (1984) monitored the NO_3 status of over 100 wells within the sand plain aquifers of Hubbard, Morrison, Otter Tail, and Wadena counties during a 3-year period. Nitrate concentrations were 9.6 and 1.7 mg/L from downgradient and upgradient irrigated areas, respectively. Sampling depth within the aquifer had a profound effect on the NO_3 results. Samples taken downgradient of the Staples

Irrigation Center at depths of 4, 8, and 15 feet below the surface of the aquifer resulted in NO_3-N concentrations of 15, 3, and 2 mg/L, respectively. Magner et al. (1990) monitored 15 wells located in surficial outwash sands of a 2 $mile^2$ area in Stearns County. Twenty percent of the wells exceeded 10 mg/L but the relationship between irrigation (and its associated higher N rates) and elevated NO_3 levels was not clear.

Studies done in neighboring states under similar hydrogeology have also investigated the effects of irrigation. Forty percent of the irrigation wells in the outwash sands of central Wisconsin were found to exceed 10 mg/L (Saffigna and Kenney, 1977b). Irrigated crop land was concluded as the major source of N in the ground water. Tile drain lines in southeast North Dakota from twenty five fields were monitored under a variety of crops and management practices (Montgomery et al.,1988). Fertilizer N rates averaged 184 and 65 lb/A, respectively, for the irrigated and dryland sites. The resulting NO_3-N concentrations were three times higher under irrigated conditions (irrigated and dryland concentrations were 8.3 and 2.5 mg/L, respectively) but individual management strategies appeared to dominate the likelihood of elevated NO_3 occurring.

Current Management Strategies

Criteria for determining frequency and application amounts (Bauder and Montgomery, 1980; Russelle et al., 1981) and timing based on plant physiology (Stegman, 1986) are important considerations in effective irrigation management. Deficit scheduling has been shown to effectively reduce drainage losses. In this method, the soil profile is only partially filled during an irrigation event which allows storage in the event of a rainfall. Researchers have also determined that some crops can be subjected to more water stress during various growth stages without significantly reducing yields. This could result in both a water savings for the grower as well as reduced leaching risks. Stegman (1986) compared the effectiveness of variable deficit scheduling to a full replenishment program. Stressing corn during the vegetative stages and carefully replacing water use during reproduction stages was found to be an effective strategy. Relationships indicated that 95% of the maximum yield could be attained by deficit scheduling and reduce seasonal water applications by 23 and 30% for coarse and medium textured soils, respectively. Effects on leachate components were not measured.

Bosch and Ross (1990) examined the economic issues associated with irrigation scheduling through a computer model called CRPSM. Irrigation scheduling, as opposed to the "guess" method commonly utilized by farmers, reduced water inputs from 11.7" to 5.5" without affecting yields. This research strongly supports the concept of substituting information and management time for water. Better scheduling increased average per-acre returns from $20 to $32.

Procedures have been described for an effective irrigation scheduling program, crop specific strategies for allowable soil water depletion, and methods for making soil water status measurements (Wright, 1989; Wright and Bergsrud, 1991). Various models and methods for accounting the soil water deficit during the growing season have been developed (Lundstrom and Stegman, 1988; Stegman et al., 1977). Some utilize computer software (Stegman and Coe, 1984; Anonymous, 1989).

Subjected to actual field testing under potato production, the Wisconsin Irrigation Scheduling Program (WISP) has effectively reduced irrigation rates by 12% (Curwen, 1989).

Wright[27] surveyed 121 irrigators in west-central Minnesota in 1982 and 1984 to determine current management programs and also to observe trends as a result of earlier educational efforts. Irrigators continued to rely heavily on basic climatic information (such as recent rainfall and number of rain-free days) and estimation of available water by hand probing. Few growers utilized the more advanced technology such as soil moisture indicators (tensiometers and blocks) and water balancing techniques ("checkbook" and computer driven programs).

Effects of Irrigation Management on Return Flow Characteristics

Increased NO_3 losses associated with irrigation are difficult to compare directly to dryland agriculture. Since a very strong interaction exists between N and water potential, most irrigators increase N inputs 50 to 100%. Average dryland and irrigated corn yields in the Central Sands[28] are 60 and 150 lb/A, respectively. Esser[29] determined in a 5-year survey covering 10,000 acres that dryland corn producers in southeast North Dakota applied 80 lb N/acre and the 5-year yield (1986-90) averaged 70 bu/A. Nitrogen inputs and yields were doubled under irrigation (158 lb/A and 137 bu/A).

Leaching losses, even under proper irrigation management, cannot be completely eliminated. Studies have attempted to differentiate natural percolation losses from those due to irrigation. Net increases in the drainage component found in midwestern coarse textured soils from dryland to irrigated conditions ranged from 1.6" (Timmons and Dylla, 1981; Watts and Martin, 1981) to an estimated 2.2" (Prunty and Montgomery, 1987). Drainage losses during the irrigation season (July-Sept.) were found to be 23% of the annual water balance (Montgomery et al., 1988) when irrigation scheduling techniques were followed.

Due to the popularity of corn in irrigated agriculture, most of the data regarding return flows is collected under this crop. Effects of two irrigation schemes on corn were studied in west central Minnesota using non-weighing lysimeters (Timmons and Dylla, 1981) over a 5-year period. Supplemental irrigations were applied as either a partial (1") or full replenishment (2") when 50% of the available water was depleted in an Esterville sandy loam. Even for irrigated conditions, the N applications were excessive in this experiment (232 lb/A). Average annual NO_3-N losses were 62, 72, and 100 lb/A for the non-irrigated, partial, and full replacement levels, respectively. Concentrations and N losses under this particular dryland treatment were somewhat meaningless due to the excessive N rate. It is worth noting that the yields were increased substantially going from dryland to conservative irrigation yet leachate

27. Irrigation management survey results from Pomme de Terre and Chippewa River Valleys, April, 1988. Jerry Wright, Area Extension Agricultural Engineer, Morris,Mn.

28. Personal communication with Jerry Wright, Area Extension Agricultural Engineer, Morris, MN.

29. Irrigation Advisor, Garrison Diversion Irrigation Project, Oakes, N.D. Personal Communication.

differences were statistically nonsignificant. Nitrate losses, when applying larger water applications, were increased 30% over the partial replacement with no increase in yield. Flow-weighted NO_3-N concentrations were 93, 70, and 51 mg/L for the non-irrigated, partial, and full replenishment treatments. In this study the concentration and NO_3 leaching losses were inversely proportional, demonstrating the danger of basing environmental risk on concentrations alone. Assessments should be based on mass emission (loading per unit area) whenever possible.

In another Minnesota study, Gerwing et al. (1979) concluded that the effects from irrigated corn in a Sverdup sandy loam were minor when reasonable N rates were applied with a split application. The authors felt that the 160 lb/A rate was a reasonable compromise between maximum yield and minimal ground water problems. No changes in NO_3 levels were observed. When the same rate was applied in a single application, the ground water concentration increased by 7 mg/L.

Hergert (1986) evaluated the effects of two irrigation schemes on leaching losses through a fine sand in Nebraska. Irrigation rates were 85% (slight deficit) and 130% (leaching irrigation) of the evapotranspiration. No significant differences from the 0.85 and 1.30% treatments were detected in yields or NO_3 concentrations (65 and 63 mg/L, respectively) but NO_3-N loss differences were significant (54 and 100 lb/A, respectively).

Effects of irrigation strategies described by Stegman (1986) on corn were evaluated over 4-year study in southeast North Dakota (Montgomery et al., 1990; Prunty and Montgomery, 1991). Soils and climatic conditions are similar to those found in the western section of the Minnesota "Central Sands". Effects of two irrigation management schemes (fixed 40% depletion vs. variable depletion based on crop phonology) and two N rates (85 and 130 lb/A/yr) were examined using large drainage lysimeters. Irrigation scheduling strongly influenced leachate losses. Variable depletion scheduling required 25% less water, had no effect on yield, and reduced NO_3-N losses by 30% in comparison to the fixed depletion. Effects of the additional 45 lb/A N increased yields by 15% (15 Bu/A) but did not significantly affect the leachate quality. Optimal N rates, proper timing, and sound irrigation management were responsible for the high N recovery in the plant and minimal amounts left in the soil profile.

Potatoes present a difficult challenge in irrigation management due to this crop's shallow root system and high N and water use. In an Idaho study, potatoes were fertilized with 300 lb N/A, then irrigated at rates equivalent to 1.0, 1.2, or 1.4 times the estimated ET (Stark et al., 1991). The 1.2 and 1.4 ET irrigation treatments increased the amount of NO_3-N leached below the rootzone by 30 and 124 lb/A, respectively. The hilling process also causes some problems due to channelized flow patterns and uneven root growth (Annadale et al., 1991) making efficient irrigation scheduling difficult.

Efficacy of Fertigation

Fertigation, the process by which N is applied through the irrigation water, has been identified as a beneficial management tool under certain conditions. One authority estimates that 40% of the nation's irrigators apply either N or agricultural chemicals through their systems (Schepers and Hay, 1987). Wright[30] found that 93% of the Minnesota irrigators in the west-central survey group had

30. Irrigation management survey results from Pomme de Terre and Chippewa River Valleys, April, 1988. Jerry Wright, Area Extension Agricultural Engineer, Morris, Mn.

the equipment to inject liquid N and that 73% indicated that they were actually fertigating. Seasonal amounts of N applied through the systems were categorized in three groups: 10-30 lb/A (22%); 30-60 lb/A (55%), and 60-90 lb/A (22%).

Several advantages of fertigation are: the grower can apply small increments of N, basically "spoonfeeding" the crop as its needs for N change; the amount of N in the NO_3 form is limited at any point in time thus reducing the leaching potential; and the grower has more flexibility in the N management. The grower can afford to be conservative with early N applications and then elect to later increase the N status based on plant appearance or favorable growing conditions.

The practice of fertigation has been under intense public criticism due to the potential non-point and point sources of contamination. Associated risks and benefits have been discussed by Schepers and Hay (1987). A major concern with fertigation is potential back-siphoning problems. Liquid N could be directly deposited into ground water caused by an unexpected shut down of the pumping equipment. Appropriate safety equipment will be required by 1994 for all Minnesota fertigators as a result of the 1989 Ground water Protection Act (18C). The MDA is currently drafting rules for fertigation.

A limited number of studies have investigated the yield and environmental ramifications of fertigation. In a coarse-textured Georgia soil, Gascho et al. (1984) compared the efficiency of conventional sidedress to fertigation and also a combination of the two methods. Based on yield and N use efficiency, the combination of the two application methods was superior. The highest yields attained in a Nebraska study were with sidedress applications in conjunction with supplemental N through the irrigation system (Rehm and Wiese, 1975).

Effects of injecting N through the irrigation system (4 times yearly) were compared to preplant applications under two irrigation application rates (Timmons and Dylla, 1981). Under deficit scheduling, no difference in NO_3 leaching losses were found. Fertigation reduced losses by about 10% when full replenishment irrigations were applied. Watts and Martin (1981) utilized a computer simulation model to assist in evaluating N and water management practices. Nitrogen losses were similar under either preplant anhydrous and fertigation provided that cropping season percolation losses did not exceed six inches. In years with high drainage losses, fertigation leaching losses were 13% less than preplant NH_3 applications. Fertigation, in conjunction with proper water management, was found to greatly reduce the potential for NO_3 losses under corn production in coarse soils (Smika and Watts, 1978).

These data indicate that fertigation can be a valuable tool for controlling NO_3 leaching losses when the opportunity for high drainage is present. Fertigation would also be beneficial where high soil variability across the field exists and where the grower does not have the skills to successfully schedule irrigations. Most studies conclude that a combination of preplant and fertigation or sidedress and fertigation is superior to any single method. University of Minnesota (Rehm et al., 1989) suggest two options for fertilizing corn on irrigated sandy soils. Both options include applying approximately one-third of the fertilizer N through the irrigation system. Growers should not rely upon fertigation for delivering more than one-third of the N required. In the event of a wet season, they would not want to be caught in the situation where irrigations have to be applied just to get adequate N to the crop.

Fertigation should be regarded as an essential practice for maintaining high productivity on Minnesota's environmental sensitive outwash plains. Yet the practice warrants a great deal of caution. Only irrigators who are confident of the delivery system's ability to apply uniform water applications should fertigate. In summary, fertigation must be carefully examined as a composite of processes and be addressed on individual soils, climate, and irrigation system/operator basis.

Summary of Irrigation Effects on Ground Water Quality

A number of state and national studies strongly indicates a correlation between irrigation development and NO_3 concentrations in ground water. There are a number of contributing factors including higher N rates, these sites are generally on coarse-textured soils, and increased leaching due to the additional water inputs. The 'cause and effect' relationship is poorly understood.

Irrigation, even on some of the coarse-textured soils of Minnesota, does not necessarily mean a significant increase in drainage under corn. Under irrigated corn, the grower has the opportunity to develop a healthy, well-rooted crop capable of utilizing high amounts of water and N. Several of the studies reviewed strongly suggested that additional percolation, due to irrigation when carefully applied, is minor in comparison to the entire hydrologic year. Information on other crops is lacking but it is speculated that increased drainage under some of the specialty crops could be considerably higher than under dryland conditions. Potatoes, for example, have a shallow root system and due to the hilling process, differential water flow pathways between the rows and the hills exist. Research data from grower-operated irrigated fields of any crop type is lacking. Most small plot research studies do not incorporate variabilities found under a pivot such as soil differences, sprinkler uniformity differences, etc.

Research studies from the Midwest also indicate that when reasonable rates of N are applied, the effects on the environment can be minimal. In general, when less than 150 to 180 lb/A of N is available (sum of fertilizer, residual, previous crop, etc.), corn is relatively efficient in recovering the N. Irrigation is good insurance that a healthy stand of plants will be developed.

Under realistic N rates the bulk of the leaching losses will occur during Minnesota's off-season recharge period, not during the irrigation season. Like dryland agriculture, it is crucial that the grower manages the N to maintain peak N use efficiency and minimize carry over nitrogen. It is residual N which is most prone to be lost below the root zone during the non-cropping season.

Irrigators need to be well educated in all facets of irrigation/N management. Efforts must be made to keep irrigation an asset rather than an environmental liability. The potential for environmental degradation under poor management is extremely high.

Nitrate concentrations should not be the only criteria used to establish the effectiveness of a management practice. A number of studies comparing irrigated to dryland conditions revealed that the concentrations traveling through the vadose zone can be higher under dryland conditions. This may be due to lower percolation values. Commonly the actual amount of NO_3 flux is higher under irrigation since it is the mathematical product of the concentration times the drainage component.

Keeping losses of N and other agricultural chemicals to an acceptable level may by extremely difficult in some of Minnesota's coarse-textured soils. Due to the low moisture holding capacity of soils textured as "sands" and "loamy sands", it is extremely difficult to schedule irrigations to satisfy crop needs yet minimize leaching losses. The cation exchange capacity (the soils ability to chemical hold nutrients) of these type of soils is typically very low which aggravates the problem.

Fertigation is a valuable tool for minimizing the amount of available N in the soil profile at any one time during the cropping season. Benefits of a fertigation will generally outweigh the risks when the proper safety equipment is utilized.

Literature Cited

Adriano, D.C., F.H. Takotori, P.F. Pratt and O.A. Lorenz. 1972. Soil nitrogen balance in selected row-crop sites in southern California. J. Environ. Qual. 1:279-283.

Anderson, H.W., Jr. 1989. Effects of agriculture on quality of water in surficial sand-plain aquifers in Douglas, Kandiyohi, Pope, and Stearns counties, Minnesota. Report 87-4040. U.S. Geological Survey, St. Paul, Mn.

Andraski, B.J., D.H. Mueller, and T.C. Daniels. 1985. Effects of tillage and rainfall simulation date on water and soil losses. Soil Sci. Soc. Am. J. 49:1512-1517.

Annadale, J.G., G.S. Campbell, D.J. Mulla, A.P. Mallawatantri, and C. Calissendorff. 1991. Two dimensional soil water and nitrate movement in center pivot irrigated potato hills and furrows. p. 212. In Agronomy abstracts. ASA, Madison, WI.

Anonymous. 1990. Progress report-survey of behavior and attitudes of dairy farmers in Beltrami County. Cooperative study between SWCD, BOSWR, Bemiji State University, and Headwater Regional Commission.

Anonymous. 1989. Nutrient and pesticide best management practices for Wisconsin farm. WDATCP Bulletin ARM-1.

Baker, J.L. 1987. Hydrologic effects of conservation tillage and their importance relative to water quality. In T.J. Logan et al. (ed.) Effects of conservation tillage on groundwater quality: Nitrates and pesticides. Lewis Publ., Inc., Chelsea, MI., p. 113-124.

Baker, J.L. and J.M. Laflen. 1983. Water quality consequences of conservation tillage. J. Soil and Water Cons. 38:186-193.

Baker, J.L. and H.P. Johnson. 1981. Nitrate-nitrogen in tile drainage as affected by fertilization. J. Environ. Qual. 10:510-522.

Baker, J.L. K.L. Campbell, H.P. Johnson, and J.J. Hanway. 1975. Nitrate, phosphorus, and sulfate in subsurface drainage water. J. Environ. Qual. 4:406-412.

Barbee, G.C. and K.W. Brown. 1986. Comparison between suction and free-drainage soil solution samplers. Soil Sci. 141:149-154.

Bauder, J.W. and B.R. Montgomery. 1979. Overwinter redistribution and leaching of fall-applied nitrogen. Soil Sci. Soc. Am. J. 43:744-747.

Bauder, J.W. and B.R. Montgomery. 1980. N-source and irrigation effects on nitrate leaching. Agron. J. 72:593-596.

Bergstrom, L. 1987. Nitrate leaching and drainage from annual and perennial crops in tile-drained plots and lysimeters. J. Environ. Qual. 16:11-18.

Berry, J.T. and N.L. Hargett. 1988. Fertilizer summary data 1988. ISSN: 0146-1850. National Fertilizer Development Center, TVA, Muscle Shoals, AL.

Binford, G.D., A.M. Blackmer, and N.M. El-Hout. 1990. Tissue test for excess nitrogen during corn production. Agron. J. 82:124-129.

Bischoff, J.H., A.R. Bender, and C.G. Carlson. 1990. The effects of no-till and moldboard plow tillage on the movement of nitrates and pesticides through the vadose zone. National Water Well Association Conference, Feb. 20-21, 1990. Kansas City, Missouri.

Blackmer, A.M., T.F. Morris, and G.D. Binford. 1991. Predicting N fertilizer needs for corn in humid regions: advances in Iowa. p. 359. Agronomy abstracts. ASA, Madison, WI.

Blackmer, A.M. 1989. Indices for adjusting fertilizer nitrogen rates. In Proceedings crop production and protection conference, December 18-19, 1989, Ames, Iowa.

Blackmer, A.M., D. Pottker, M.E. Cerrato and J. Webb. 1989. Correlations between soil nitrate concentrations in late spring and corn yields in Iowa. J. Prod. Agric. 2:103-109.

Blackmer, T., Schepers, J.S., and M.F. Vigil. 1991. Chlorophyll meter readings in corn as affected by the uniformity of plant stand. p. 283. In Agronomy abstracts. ASA, Madison, WI.

Bock, B.R., J.J. Meisinger, and K.R. Kelley. 1991. Predicting N fertilizer needs for corn in humid regions: summary and future directions. p. 359. In Agronomy abstracts. ASA, Madison, WI.

Bock, B.R. and G.W. Hergert. 1991. Fertilizer nitrogen management. In R.F. Follett, D.R. Kenney, and R.M. Cruse (ed) Managing nitrogen for groundwater quality and farm profitability. Soil Sci. Soc. of Amer., Madison, WI.

Bolton, E.F., J.W. Aylesworth, and F.R. Hore. 1970. Nutrient losses through tile drains under three cropping systems and two fertility levels on a Brookston clay loam. Can. J. Soil Sci. 50:275-279.

Bongard, P.M., T. Wagar, and R. Hamar. 1991. Corn yields as affected by nitrogen rate, legume, and manure histories. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Bosch, D.J. and B.B. Ross. 1990. Improving irrigation schedules to increase returns and reduce water use in humid regions. J. of Soil Water Conserv. 44:486-489.

Branson, R.L., P.F. Pratt, J.D. Rhoades, and J.D. Oster. 1975. Water quality in irrigated watersheds. J. Environ. Qual. 4:33-40.

Broadbent, F.E. and A.B. Carlton. 1980. Methodology for field trials with nitrogen-15-depleted nitrogen. J. Environ. Qual. 9:236-242.

Bronson, K.F., J.T. Touchton, R.D. Hauck, and K.R. Kelley. 1991. Nitrogen-15 recovery in winter wheat as affected by application timing and dicyandiamide. Soil Sci. Soc. Am. J. 55:130-135.

Brown, K.W., J.C. Thomas, and M.W. Aurelius. 1985. Collecting and testing barrel sized undisturbed soil monoliths. Soil Soc. Sci. Am. J. 49:1067-1069.

Brown, K.W., C.J. Gerard, B.W. Hipp, and J.T. Ritchie. 1974. A procedure for placing undisturbed monoliths in lysimeters. Soil Soc. Sci. Am. Proc. 38:981-983.

Bruulsema, T.L. and B.R. Christie. 1987. Nitrogen contribution to succeeding corn from alfalfa and red clover. Agron. J. 79:96-100.

Bundy, L.G. and E.S. Malone. 1988. Effect of residual profile nitrate on corn response to applied nitrogen. Soil Sci. Soc. Am. J. 52:1377-1383.

Burwell, R.E., G.E. Schuman, K.E. Saxton, and H.G. Heinemann. 1976. Nitrogen in subsurface discharge from agricultural watersheds. J. Environ. Qual. 5:325-329.

Buzicky, G.C. 1982. Nitrogen fertilizer distribution between corn plants, soil and aquifer under irrigated conditions. M.S. thesis, Univ. of MN, St. Paul, MN.

Cerrato, M.E. 1989. Sensitivity of various methods used to assess the nitrogen status of corn. Ph. D. dissertation. Iowa State University, Ames, Ia.

Chalk, P.M., D.R. Kenney, and L.M. Walsh. 1975. Crop recovery and nitrification of all and spring applied anhydrous ammonia. Agron. J. 67:33-37.

Chichester, F.W. 1977. Effects of increased fertilizer rates on nitrogen content of runoff and percolate from monolith lysimeters. J. Environ. Qual. 6:211-217.

Clay, D.E., G.L. Malzer, and J.L. Anderson. 1990. The impact of residual fertilizer treatments and tillage on soybean yields and NO_3 leaching. Misc. Publ. 62. Mn. Agric. Exp. Sta., Univ. of Mn. St. Paul, Mn.

Clay, D.E., G.L. Malzer, and J.L. Anderson. 1989. Tillage, N rate, and nitrification inhibitor influence on corn production and NO_3 leaching. Misc. Publ. 61. Mn. Agric. Exp. Sta., Univ. of Mn. St. Paul, Mn.

Cole, D.W. 1958. Alundum tension lysimeter. Soil Sci. 85:293-296.

Comfort, S.D., K.A. Kelling, D.R. Keeney, and J.C. Converse. 1988. The fate of nitrogen from injected liquid manure in a silt loam. J. Environ. Qual. 17:317-322.

Curwen, D. 1989. PCM-"On farm" demonstrations. The Badger Common'tater. March ed. p. 12-14.

Daberkow, S., L. Hansen, and H. Vroomen. 1988. Low input practices. Outlook No. 148. Econ. Res. Serv., U.S. Dept. Agr., Washington, D.C. pp. 22-25.

Dick, W., W.M. Edwards, and F. Haghiri. 1986. Water movement through soil to which no-tillage cropping practices have been continuously applied. In Proceedings of the agricultural impacts on groundwater-a conference. Aug. 11-13, 1986. Omaha, Nebraska. p.243-252.

Duke, H.R. and H.R. Haise. 1973. Vacuum extractors to assess deep percolation losses and chemical constituents of soil water. Soil Sci. Soc. Am. Proc. 37:963-964.

Edmisten, K.L., C.W. Wood, and D.W. Reeves. 1991. Predicting corn N needs with chlorophyll measurements. p. 286. In Agronomy abstracts. ASA, Madison, WI.

El-Hout, N.M. and A.M. Blackmer. 1990. Nitrogen status of corn after alfalfa in 29 Iowa fields. J. Soil and Water Conserv. 45:115-117.

Environmental Protection Agency. 1991. Nitrogen Action Plan (Draft). PM-221.

Environmental Protection Agency. 1990. County-level fertilizer sales data. PM-221.

Evans, S.D., P.R. Goodrich, R.C. Munter, and R.E. Smith. 1977. Effects of solid and liquid beef manure and liquid hog manure on soil characteristics and on growth, yield, and composition of corn. J. Environ. Qual.6:361-368.

Ferguson, R. 1990. Agricultural best management practices to minimize nitrate movement to groundwater. In 1990 North Dakota Water Quality Symposium Proceedings. P. 115-120. March 20-21, 1990, Fargo, N.D.

Follett, R.F.(Ed.). 1989. Nitrogen management and ground water protection. Developments in agricultural and managed-forest ecology 21. Elsevier Science Publication, New York.

Fox, R.H., G.W. Roath, K.V. Iversen, and W.P. Piekielek. 1989. Soil and tissue nitrate tests compared for predicting soil nitrogen availability to corn. Agron. J. 81:971-974.

Fuchs, D.J. and W.W. Nelson. 1990. Thirty years of field experimentation with nitrogen source, placement, and time of application to a Webster clay loam at the Southwest Experiment Station. 1990 Blue Book, Misc. Publ. 62. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn., p.44-45.

Fuchs, D.J. and W.W. Nelson. 1990. The retention of nitrogen fertilizer applied during the winter. 1990 Blue Book, Misc. Publ. 62. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn., p.52-53.

Gascho, G.J., J.E. Hook, and G.A. Mitchell. 1984. Sprinkler-applied and side-dressed nitrogen for irrigated corn grown on sand. Agron. J. 76:77-81.

Gast, R.G., W.W. Nelson, and G.W. Randall. 1978. Nitrate accumulation in soils and loss in tile drainage following nitrogen applications to continuous corn. J. Environ. Qual. 7:258-261.

Gerwing, J.R., A.C. Caldwell, and L.L. Goodroad. 1979. Fertilizer nitrogen distribution under irrigation between soil, plant, and aquifer. J.Environ. Qual.8:281-284.

Gilliam, J.W. and G.W. Hoyt. 1987. Effect of conservation tillage on fate and transport of nitrogen. In T.J. Logan et al. (ed.) Effects of conservation tillage on groundwater quality: Nitrates and pesticides. Lewis Publ., Inc., Chelsea, MI.p.-240.

Hahne, H.C.H., W. Kroontje, and J.A. Lutz, Jr. 1977. Nitrogen fertilization:I. Nitrate accumulation and losses under continuous corn cropping. Soil Sci. Soc. Am. J. 41:562-567.

Hallberg, G.R. 1989. Nitrate in groundwater in the United States. In R.F. Follett (ed) Nitrogen management and ground water protection. Elsevier, New York. p. 35-74.

Hallberg, G.R. 1987. Nitrate in Iowa groundwater. In F.M. D'Itri and Wolfson, L.G. (ed) Rural groundwater contamination. Lewis Publ., Inc., Chelsea, MI. p. 23-68.

Hansen, E.A. and A.R. Harris. 1975. Validity of soil-water samples collected with porous ceramic cups. Soil Sci. Soc. Proc. 39:528-536.

Hargett, N.L. and J.T. Berry. 1990. Commercial fertilizers. Tennessee Valley Authority, National Fertilizer & Environmental Research Center. Muscle Shoals, AL.

Harris, G.H. and O.B. Hesterman. 1990. Quantifying the nitrogen contribution from alfalfa to soil and two succeeding crops using nitrogen-15. Agron. J. 82:129-134.

Hausenbiller, R.L. 1972. Soil science:Principles and practices. Wm. Brown Co., Dubuque, Iowa.

Hergert, G.W. 1986. Nitrate leaching through sandy soil as affected by sprinkler irrigation management. J. Environ. Qual. 15:272-278.

Hergert, G.W. and R.A. Wiese. 1980. Performance of nitrification inhibitors in the Midwest (west). p. 89-105. In J.J. Meisinger et al. (ed) Nitrification inhibitors-Potentials and limitations. ASA Spec. Publ. 38. ASA and SSSA, Madison, WI.

Herron. G.M., G.L. Terman, A.F. Dreier, and R.A. Olson. 1968. Residual nitrate nitrogen in fertilized deep loess-derived soils. Agron. J. 60:477-482.

Hesterman, O.B., M.P. Russelle, C.C. Sheaffer, and G.H. Heichel. 1987. Nitrogen utilization from fertilizer and legume residues in legume-corn rotations. Agron. J. 79:726-731.

Hesterman, O.B., C.C. Sheaffer, D.K. Barnes, W.E. Lueschen, and J.H. Ford. 1986. Alfalfa dry matter and nitrogen production, and fertilizer nitrogen response in legume-corn rotations. Agron. J. 78:19-23.

Houtsma, J. 1990. Cutting-edge system tests N on the go. Farmer Magazine, August 4, 1990. p. 24.

Inada, K. 1963. Studies on a method for determining the deepness of green color and chlorophyll content of intact crop leaves and its practical applications. I. Principle for estimating the deepness of green color and chlorophyll content of whole leaves. Proc. Crop. Sci. Soc. Jpn. 32:157-162.

Iowa State Extension Service. 1990. Estimating nitrogen needs by soil testing. Ames, Iowa.

Jackson, W.A., L.E. Asmussen, E.W. Hauser, and A.W. White. 1979. Nitrate in surface and subsurface flow from a small agricultural watershed. J. Environ. Qual. 2:480-482.

Johnson, B.S., G.R. Blake, and W.W. Nelson. 1987. Midseason soil water recharge for corn in the northwestern corn belt. Agron. J. 79:661-667.

Joshi, J.R., J.B. Swan, J.F. Moncrief, and P. Buford. 1991. Effect of tillage and frequency of liquid dairy manure application on corn production: A long-term study. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn. p. 269-275.

Joshi, J.R., J.B. Swan, J.F. Moncrief, and G.L. Malzer. 1991. Effect of tillage and frequency of liquid dairy manure application on the availability of N to corn, soil N distribution and N concentration in soil water. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.p. 276-290.

Jung, P.E., Jr., L.A. Peterson, and L.E. Schrader. 1972. Response of irrigated corn to time, rate, and source of applied N on sandy soils. Agron. J. 64:668-670.

Kaap, J.D., 1986. Implementing best management practices to reduce nitrate levels in northeast Iowa groundwater. In Proceedings of the Agricultural Impacts on Ground Water-A conference. p. 412-427, August 11-13, 1986, Omaha, Neb.

Kapp, J.D. 1986. Implementing best management practices to reduce nitrate levels in northeast Iowa groundwater.

Kanwar, R.S., J.L. Baker, and D.G. Baker. 1988. Tillage and split N-fertilization effects on subsurface drainage water quality and crop yields. TRANS. ASAE 31:453-461.

Klausner, S.D. and R.W. Guest. 1981. Influence of NH_3 conservation from dairy manure on the yield of corn. Agron. J. 73:720-723.

Lamb, J.A., T.E. Cymbaluk, and P. Nesse. 1990. Comparison of spring-applied DCD-ureas and urea of hard red spring wheat in northwestern Minnesota. Misc. Publ. 62. Mn. Agric. Exp. Sta., Univ. of Mn. St. Paul, Mn.

Lane, M. 1991. Soil nitrate tests gain in accuracy and respectability. Solutions. Mar/Apr.

Legg, T., W. Lazarus, R. Levins and M. Schmitt. 1990. Reducing nitrogen applications to manured corn: An opportunity to save money and protect the environment. Report P90-28. Univ. of Minn.-Dept. of Agric. and Applied Economics, St. Paul, Mn.

Legg, T.D., J.J. Fletcher, and K.W. Easter. 1989. Nitrogen budgets and economic efficiency: A case study of southeastern Minnesota. J. Prod. Agric. 2:110-116.

Liang, B.C., M. Remillard, and A.F. MacKenzie. 1991. Influence of fertilizer, irrigation, and non-growing season precipitation on soil nitrate-nitrogen under corn. J. Environ. Qual. 20:123-128.

Linville, K.W. and G.E. Smith. 1971. Nitrate content of soil cores from corn plots after repeated nitrogen fertilization. Soil Sci. 112:249-255.

Lockyer, D.R., B.F. Pain, and J.V. Klarenbeek. 1989. Ammonia emissions from cattle, pig, and poultry wastes applied to pasture. Environ. Pollut. 56:19-30.

Lory, J.A., G.W, Randall, and M.P. Russelle. 1991. N credits for manure and alfalfa. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Lundstrom, D.R. and E.C. Stegman. 1988. Irrigation scheduling by the checkbook method. AE-792. North Dakota State Univ. Ext. Service. Fargo, ND.

MacGregor, J.M., G.R. Blake, and S.D. Evans. 1974. Mineral nitrogen movement into subsoils following continued annual fertilization for corn. Soil Sci. Soc. Amer. Proc. 38:110-113.

Magdoff, F.R., D. Ross, and J. Amadon. 1984. A soil test for nitrogen availability to corn. Soil Sci. Soc. Am. J. 48:1301-1304.

Magette, W.L., R.A. Weismiller, J.S. Angle, and R.B. Brinsfield. 1989. A nitrate groundwater standard for the 1990 farm bill. J. of Soil and Water Cons. p. 491-494.

Magner, J.A., D.B. Wall, and N.E. Wilson. 1990. Agricultural and lake influences on water quality in outwash aquifers in the School Sections Lake area of Stearns County, Minnesota. Minnesota Pollution Control Agency, St. Paul, Mn.

Masterson, S.D., D.D. Francis, M.F. Vigil, and J.S. Schepers. 1991. Chlorophyll production and nitrogen use patterns of 10 irrigated corn hybrids grown at 3 nitrogen rates. p. 294. In Agronomy abstracts. ASA, Madison, WI.

Meisinger, J.J., V.A. Bandel, and J.S. Angle. 1991. Evaluation of the pre-sidedress nitrate soil test in Maryland. p. 294. In Agronomy abstracts. ASA, Madison, WI.

Midwest Plan Service. 1985. Livestock waste facilities handbook. Iowa State Univ., Ames, Ia.

Mielke, L.M. and J.R. Ellis. 1976. Nitrogen in soil cores and groundwater under abandoned cattle feedlots. J. Environ. Qual. 5:71-75.

Minnesota Agricultural Statistics Service. 1990. Minnesota Agriculture Statistics, St. Paul, Mn.

Minnesota Department of Agriculture (MDA). 1990. Recommendations of the nitrogen fertilizer task force on the nitrogen fertilizer management plan. St. Paul, Mn.

Moncrief, J.F., M.J. Wiens, and J.J. Kuznia. 1991a. Tillage effects on N available to irrigated corn from alfalfa and urea N sources. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Moncrief, J.F., M.J. Wiens, and J.J. Kuznia. 1991b. Tillage effects on N available to irrigated corn from turkey manure and urea N sources. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn. p. 117-127.

Montgomery, B. 1984. In situ extractors of quantifying drainage and nitrate losses under irrigated corn. M.S. Thesis. North Dakota State University, Fargo.

Montgomery, B.R., L. Prunty, A.E. Mathison, E.C. Stegman, and W. Albus. 1988. Nitrate and pesticide concentrations in shallow ground water aquifers underlying coarse textured soils of S.E. North Dakota. In Proceedings of the Agricultural Impacts on Ground Water-A Conference. p. 361-387, March 21-23. Des Moines, Ia.

Montgomery, B.R., L. Prunty and E.C. Stegman. 1990. Influence of irrigation and nitrogen management on nitrate leaching losses. In 1990 North Dakota Water Quality Symposium Proceedings. P. 95-114. March 20-21, 1990, Fargo, N.D.

Montgomery, B.R., L. Prunty, and J.W. Bauder. 1987. Vacuum trough extractors for measuring drainage and nitrate flux through sand soils. Soil Sci. Soc. Am. J. 51:271-276.

Morris, T.F., A.M. Blackmer, and N.M. El-Hout. 1991. Optimal rates of nitrogen fertilization for first-year corn after alfalfa. p. 295. In Agronomy abstracts. ASA, Madison, WI.

Motavalli, P.P., K.A. Kelling, and J.C. Converse. 1989. First-year nutrient availability from injected dairy manure. J. Environ. Qual. 18:180-185.

Muir, J., J.S. Boyce, E.C. Seim, P.N. Moasher, E.J. Diebert, and R.A. Olson. 1976. Influence of crop management practices on nutrient movement below the root zone in Nebraska soils. J. Environ. Qual. 5:255-259.

Murdock, L.W. 1991. The Soil Doctor-a field trial. In Proceedings of the Twenty First North Central Extension-Industry Soil Fertility Conference. St. Loius, MO. Potash and Phosphate Institute, Manhattan, Kansas. p. 127-133.

Myette, C.F. 1984. Ground-water-quality appraisal of sand-plain aquifers in Hubbard, Morrison, Otter Tail, and Wadena Counties, Minnesota. U.S. Geological Survey, Water Resource Investigations Report 84-4080. St. Paul, MN.

National Agricultural Statistics Service. 1991. Agricultural chemical usage: 1990 field crops summary. United States Department of Agriculture, Economic Research Service, Washington, D.C.

O'Leary, M., G. Rehm, and M. Schmitt. 1989. Providing proper N credit for legumes. AG-FO-3769. Minnesota Extension Service, St. Paul, Mn.

Oberle, S.L. and D.R. Keeney. 1990. Factors influencing corn fertilizer N requirements in the Northern U.S. corn belt. J. Prod. Agric. 3:527-534.

Onstad, C.A. and W.B. Voorhees. 1987. Hydrologic soil parameters affected by tillage. In T.J. Logan et al. (ed.) Effects of conservation tillage on groundwater quality: Nitrates and pesticides. Lewis Publ., Inc., Chelsea, MI. p. 95-112.

Owens, L.B. 1990. Nitrate-nitrogen concentrations in percolate from lysimeters planted to a legume-grass mixture. J. Environ. Qual. 19:131-135.

Owens, L.B. 1987. Nitrate leaching losses from monolith lysimeters as influenced by nitrapyrin. J.Environ. Qual. 16:34-38.

Pain, B.F., R.B. Thompson, Y.J. Rees, and J.H. Skinner. 1990. Reducing gaseous losses of nitrogen from cattle slurry applied to grassland by the use of additives. J. Sci. Food Agric. 50:141-153.

Peterson, T.A. and M.P. Russelle. 1991. Alfalfa and the nitrogen cycle in the Corn Belt. J. Soil Water Conserv. pp. 229-235.

Peterson, G.A., and W.W. Frye. 1989. Fertilizer nitrogen management. In R.F. Follett (ed) Nitrogen management and ground water protection. p. 183-220 Elsevier Publ., New York, New York.

Pionke, H.B. and J.B. Urban. 1985. Effect of agricultural land use on ground water quality in a small Pennsylvania watershed. Groundwater 23: 68-80.

Pocock, J. 1991. Put your money where it counts. Prairie farmer. Jan.

Prunty, L. and B.R. Montgomery. 1987. Lysimeter measurement of groundwater recharge and quality under irrigation. In Yu-Si fok (ed.) Infiltration development and application. Proceedings of the International Conference on Infiltration Development and Application. Honolulu, Hawaii. p. 311-318.

Prunty, L. and B.R. Montgomery. 1991. Lysimeter study of nitrogen fertilizer and irrigation rates on quality of recharge water and corn yield. J. Environ. Qual. 20:373-380.

Pryor, G.R. 1988. The use of dicyandiamide to reduce leaching loss of applied nitrogen fertilizers. In Proceedings of agricultural impacts on ground water-a conference, March 21-23, 1988, Des Moines, Iowa.

Randall, G., J. Anderson, G. Malzer, B. Anderson, D. Buhler, D. Wyse, and J. Nieber. 1991. Impact of nitrogen and tillage management practices on corn yield and potential groundwater contamination in southeastern Minnesota. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Randall, G.W., G.C. Buzicky, and W.W. Nelson. 1986. Nitrate losses to the environment as affected by nitrogen management. In Proceedings of the Agricultural impacts on ground water-a conference. p. 561-573. National Water Well Association, Dublin, OH.

Randall, G. and B.W. Anderson. 1991. Nitrogen loss to tile lines as affected by tillage. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Randall, G.W. 1990. Nitrate-N in the soil profile and tile drainage water as influenced by tillage. Am. J. of Industrial Medicine 18:457-460.

Randall, G.W., R.H. Anderson, and P.R. Goodrich. XXXX. Soil properties and future crop production as affected by maximum rates of dairy manure. Proceedings of the Third International Symposium of Livestock Wastes. Am. Soc. of Ag. Eng., St. Joseph, MI.

Randall, G., J. Anderson, G. Malzer, D. Wyse, J. Nieber, B. Anderson, and D. Buhler. 1990. Impact of nitrogen and tillage management practices on corn yield and potential groundwater contamination in southeastern Minnesota. Misc. Publ. 62. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn. p. 105-109.

Randall, G.W. and V.A. Bandel. 1987. Overview of nitrogen management for conservation tillage systems:An overview. In T.J. Logan et al. (ed.) Effects of conservation tillage on groundwater quality: Nitrates and pesticides. Lewis Publ., Inc., Chelsea, MI., p. 39-63.

Randall, G., G.L. Malzer, B.W. Anderson. 1991. Nitrate losses to tile drainage as affected by nitrogen fertilization of corn in a corn-soybean rotation. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ. of Mn. St. Paul, Mn.

Rehm, G.W., G.L. Malzer, and J.A. Wright. 1989. Managing nitrogen for corn production on irrigated sandy soils. AG-FO-2392. Minnesota Extension Service, St. Paul, Mn.

Rehm, G., E. Oelke, M. Weins, and D. Schmitz. 1991. Comparing barley and corn production under irrigated and dryland conditions in north-central Minnesota. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Rehm, G., D. Schmitz, and A. Scobbie. 1991. Nitrogen value of red clover grown on a sandy soil. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Rehm, G. and M. Schmitt. 1991. Soil N test to improve corn N recommendation. Soil samplings. Vol. 38, May. Mn. Ext. Ser., St. Paul, Mn.

Rehm, G.W. and R.A. Wiese. 1975. Effect of method of nitrogen application on corn (Zea mays L.) grown on irrigated sandy soils. Soil Sci. Soc. Amer. Proc. 39:1217-1220.

Ritchie, S.W. and J.J. Hanway. 1984. How a corn plant develops. Spec. Rep. 48 (revised). Iowa State University, Ames, Iowa.

Robbins, C.W. and D.L. Carter. 1980. Nitrate-nitrogen leached below the root zone during and following alfalfa. J. Environ. Qual. 9:447-450.

Roth, G.W. and R.H. Fox. 1990. Soil nitrate accumulations following nitrogen-fertilized corn in Pennsylvania. J. Environ. Qual. 19:243-248.

Russelle, M.P., and W.L. Hargrove. 1989. Cropping Systems: Ecology and management. In R.F. Follett (ed) Nitrogen management and ground water protection. (Fill in rest). p. 277-317.

Russelle, M.P., E.J. Deibert, R.D. Hauck, M. Stevenovic, and R.A. Olson. 1981. Effects of water and nitrogen management of yield and ^{15}N-depleted fertilizer-use efficiency of irrigated corn. Soil Sci. Soc. Am. J. 45:553-558.

Saffigna, P.G. and D.R. Keeney. 1977a. Nitrogen and chloride uptake by irrigated Russet Burbank potatoes. Agron. J. 69:258-264.

Saffigna, P.G. and D.R. Keeney. 1977b. Nitrate and chloride in groundwater under irrigated agriculture in central Wisconsin. Groundwater 15:170-177.

Sawyer, J.H. and R.G. Hoeft. 1990. Effect of injected liquid beef manure on soil chemical properties and corn root distribution. J. Prod. Agric. 3:50-55.

Schepers, J.S., M.G. Moravak, E.E. Alberts, and K.D. Franks. 1991. Maize production impacts on groundwater quality. J. Environ. Qual. 20:12-16.

Schepers, J.S., D.D. Francis, M.F. Vigil, and T. Blackmer. 1991. Use of chlorophyll meters to evaluate N status of corn. p. 300. In Agronomy abstracts. ASA, Madison, WI.

Schepers, J.S. and R.H. Fox. 1989. Estimation of N budgets for crops. In E.F. Follett (ed) Nitrogen management and groundwater protection. Elsevier, New York. p. 221-246.

Schepers, J.S. and D.R. Hay. 1987. Impacts of chemigation on groundwater contamination. In F.D'Itri and L. Wolfson (ed.) Rural groundwater contamination. Lewis Pub., Chelsea, Mi. pp.105-114.

Schmitt, M.A., G.W. Randall, G.W. Rehm, and G.L. Malzer. 1990a. Corn production and nitrogen use: are soil tests improving N recommendations? Soils, fertilizer and agricultural pesticides short course. December 12-13, 1990. Minneapolis, Mn.

Schmitt, M.A. and R.G. Hoeft. 1986. Manure injection can sometimes injure corn crop. Crops Soils 38(8):19-22.

Schmitt, M.A., R.H. Beck, and W. Connolly. 1990b. Urea management--rate and time of application and nitrification inhibitors--for corn production. Misc. Publ. 62. Mn. Agric. Exp. Sta., Univ. of Mn. St. Paul, Mn.

Schmitt, M.A. 1989. Manure management in Minnesota. AG-FO-3553. Mn. Ext. Ser., Univ. of Mn., St. Paul, Mn.

Schmitt, M. G. Rehm, and G. Malzer. 1989. Nitrification inhibitors and use in Minnesota. AG-FO-3774. Mn. Ext. Ser., Univ. of Mn., St. Paul, Mn.

Schuman, G.E. and L.F. Elliot. 1978. Cropping an abandoned feedlot to prevent deep percolation of nitrate-nitrogen. Soil Sci. 126:237-243.

Schuman, G.E., T.M. McCall, K.E. Saxton, and H.T. Knox. 1975. Nitrate movement and its distribution in the soil profile of differentially fertilized corn watersheds. Soil Sci. Soc. Am. Proc. 39:1192-1196.

Sims, J.T. 1986. Nitrogen transformations in a poultry manure amended soil: Temperature and moisture effects. J. Environ. Qual. 15:59-63.

Smika, D.E. and D.G. Watts. 1978. Residual nitrate-N in fine sand as influenced by N fertilizer and water management practices. Soil Sci. Soc. Am. J. 42:923-926.

Smith, J.H. and J.R. Peterson. 1982. Recycling of nitrogen through land application of agricultural, food processing, and municipal wastes. In F.J. Stevenson (ed) Nitrogen in agricultural soils, Agronomy 22: 791-831.

Sommer, S.G. and J.E. Olesen. 1991. Effects of dry matter content and temperature on ammonia loss from surface-applied cattle slurry. J. Environ. Qual. 20:679-683.

Spalding, R.F., J.R. Gormly, B.H. Curtiss, and M.E. Exner. 1978. Nonpoint nitrate contamination of groundwater in Merrick County, Nebraska. Ground Water 16:86-95.

Spalding, R.F., A.P. Bryda, and L.A. Kitchen. 1988. Intermediate vadose zone nitrate. In Proceedings of the Agricultural Impacts on Ground Water-A Conference. p. 635-647, March 21-23. Des Moines, Ia.

Stark, J.C., I.R. McCann, T.A. Tindall, and D.T. Westermann. 1991. Minimizing nitrate leaching potential in irrigated potato cropping systems. p. 342. In Agronomy abstracts. ASA, Madison, WI.

Stegman, E.C. and D.A. Coe. 1984. Irrigation scheduling software for microcomputers. ASAE 84-301. St. Joseph, Mi.

Stegman, E.C., A. Bauer, J.C. Zubriski, and J.W. Bauder. 1977. Crop curves for water balance irrigation scheduling in S.E. North Dakota. N.D. Research Report No. 66. North Dakota State Univ., Fargo, N.D.

Stegman, E.C. 1986. Efficient irrigation timing methods for corn production. Trans ASAE. 29:203-210.

Stevenson, F.J. (Ed.) 1982. Nitrogen in agricultural soils. Agronomy 22. Am. Soc. of Agron., Madison, WI.

Sutton, A.L., D.W. Nelson, D.T. Kelley, and D.L. Hill. 1986. Comparison of solid vs liquid dairy manure applications on corn yield and soil composition. J. Environ. Qual. 15:370-375.

Sutton, A.L., D.W. Nelson, and D.D. Jones. 1985. Utilization of animal manure as fertilizer. AG-FO-2613. Mn. Ext. Ser., Univ. of Mn., St. Paul, Mn. Adopted from Purdue Univ. Publ. ID-101.

Thomas, G.W., S. Smith, and R.E. Phillips. 1989. Impact of soil management practices on nitrogen leaching. In R.F. Follett (ed) Nitrogen management and groundwater protection. Elsevier, New York. p. 247-276.

Thompsen, R.B., J.C. Ryden, and D.R. Lockyer. 1987. Fate of nitrogen in cattle slurry following surface application or injection to grassland. J. Soil Sci. 38:689-700.

Timmons, D.R. and A.S. Dylla. 1981. Nitrogen leaching as influenced by nitrogen management and supplemental irrigation level. J. Environ. Qual. 10:421-426.

Timmons, D.R. 1984. Nitrate leaching as influenced by water application levels and nitrification inhibitors. J. Environ. Qual. 13:305-309.

Turner, F.T. and M.F. Jund. 1991. Chlorophyll meter to predict nitrogen topdress requirement for semidwarf rice. Agron. J. 83:926-928.

TVA. 1989. Soil nitrate testing workshop: Research and extension needs in humid regions of the United States. Circular Z-250. Workshop Feb. 8 and 9, 1989, Muscle Shoals, Al.

United States Department of Commerce. 1987. Census of Agriculture. AC87-A-23. Geographic Area Series.

Vivekanandan, M., G.L. Malzer, and J.L. Anderson. 1991. Residual effects of nitrogen fertilizer treatments and tillage on grain yield, uptake and NO_3 leaching in irrigated corn. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Wagar, T., G. Cremers, A. Scobbie, and G. Rehm. 1989. Nitrogen rates for corn production in southeast Minnesota. Misc. Publ. ??. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Wagner, G.H. 1962. Use of porous ceramic cups to sample soil water within the profile. Soil Sci. 94:379-386.

Walters, D.T. and G.L. Malzer. 1990a. Nitrogen management and nitrification inhibitor effects on nitrogen-15 urea: I. Yield and fertilizer use efficiency. Soil Sci. Soc. Am J. 54:115-122.

Walters, D.T. and G.L. Malzer. 1990b. Nitrogen management and nitrification inhibitor effects on nitrogen-15 urea: II. Nitrogen leaching and balance. Soil Sci. Soc. Am J. 54:122-130.

Watts, D.G. and D.L. Martin. 1981. Effects of water and nitrogen management of nitrate leaching losses from sands. Trans. ASAE. 24:911-916.

Wright, J.W. 1989. Irrigation water management considerations for sandy soils in Minnesota. AG-FO-3875. Minnesota Extension Service, St. Paul, Mn.

Young, P.G. and S. C. Woods. 1987. Water use in Minnesota agriculture. Minnesota Dept. of Natural Resources.

Zadak, C.G., G.W. Randall, and M.P. Russelle. 1989. Residual soil nitrate following alfalfa as influenced by tillage and corn hybrid. Misc. Publ. ??. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Zubriski, J.C., R.A. Utter, and W. Albus. 1983. Corn, water level, nitrogen rate, and time of application. Oakes Field Trials, 14th Annual Report.

FEEDLOTS

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INTRODUCTION

An adequate supply of healthy livestock, poultry, and other animals is essential to the well-being of Minnesota citizens and the nation. However, domestic animal manure may have a negative effect on Minnesota's environment when improperly stored, transported or utilized.

The focus of this chapter is on ground water nitrogen associated with feedlots. A feedlot is defined as a lot or building or combination of lots and buildings intended for the confined feeding, breeding, raising, or holding of animals. Feedlot operations also include facilities for manure storage. Field application of manure is discussed in Chapter G.

Based on the number of livestock operations in Minnesota during 1990 (Table H-1) there is likely between 45,000 and 60,000 dairy, beef, swine, and poultry feedlots (> 10 animal units) in Minnesota. The amount of nitrogen (N) and other nutrients produced by different types of livestock are listed in Table H-2. A 1,000 pound dairy animal produces about 150 pounds of N per year and 1000 pounds of broilers produce about 438 pounds of N per year. Some of this N will be lost through ammonia volatilization (See manure section of Crop Production chapter). The estimated amount of manure N generated in Minnesota is equivalent to the amount of excreted waste N generated by about 77 million people, over 17 times the number of people in Minnesota (Table H-3).

Nine of the ten largest livestock operations permitted by the MPCA are poultry operations. The highest concentrations of manure production are in central, southeastern and southwestern Minnesota (See manure application section in Chapter G).

Table H-1. Minnesota Number of Livestock Operations[1] (From 1991 Minnesota Agricultural Statistics Service).

Year	Cattle	Milk Cows	Beef Cows	Hogs	Sheep
1986	48,000	21,000	17,500	18,000	4,800
1987	44,000	18,500	16,000	16,500	4,800
1988	43,000	17,500	16,000	16,000	4,800
1989	43,000	16,500	16,000	16,300	5,000
1990	40,000	15,500	15,000	15,000	5,200

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[1] An operation is any farm having one or more head of livestock on hand at any time during the year.

Table H-2 Typical Manure Volumes and Content

Animal	Size lb	Total manure production lb/day	ft^3/day	gal/day	BOD5 lb/day	Nutrient content, lb/day N	P_2O_5	K_2O
Dairy								
Cattle	150	12	0.19	1.5	0.26	0.06	0.023	0.048
	250	20	0.32	2.4	0.43	0.10	0.045	0.084
	500	41	0.66	5.0	0.86	0.20	0.082	0.169
	1,000	82	1.32	9.9	1.70	0.41	0.166	0.325
	1,400	115	1.85	13.9	2.38	0.57	0.232	0.458
Beef								
Cattle	500	30	0.50	3.8	0.8	0.17	0.127	0.145
	750	45	0.75	5.6	1.2	0.26	0.191	0.229
	1,000	60	1.00	7.5	1.6	0.34	0.250	0.289
	1,250	75	1.20	9.4	2.0	0.43	0.318	0.373
Cow*		63	1.05	7.9	1.7	0.36	0.273	0.313
Swine								
Nursery pig	35	2.3	0.038	0.27	0.07	0.016	0.0118	0.012
Growing pig	65	4.2	0.070	0.48	0.13	0.029	0.0223	0.024
Finishing pig	150	9.8	0.16	1.13	0.30	0.068	0.050	0.054
	200	13.0	0.22	1.5	0.39	0.090	0.068	0.071
Gestating sow	275	8.9	0.15	1.1	0.27	0.062	0.048	0.048
Sow & Litter	375	33.0	0.54	4.0	1.00	0.230	0.173	0.181
Boar	350	11.0	0.19	1.4	0.35	0.078	0.059	0.061
Sheep	100	4.0	0.062	0.46	0.09	0.045	0.015	0.039
Poultry								
Layers	4	0.21	0.0035	0.027	0.014	0.0029	0.0025	0.0014
Broilers	2	0.14	0.0024	0.018	0.0023	0.0024	0.00123	0.0009
Turkeys	16	--	--	--	--	0.0096	--	--
Horse	1,100	45	0.75	5.63	--	0.27	0.105	0.205

Source: Midwest Planning Service - 18

Table H-3. Number of Livestock and Human Nitrogen Population Equivalent* in Minnesota (From 1991 Agricultural Statistics).

Livestock Type	Head	Animal Units	T. Nitrogen Produced Annually (Tons)	*Estimated Human Nitrogen Population Equivalent (P.E.)
Dairy Cattle	1,005,000	1,160,000	84,371	24,106,000
Beef Cattle	1,180,000	1,180,000	73,220	20,920,000
Calves	575,000	230,000	15,739	4,497,000
Swine	4,500,000	900,000	57,488	16,425,000
Sheep	300,000	30,000	2,464	704,000
Layer Hens	10,200,000	51,000	5,397	1,542,000
Broiler Hens	41,300,000**	23,000	3,283	938,000
Turkeys	46,300,000**	278,000	27,038	7,725,000
Total		3,852,000	269,000	76,857,000

*The nitrogen human population equivalent was calculated by assuming excreted human waste of 7 lbs N/person/year and nitrogen contents as stated in Table H-1. Assumed 5.5 batches of broiler hens and 3 batches of turkeys per year. Horse manure is not included in the above table.

**Total number of birds produced in 1990.

Several studies have implicated feedlot areas as being sources of ground water nitrogen pollution (Egboka, 1984; Tjostem et al., 1977; Ritter and Chirnside, 1987). There are a number of potential pathways for nitrogen to move from feedlot areas to ground water, including:

1) leaching directly below outdoor animal holding areas;
2) runoff and subsequent infiltration;
3) runoff into unused wells, improperly sealed wells and sinkholes;
4) runoff to surface water that eventually discharges into ground water;
5) leaching below manure storage areas;
6) over application of manure to fields (often near the feedlot);
7) leaching below abandoned feedlots; and
8) leaching from burial pits or composting of dead animals.

The focus of this chapter is on ground water nitrogen contamination and best management practices associated with outdoor animal holding areas, abandoned feedlots and earthen manure storage basins. An overview of the MPCA feedlot program is also presented.

OUTDOOR ANIMAL HOLDING AREAS

Ground Water Nitrogen Impacts

Outdoor confined animal holding areas are generally used for beef cattle feedlots and dairy feeding and resting areas. Nitrogen impact results from studies of soil cores and ground water near outdoor animal holding areas are quite varied. Several investigators have stated that an intact manure pack covering an earth-surfaced feedlot with continual use at a sufficient stocking rate will provide an effective soil seal, preventing most nitrogen movement through the soil profile (Norstadt and Duke, 1985; Mielke and Mazurak, 1976; Mielke, et al., 1974). Mielke, et al. (1974) stated that the texture of the soil profiles under the feedlots appeared to have little effect on the water movement into the profile. Ellis, et al. (1975) examined 129 soil cores taken from 15 eastern Nebraska feedlots and concluded that most feedlots did not constitute a nitrogen leaching problem. Where problems existed, the NO_3 contamination was localized. Feedlots less than five years old had about the same total nitrogen in soil cores as older feedlots. Feedlots that were used intermittently had higher NO_3 than those used more than 10 months out of the year. Norstadt and Duke (1985) concluded from existing studies that stocking rate, manure pack maintenance and amount of use throughout the year are the primary factors influencing the effectiveness of the soil seal in a feedlot. Norstadt and Duke, however, did find high NH_4 concentrations in the soil directly below an "effective" seal. Lack of aeration and water infiltration prevented nitrification and subsequent leaching of NO_3.

Elliot, et al. (1972) found that a level feedlot in an area of silt loam soil changing to sand with depth contributes no more NO_3, NH_4, or total nitrogen to the shallow water table beneath it than an adjacent cropped field. One of the reasons believed for the minor impacts was denitrification occurring beneath the soil surface (following nitrification during summer months). A low redox potential and high amounts of organic carbon in the soil below the feedlot provided conditions favorable for denitrification. Data from a three-year period showed that the shallow water table beneath a beef cattle feedlot contained more than 10 mg/l NO_3-N in only two instances (Elliot and McCalla, 1972). Very little NO_3 movement and moderate NH_4 movement was found by Mielke, et al. (1970) below a Nebraska cattle feedlot. A low redox indicated that denitrification was also possible at this site. Very little manure had been removed at this site and a deep organic cover had been built up.

The soil seal in a feedlot can be broken if the manure shrinks and cracks while drying. The following conditions will contribute to cracks in the seal: 1) incomplete covering of manure pack, 2) too low stocking rate (< 40 cattle per acre), 3) complete removal of manure and prolonged exposure of soil surface when cleaning, and 4) seasonal use or abandoned feedlots (Norstadt and Duke, 1985). Gillham and Webber (1969) calculated a 4.4 pound loss of nitrogen to ground water under a feedlot holding 65 head of cattle during five months of study. While this loss was enough to greatly affect ground water quality, it was very small compared to the total manure-N produced. In the top 20 feet of soil under beef cattle feedlots, Stewart, et al. (1967) found an average of 1436 pounds of NO_3-N per acre in 47 soil core profiles. This amount was 17 times greater than native grassland or alfalfa.

It is also possible to have very low NO_3-N under a feedlot but a build-up of NH_4 (Schuman and McCalla, 1975). Ammonium is much less likely to leach to ground water than NO_3, but it could eventually be converted to NO_3 and later affect ground water. Therefore, high NH_4 in soil below feedlots should still be viewed as a potential threat to ground water.

Three unpaved, annually-cleaned cattle feedlots in southern Alberta were studied (Sommerfeldt, et al., 1973). Ground water below one of the feedlots remained below 3 mg/l in all wells. At another site ground water NO_3-N exceeded 10 mg/l during a few months out of the year. At the third site NO_3-N exceeded 10 mg/l all year adjacent to the lot and exceeded 10 mg/l during four months at a point nearly one-third of a mile away. Some elevated NO_3 in soils near the feedlots indicated that runoff from the feedlot to adjacent soils could be contributing to the ground water NO_3. Terry et al. (1981) also found runoff from a feedlot to have an impact on ground water. While relatively few studies have examined feedlot surface runoff impacts on ground water, this pathway is perhaps contributing more nitrogen to ground water than leaching directly below outdoor animal holding areas. Runoff from feedlots can infiltrate in adjacent soils, move into sinkholes, unsealed wells, or surface waters.

Best Management Practices - Outdoor Animal Holding Areas

To minimize ground water nitrogen impacts from outdoor animal holding areas, the following best management practices are recommended:

- Surface soil should be left relatively undisturbed when removing manure from the feedlot.

- Temporary holding facilities (used less than 10 months/year) should be constructed with impervious surfaces.

- All practices necessary to prevent surface runoff of manure should be implemented. These practices include clean water diversions, construction of catchment basins, and vegetative filter strips (in accordance with SCS standards). Installing adequate gutters and drain pipes around all basins and machine sheds is also recommended.

- Closely calibrate the nutrient content of feed to the nutritional needs of livestock in order to reduce nitrogen and phosphorous in feed, and subsequently in manure.

ABANDONED FEEDLOTS

Ground Water Nitrogen Impacts

The number of feedlots abandoned each year is unknown and variable. From 1980 to 1990 there was a drop in the number of Minnesota cattle operations from 62,000 to 40,000 (Minnesota Agricultural Statistics Service, 1991). The number of swine and milk cow operations dropped nearly in half over this same period. Based on this information, there are likely thousands of feedlots that have been abandoned over the last decade in Minnesota.

After a feedlot is abandoned, a great potential exists for the organic-N and NH_4 built up over the years in the soil surface to be eventually converted to NO_3. The soil seal will eventually be destroyed due to freeze/thaw and wetting/drying cycles.

Ellis et al. (1975) and Mielke and Ellis (1976) reported the results of soil coring and ground water nitrogen under abandoned beef cattle feedlots in Nebraska. Ground water NO_3-N concentrations under two feedlots abandoned for six and four years were 43.7 and 77.2 mg/l, respectively. Nitrate in soil under abandoned feedlots was 5.6 times higher on average than below active feedlots. However, not all abandoned sites had high NO_3 levels. One site cropped to alfalfa-corn rotation for 15 years had low NO_3. Another site with low NO_3 had been intermittently used for 30 years before it was abandoned.

Schuman and Elliot (1978) reported a study on an abandoned feedlot cropped part to corn and part to alfalfa. From plant N uptake and soil analysis, the study concluded that alfalfa removed considerably more nitrogen from the soil profile than did corn or the control plot. While the soil profile under alfalfa (with topsoil removed) did not exceed 10 mg/l NO_3-N below the 2.75 meter depth, significant amounts of NO_3-N appeared to have been leached below the corn treatments. Alfalfa has been shown in other studies to effectively reduce NO_3-N levels in soil profiles (Mathers et al. 1975). Although alfalfa has the ability to convert atmospheric nitrogen to NO_3, it will utilize NO_3 from the soil first. Mathers et al. (1975) stated that 270 lb/acre or more NO_3-N removal was possible with alfalfa, and since alfalfa frequently roots to depths over 20 feet, it could remove nitrogen found in deep soils far below abandoned feedlots.

Best Management Practices - Abandoned Feedlots

- Alfalfa and/or other nitrogen scavenging crops should be planted on abandoned feedlot soils. Soil nitrogen tests should be conducted periodically until nitrogen levels are similar to nearby cropland.

- Scraping the upper layer of soil from abandoned feedlots and redistributing this nitrogen rich soil over a large area should be considered.

MANURE STORAGE

Overview

Manure can be handled in a solid, semi-solid, or liquid form. The form of manure will influence the type of manure storage that may be used. Liquid manure can be stored in the following: below-ground lined pits; earthen storage basins (lined or unlined), and above ground tanks. Semi-solid manure is often kept in drained storages. Where manure is dried sufficiently or where bedding is added to make it a stackable, solid manure handling is an option. Solid manure can be stored in a walled structure outside, piled on concrete pads or the ground, and in composting piles. Lagoons are rarely used in Minnesota to pretreat swine wastes. The type of collection system installed will depend on the type of livestock, soil type, geology and the size of operation.

Since the early 1970's, the trend in Minnesota farms has been to include a manure and/or runoff storage structure as part of manure management systems. The stored wastes are then applied to cropland. Manure storage gives the farmer more options for better overall management of nutrient resources. If no storage or inadequate storage is used, it is likely that manure application will occur during times of the year when crop uptake of nutrients is negligible. In a modeling exercise, Heatwole et al. (1990) estimated that 30 percent less nitrogen would be leached in a conventionally-tilled field if four to six month manure storage is utilized compared to no storage.

Earthen facilities used for the storage and treatment of manure and wastewater are the waste management system components that present the greatest threat to ground water quality (Krider, 1987). A number of investigators have monitored soil and ground water in an attempt to better understand potential ground water impacts from earthen basins and lagoons. Conflicting results have been found regarding the adequacy of animal waste materials to seal the soil-waste interface in storage ponds and treatment lagoons.

Ground Water Nitrogen Impacts from Manure Storage

Municipal wastewater treatment ponds are allowed to leak 500 gal/acre/day, with typical nitrogen concentrations ranging from 10 to 30 mg/l nitrogen in the wastewater. Since nitrogen concentrations from manure in earthen basins have been measured to range between 500 and 3500 mg/l, it takes much less leakage to have an impact on soil and ground water below the manure ponds. Fortunately, manure has some ability to seal the bottoms of storage ponds. Effective sealing of soils have been found in a number of studies (Barrington et al., 1986; Gangbazo et al., 1989; Roswell et al., 1985; Miller et al., 1985). The sealing of storage basins has been found to depend on the ability of the soil to retain the manure solids at its surface and to trap them within its surface pores (Barrington et al., 1987b; Roswell et al., 1985). Biological and chemical sealing are less important mechanisms of sealing. Barrington et al. (1987b) stated that soil void geometry, particle size distribution, and dimension were more important than soil hydraulic conductivity in the ability to seal. Clay content was also found to somewhat influence the sealing of earthen basins.

With coarse textured soils, the length of time required for seal development will be greater, allowing manure to move into soil and ground water while the seal is developing (Roswell, et al., 1985; Miller, et al., 1985). Miller, et al. (1985) found that it took 12 weeks before a storage basin sealed itself in a coarse textured sand. Results from other studies indicate that sealing is never complete in some earthen basins and that nitrogen movement out of the basins is occurring (Ritter and Chirnside, 1983 and 1987; Egboka, 1984; Dalen, et al., 1983; Phillips, et al., 1983; Culley and Phillips, 1989; Miller, et al., 1976; Ritter, et al., 1984). An earthen basin holding dairy manure constructed with four layers of polyethylene liner lost 24 percent less nitrogen over a five-year period than three unlined basins constructed in clay loam, sandy loam, and sandy soils (Culley and Phillips, 1989). The clay loam and sandy loam soils retained seven percent more nitrogen than the sandy soils did. In this study, 33 percent of all nitrogen was lost to the atmosphere.

Earthen basin sideslopes undergo many more wetting/drying cycles than basin bottoms and thus are more likely to crack. Leakage may occur through these cracks upon subsequent filling of the basin.

Dalen, et al. (1983) evaluated seepage rates and ground water impacts from two manure storage ponds in southeastern Minnesota. Water was collected and analyzed from monitoring wells, lysimeters, and tile line water. Estimated seepage rates from a pond constructed in a sandy clay soil were 40 to 300 gallons per day of liquid having total nitrogen concentrations between 1200 and 1500 mg/l. Despite this seepage, only minor impacts were found in the ground water below the site. At the other site, liquid manure from a 225 head dairy operation was stored in a pond. Soil with the highest clay content was stockpiled during excavation of the pond and placed on the pond bottom and sideslopes. Rough estimates of seepage rates at this site were 70 to 540 gallons per day. The seepage front had reached a depth of at least five feet below the pond bottom after three years of operation. Total nitrogen concentrations at this depth were 30 to 40 mg/l in suction cup lysimeter water.

Soils under earthen basins constructed in fine-textured soils have shown elevated total nitrogen, but few studies have found excessive nitrogen in **ground water** below storage basins constructed in finer textured soils or with clay seals. When basins in fine-textured soils did leak, an explanation for the seal failure was given. In one case, observed migration of nitrogen was explained by the breakdown of the seal due to repeated spring time freezing and thawing (Gangbazo, et al. 1989). In another case, very high NH_4 and NO_3 was found in ground water below two swine lagoons that were emptied twice a year by the farmer. The emptying was believed to dry out and crack the clay lining seal, resulting in seepage when the liquid refilled the basin (Ritter and Chirnside, 1987).

Another possible explanation for relatively low nitrogen in ground water below leaking earthen basins is denitrification. Miller, et al. (1985) believed that some leakage occurred from a newly constructed basin, but that denitrification depleted the NO_3 from ground water flowing under the pond.

Concrete pits used to collect manure below slatted floors in barns or adjacent areas are less likely to allow seepage into ground water than are earthen basins. However, there have been instances in which an empty concrete pit has been floated out or cracked by external water pressure. Above ground storage tanks are used to store manure usually in areas of shallow depth to bedrock and high water tables. Manure is usually piped into the tank from the bottom of the storage tanks. Spills have resulted from valve failure in these tanks.

Composting is a method used to treat and store manure while reducing volume and weight and producing a stable humus product. Composted manure has advantages of reduced odor and easier transportation off-site. Some nitrogen volatilizes during composting. Most of the nitrogen in composted manure remains in the organic form, slowly releasing inorganic nitrogen once applied to cropland. More studies are needed to determine nitrogen impacts from manure composting piles and the advantages and disadvantages of composting manure.

Many operators in the state just pile solid manure without actually trying to compost the manure. Manure storage piles should be kept covered to prevent runoff or direct leaching from these sites.

Best Management Practices - Manure Storage

Most of the practices to minimize N movement from manure storage basins are actually siting and design criteria. A state-certified engineer or Soil Conservation Service employee should assist in the design, location and construction of manure management systems. MPCA currently requires all earthen basins to be designed by the Soil Conservation Service or a state-certified engineer. MPCA has recently drafted recommended design criteria for earthen basins to be used when construction is not supervised by the Soil Conservation Service.

THE MINNESOTA FEEDLOT PROGRAM

Policy for regulating pollution from animal feedlots is governed largely by Minnesota Pollution Control Agency (MPCA) Chapter 7020. The MPCA has had rules for the control of pollution from animal waste facilities since 1971. In December 1979, new rules were adopted to allow the processing of feedlot permits by the counties. There are several advantages to county participation. In most cases, county officials have better communication with feedlot owners, knowledge of individual sites and their histories, and an awareness of local concerns and conditions. County feedlot officers can provide applications and materials for the farmer, help ensure that applications are completely filled out and process most applications. The MPCA uses the information on the feedlot permit application to evaluate a feedlot's compliance with state rules.

Twenty-five counties in the state have volunteered to participate in the feedlot program. Thirty-five county zoning administrators responded to a questionnaire in 1989. The survey indicated that most counties know about the program, but barriers to entering the program include reservations about enforcement, lack of staff, funds, and technical assistance.

The following requirements must be met for all feedlots in Minnesota:

1) No feedlot or manure storage area shall be constructed, located, or operated so as to create or maintain a potential pollution hazard unless a certificate of compliance or an agency permit has been issued,

2) Animal manure, when utilized as domestic fertilizer, shall not be stored for longer than one year and shall be applied at rates not exceeding local agricultural crop nutrient requirements except where allowed by permit, and

3) Any animal manure not utilized as domestic fertilizer shall be treated or disposed of in accordance with applicable state rules.

Owners of feedlots with more than 10 animal units[1] are required to complete a feedlot permit application whenever any of the following conditions occur:

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[1]Ten animal units are roughly equivalent to 10 steers, 7 mature dairy cows, 25 swine over 55 lbs., 10 horses, 555 turkeys, or 1000 chickens.

1) a new feedlot is constructed,

2) a feedlot is expanded or modified,

3) a change in ownership takes place,

4) an existing feedlot is restocked after being abandoned for more than five years,

5) an inspection by MPCA staff reveals that the feedlot is creating a potential pollution hazard.

The feedlot permit application will be reviewed by a county feedlot officer or an MPCA staff member. The facility is often checked by county feedlot officers and sometimes by MPCA staff. Applications are forwarded to the MPCA for evaluation if the feedlot has greater than 1000 animal units, 300 to 1000 animal units with potential pollution hazards, or a potential pollution hazard which will not be corrected within a ten month period. If the review indicates that all manures are being used as fertilizer and that any potential pollution hazards have been addressed with corrective measures, a Certificate of Compliance will be issued. If the application review indicates that the feedlot is creating a pollution hazard, the MPCA may issue an Interim Permit. Interim Permits are issued when the potential pollution hazard can be corrected within 10 months. When all corrective measures are in place, a Certificate of Compliance will be issued.

In special circumstances, a five-year permit may be issued. These permits are used when technical considerations or financial hardship prevent correction of the pollution problem within 10 months. A timetable outlining steps to be taken to reduce pollution will be included with each five-year permit.

To date, over 16,000 feedlot permit applications have been reviewed in the last 20 years in Minnesota, with 5300 Certificates of Compliance (4300 by MPCA and 1000 by counties) and 1100 interim permits issued since 1980. Also, 35 five-year permits and eight NPDES permits are in effect. The existing program has, at a minimum, communicated to those operators and county personnel the intent of the state to protect water quality from animal waste and the methods to accomplish that.

A potential downside to the existing feedlot rules is that there is little or no incentive for operators to continue to meet guidelines once they have a certificate of compliance. Regulatory personnel do not have the time for ongoing monitoring of feedlot operations.

Until a couple of years ago the focus of the feedlot program was on surface water protection. Nutrient management for cropped fields and animal waste storage methods have received greater attention in recent years. There are no requirements for abandoned feedlots and only general conditions for manure application to fields are in the current permitting system. Soil Conservation Service or Midwest Plan Service design criteria must be met when constructing storage basins.

SUMMARY

A rough estimate of the number of feedlots in Minnesota is 45,000 to 60,000. The estimated amount of manure N produced by livestock operations in the state is 269,000 tons, equivalent to the amount of waste N generated by about 77 million people. Manure-N can move into ground water below outdoor animal holding areas, manure storage areas, fields with applied manure and abandoned feedlots.

A soil seal will usually develop under animal holding areas that are continually used, preventing much movement of water through the soil surface. Saturated conditions in the feedlot surface, coupled with high amounts of organic carbon, makes a feedlot surface conducive for denitrification. This seal can be broken and a number of investigators have found NO_3 and NH_4 moving through the soil profile and into ground water below active feedlots. This is especially a problem with abandoned feedlots. In many abandoned feedlot situations, planting alfalfa or other high N use crops may reduce the potential for NO_3 leaching to ground water. Some of the leached nitrogen is likely to be lost through denitrification. Runoff from active feedlots and subsequent infiltration can also contribute to ground water nitrogen.

Earthen basins used to store or treat manure have been found to leak in certain areas, releasing nitrogen to ground water. A number of other studies have shown earthen basins to effectively seal themselves, with minimal ground water impacts. The potential for N movement below earthen basins can be greatly minimized when designed by a state-certified engineer or properly trained SCS employee. The high nitrogen concentrations in basins, great number of basins in the state, and contribution to ground water NO_3 found in some areas makes it critical that earthen basins are properly sited, designed and constructed.

An integral part of the manure management system that can potentially cause ground water nitrogen problems is field application of manure. This aspect of manure management is discussed in the crop production Chapter G of this report.

The Minnesota Pollution Control Agency has had rules to control pollution from animal waste facilities since 1971. In 1979, the rules were changed to allow counties to process feedlot permits, and since that time twenty-five counties have volunteered to participate in the program. Over 16,000 feedlot permit applications have been reviewed in the last 20 years. Until a couple of years ago, the focus of the feedlot program was on surface water protection. Nutrient management for cropped fields and animal waste storage methods have received greater attention in recent years.

RECOMMENDATIONS

- Specifics on manure storage site selection, design, and construction criteria should be included into Chapter 7020. Methods of construction and testing results should be documented and reported to the county or MPCA.

- It is recommended that more accurate information be obtained in Minnesota regarding the number of feedlots, type of feedlots, number of animal units, number of feedlots with pollution hazards and the number of feedlot changes occurring that should trigger permits. The information would be very useful for redefining, restructuring, and better managing the feedlot program.

- Local government, through local water planning could make a valuable contribution by collecting data pertaining to storage, crediting, and application of manure. Information on abandoned feedlots should also be part of the data collection effort. Technical and financial assistance will be necessary to design and evaluate inventories.

- It is recommended that a more aggressive approach be taken to encourage Best Management Practice implementation to prevent surface runoff from feedlots, infiltration below feedlots, and over application of manure.

- State funded incentive programs should be developed, training should be provided, and technical assistance increased to encourage counties to adopt and actively administer MPCA's Feedlot Rules (Chapter 7020). Counties should submit annual reports describing program status.

- Detailed manure management plans should be required with feedlot permit applications (MPCA is starting to require this for large facilities).

- New large concentrated feedlots should be encouraged to locate in agronomic areas of the state so that the nutrients in manure can be properly utilized for cropland production. Geologic sensitivity must also be considered when locating new feedlots.

- Soil and Water Conservation District personnel should be further trained in manure management in order to further assist Soil Conservation Service personnel in providing technical assistance. Alternative and additional technical assistance sources should be sought and funded.

- The feasibility of scraping the upper layer of soil from abandoned feedlots and redistributing this nitrogen rich soil over larger areas should be studied.

- Studies should be conducted to determine nitrogen impacts from manure composting piles and the advantage and disadvantages of composting manure.

- Additional staff should be added to the MPCA feedlot program.

Literature Cited

Alabrecht, K.A. and K. Cartwright. 1989. Infiltration and hydraulic conductivity of a compacted earthen liner. Ground Water. 27:1:4-19.

Barrington, S.F. and R.S. Broughton. 1988. Designing earthen storage facilities for manure. Canadian Agricultural Engineering. 30:2:289-292.

Barrington, S.F. and C.A. Madramootoo. 1989. Investigating seal formation from manure infiltration into soils. Transactions of the ASAE. 32:3:851-856.

Barrington, S.F. and P.J. Jutras, and R.S. Broughton. 1987(a). The sealing of soils by manure. I. Preliminary investigations. Canadian Agricultural Engineering. 29:2:99-103.

Barrington, S.F., P.J. Jutras, and R.S. Broughton. 1987(b). The sealing of soils by manure. II. Sealing mechanisms. Canadian Agricultural Engineering. 29:2:105-108.

Comfort, S.D., K.A. Kelling, D.R. Keeney, and J.C. Converse. 1988. The fate of nitrogen from injected liquid manure in a silt loam soil. J. Environ. Qual. 17:2:317-322.

Culley, J.L.B. and P.A. Phillips. 1989. Retention and loss of nitrogen and solids from unlined earthen manure storages. Transactions of the American Society of Agricultural Engineers. 32:2:677-683.

Dalen, L.D., W.P. Anderson, and R.M. Rovang. 1983. Animal manure storage pond ground water quality evaluation. Paper 83-4572 presented at the 1983 winter meeting of the American Society of Agricultural Engineers. Chicago, Illinois, Dec. 1-19, 1983.

Egboka, B.C.E. 1984. Nitrate contamination of shallow groundwaters in Ontario, Canada. The Science of the Total Environment. 35:53-70.

Elliott L.F., and T.M. McCalla. 1972. The composition of the soil atmosphere beneath a beef cattle feedlot and a cropped field. Soil Sci. Soc. Amer. Proc. 36:68-70.

Elliott, L.F., A.M. McCalla, L.N. Mielke, and T.A. Travis. 1972. Ammonium, nitrate, and total nitrogen in the soil water of feedlot and field soil profiles. Applied Microbiology. 23:4:810-813.

Ellis, J.R., L.N. Mielke, and G.E. Schuman. 1975. The nitrogen status beneath beef cattle feedlots in eastern Nebraska. Soil Sci. Soc. Amer. Proc. 39:107-111.

Gangbazo, G., D. Cluis, and M. Vallieres. 1989. Nitrogen seepage from earthen-built manure storage tanks. A paper presented at the 1989 international summer meeting by the American Society of Agricultural Engineers and the Canadian Society of Agricultural Engineering, June 25-28, 1989.

Gillham, R.W. and L.R. Webber. 1969. Nitrogen contamination of groundwater by barnyard leachates. J. Water Pollution Control Federation. 41:10:1752-1762.

Heatwole, C.D., P.L. Diebel, and J.M. Halstead. 1990. Management and policy effects on potential groundwater contamination from dairy waste. Water resources bulletin. 26:1:25-33.

Jones, J.H., F.J. Olsen, J.J. Patterson and K.A. Rushing. 1984. The pollution potential of rainfall runoff from a dairy liquid waste disposal site. Agricultural wastes. 10:3:177-186.

Legg, Thomas D., Jerald J. Fletcher, and K. William Easter. 1989. Nitrogen budgets and economic efficiency: A case study of southeastern Minnesota. J. Prod. Agric. 2:2:110-116.

Lorimor J.C., L.N. Mielke, L.F. Elliott, and J.R. Ellis. 1972. Nitrate concentrations in ground water beneath a beef cattle feedlot. Water resources bulletin. 8:5:110-116.

Magette, William L., Russell B. Brinsfield, Robert E. Palmer, and James D. Wood. 1989. Nutrient and sediment removal by vegetated filter strips. Transactions of the American Society of Agricultural Engineers. 32:2:663-667.

Massie,L.R.. Livestock yard runoff control systems. 1980. A cooperative extension program publication #A3077 University of Wisconsin, Madison, WI.

Mathers, A.C., B.A. Stewart and Betty Blair. 1975. Nitrate-nitrogen removal from soil profiles by Alfalfa. J. Environ. Qual. 4:3:403-405.

Midwest Planning Service. 1985. Livestock waste facilities handbook. MWPS-18.

Mielke, Lloyd N., Norris P. Swanson, and Thomas M. McCalla. 1974. Soil profile conditions of cattle feedlots. J. Environ. Quality. 3:1:14-17.

Mielke, L.N., J.R. Ellis, N.P. Swanson, J.C. Lorimor and T.M. McCalla. 1970. Groundwater quality and fluctuation in a shallow unconfined aquifer under a level feedlot. Cornell Agr. Waste Mgmt. Conf. Proc., Rochester, N.Y., Jan. 1970. pp. 31-40.

Mielke, L.N., and A.P. Mazurak. 1976. Infiltration of water on a cattle feedlot. Transactions of the ASAE 1976. pp. 341-344.

Mielke L.N. and J.R. Ellis. 1976. Nitrogen in soil cores and ground water under abandoned cattle feedlots. J. Environ. Qual. 5:1:71-75.

Miller, W.D. 1971. Subsurface distribution of nitrates below commercial cattle feedlots, Texas high plains. Water resources bulletin. 7:5:941-950.

Miller, M.H., J.B. Robinson, and D.W. Gallagher. 1976. Accumulation of nutrients in soil beneath hog manure lagoons. J. Environ. Qual. 5:3:279-282.

Miller, M.H., J.B. Robinson, and R.W. Gilham. 1985. Self-sealing of earthen liquid manure storage ponds: I. A case study. J. Environ. Qual. 14:4:533-538.

Minnesota Agricultural Statistics Service. 1991. Issued cooperatively by U.S. Department of Agriculture and Minnesota Department of Agriculture. 99 pp.

Norstadt and Duke. 1985. Soil profiles: nitrogen conversion and salt motility altered by feedlot manure management. Soil Sci. Soc. Am. J. 49:658-664.

Olson, R.V., R.V. Terry, W.L. Powers, C.W. Swallow, and E.T. Kanemasu. 1982. Disposal of feedlot-lagoon water by irrigating Bromegrass: II. Soil accumulation and leaching of nitrogen. J. Environ. Qual. 11:3:400-404.

Phillips, P.A., J.L.B. Culley, J.M. Leclerc, and C. Fox. 1983. Groundwater seepage from earthen manure storages. Paper 83-4570 Presented at the 1983 winter meeting of the American Society of Agricultural Engineers. Chicago, Illinois, Dec. 13-16, 1983.

Ritter, W.F. 1983. Influence of animal waste lagoons on groundwater quality. Paper 83-4573 Presented at the 1983 winter meeting of the American Society of Agricultural Engineers. Chicago, Illinois, December 13-16, 1983.

Ritter, W.F. and A.E.M. Chirnside. 1987. Influence of agricultural practices on nitrates in the water table aquifer. Biological wastes. 19:165-178.

Ritter, E.W., E.W. Walpole and R.P. Eastburn. 1984. Effect of an anaerobic swine lagoon on groundwater quality in Sussex County, Delaware. Agricultural Wastes. 10:267-284.

Rowsell, J.F., M.H. Miller, and P.H. Groenevelt. 1985. Self-sealing of earthen Liquid manure storage ponds: II. Rate and mechanism of sealing. J. Environ. Qual. 14:4:539-542.

Schuman, G.E. and L.F. Elliott. 1978. Cropping an abandoned feedlot to prevent deep percolation of nitrate-nitrogen. Soil Science. 126:4:237-243.

Schuman, G.E. and T.M. McCalla. 1975. Chemical characteristics of a feedlot soil profile. Soil Science. 119:2:113-118.

Sommerfeldt, Theron G., U.J. Pittman, and R.A. Milne. 1973. Effect of feedlot manure on soil and water quality. J. Environ. Quality. 2:4:423-427.

Stewart, B.A., F.G. Viets, G.L. Hutchinson, W.D. Kemper, F.E. Clark, M.L. Fairbourn, and F. Strauch. 1967. Distribution of nitrate and other water pollutants under fields and corrals in the middle south Platte Valley of Colorado. From Stewart, B.A., F.G. Viets, and G.L. Hutchinson. 1968. Agriculture's effect on nitrate pollution of groundwater. J. Soil Water Cons. 23:13-15.

Taiganides, E.P. and T.E. Hazen. 1966. Properties of farm animal excreta. American Society of Agricultural Engineers. 9:374-376.

Tjostem, J.L., J. Young, C. Hoilien, and R. Elson Iverson. 1977. Bacterial and nitrate contamination of well water in northeast Iowa. Proc. Iowa Acad. Sci. 84:14-22.

Terry, R.V., W.L. Powers, R.V. Olson, L.S. Murphy, and R.M. Rubison. 1981. The effect of beef feedlot runoff on the nitrate-nitrogen content of a shallow aquifer. J. Environ. Qual. 10:1:22-26.

United States Department of Agriculture and Environmental Protection Agency. 1982. State-of-the art review of best management practices for agricultural nonpoint source control: animal waste. Interagency publication; 12-05-300-472.

Webster, Colin and Keith W. T. Goulding. 1989. Influence of soil carbon content on denitrification from fallow land in Autumn. J. Sci. Food Agric. 49:131-142.

SEPTIC SYSTEMS

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INTRODUCTION

On-site wastewater treatment can be defined loosely as any concept for treating and disposing wastewater in the proximity of its source. There are several different types of systems for treating waste on-site. Based on the 1980 census, over 362,000 households in Minnesota used on-site wastewater treatment systems for waste treatment and disposal, a 16 percent increase over the 1970 census figures. In 1991, it is likely that there are over 400,000 homes using on-site systems.[1] Numerous septic systems are also used for seasonal cabins. The estimated average nitrogen (N) from septic systems per square mile of land for each county is shown in Figure I-1. In many cases there is no economic environmentally acceptable alternative approach for treating waste other than using some form of on-site systems.

A conventional on-site wastewater treatment system will typically consist of two main parts: the sewage tank and the soil treatment unit. The terms "septic systems" and "on-site wastewater treatment systems" will be used interchangeably in this report. Both will refer to treatment using at least a septic tank and soil treatment unit. As the sewage enters the septic tank, those solids which are heavier than water settle to the bottom and those solids lighter than water float to the top, forming a scum layer. The solids which settle to the bottom of the tank are partially decomposed by anaerobic bacteria present in the tank, forming a layer of sludge which must be periodically removed.

The water, dissolved and suspended solids will drain from the septic tank into the soil treatment unit, which provides final treatment and disposal of the septic tank effluent. Many older systems in the state do not have a soil treatment unit, but dispose of septic tank effluent directly onto the soil surface, tile lines, surface water, or allow sewage to move directly from the tank into deep soil layers via seepage pits or dry wells. There are three types of soil treatment systems used in Minnesota: 1) drainfield trenches, 2) seepage beds, 3) mounds, and 4) seepage pits. Drainfield trenches are the most common soil treatment system. Seepage beds are more commonly used on smaller lots. Seepage pits and dry wells are currently not allowed. Mound systems are one type of system that is frequently installed in Minnesota where restricting conditions exist. A mound system is similar to an elevated seepage bed. Sand fill is used on top of the existing soil to create additional separation distance between effluent release and bedrock, slow or fast percolation rate soils or subsurface saturated conditions. For more information describing the various types of soil treatment units, see the on-site sewage treatment manual published jointly by the Minnesota Extension Service and the MPCA.

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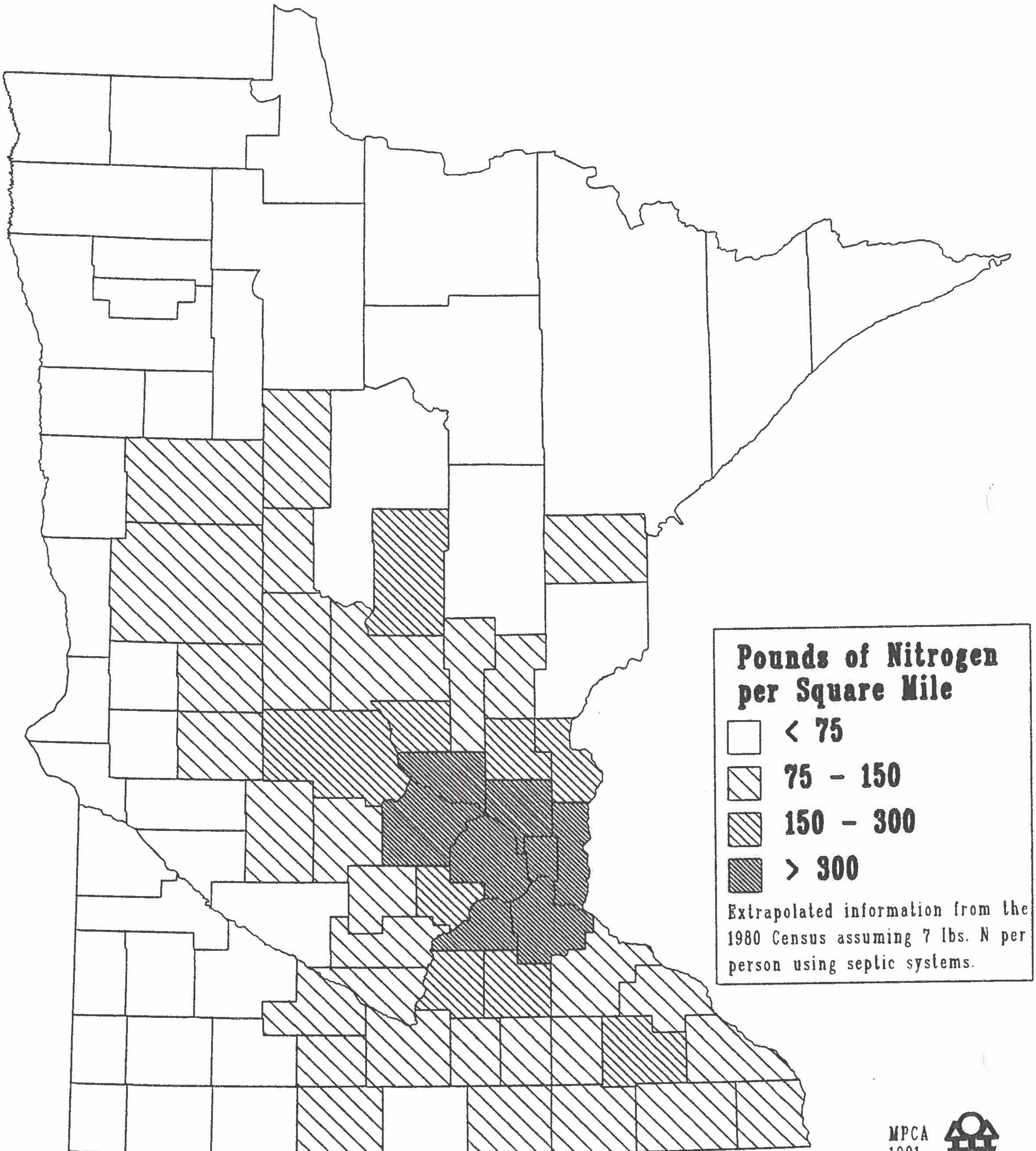
[1] 1990 census information results for the number of septic systems will not be compiled and reported until January 1992.

Figure I-1
Average Amount of Nitrogen from Septic Systems per Square Mile



Septic systems have been generally recognized over the years as acceptable means for disposing of water carried wastes. The effluent from a septic tank contains solids, biological oxygen demand (BOD), chemical oxygen demand (COD), phosphorus, N, chloride, bacteria, viruses, and organic chemicals. In a properly sited, installed, and maintained on-site wastewater treatment system many of the pollutants will be treated within two to three feet below the drainfield. The solids, BOD, and COD are largely treated or removed in the sewage tank. Many soils are capable of fixing large quantities of phosphorus and therefore phosphorus is usually retained in the soil below the system. However, phosphorus movement to lakes and streams via ground water is possible under systems where noncalcareous soils are low in iron and aluminum oxides and clay, where large cracks in the soil exist or where the soil is saturated directly below the drainfield (Chen, 1988; Jones and Lee, 1979; Gilliom and Patmont, 1983). Bacteria, viruses and other microorganisms can be carried along by the liquid flowing through the soil. Unless saturated soils exist below the drainfield, bacteria usually are attenuated in the soil until they die off (Alhajjar, et al., 1988). But, depending on the nature of the soil, temperature, and amount of water moving through the soil, viruses can move through the soil into ground water (Alhajjar, et al., 1988; Powelson, et al. 1990). The likelihood of bacteria or virus movement to ground water is greatly reduced under septic systems constructed to Minnesota Rules, Chapter 7080. Another potential contaminant from septic systems is volatile organic chemicals which are used for cleaning and other purposes and often get flushed down the toilet or dumped in the drain.

While there are several potential pollutants from septic systems, perhaps the pollutant of greatest concern is nitrate (NO_3). Septic systems are not specifically designed to remove N. This chapter will focus on N movement from septic systems into ground water. Numerous studies of N impacts on ground water from both 1) individual systems and 2) numerous systems in communities will be summarized. Best Management Practices for minimizing septic system N impacts are described along with recommendations for further study and changes in policy.

FACTORS AFFECTING NITROGEN MOVEMENT TO GROUND WATER BELOW SEPTIC SYSTEMS

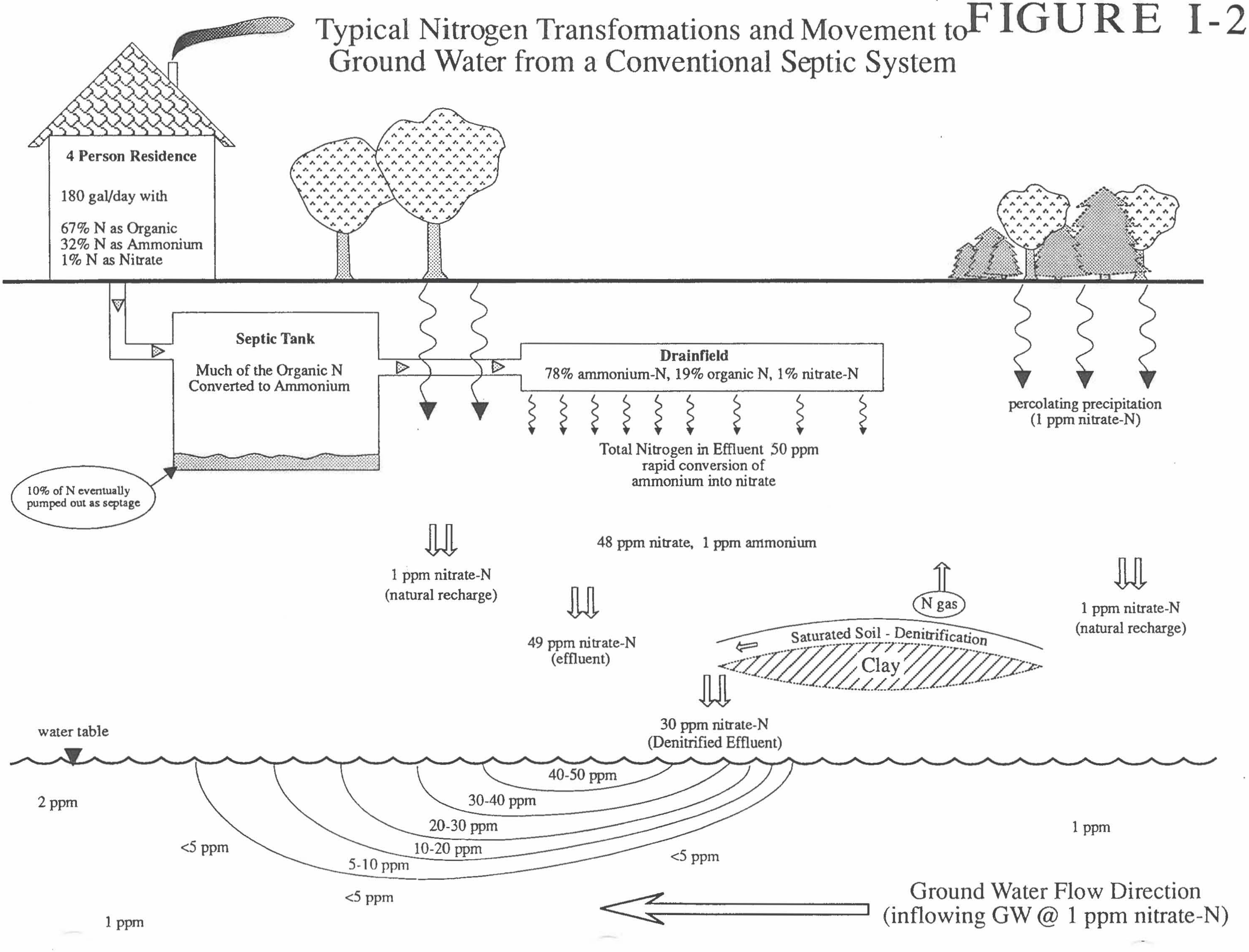
Amount of Nitrogen Originating from Septic Systems

The amount of N waste that an average family of four produces each year is stated in the literature to be between 19 and 73 pounds (Walker, et al., 1973b; Laak, 1986; Bouma, et al., 1972; Siegrist, et al., 1976). Septic system effluent total N concentrations range widely, with reported averages generally between 38 and 62 mg/l (Canter and Knox, 1985; Siegrist, Anderson, YEAR; Wehrman, 1983; Otis, et al., 1975; Bauman, 1985, Otis and Boyle, 1976).

Studies of wastewater flows have shown very similar ranges in the average amount of wastewater generated, with reported average flows between 42.6 and 47.5 gallons per person per day (Bennet and Linstedt, 1975; Cohen and Wallman, 1974; Witt et al., 1974.; Siegrist et al., 1976). Assuming that a family of four discharges 180 gallons/day with an effluent total N concentration between 40 and 60 mg/l, then the total amount of N released in septic effluent annually would range between 22 and 33 pounds for this household.

FIGURE I-2

Typical Nitrogen Transformations and Movement to Ground Water from a Conventional Septic System



Since about 1,200,000 people (400,000 households) in Minnesota discharge water into septic tanks on a regular basis, approximately 6.6 to 10.2 million pounds of N are released in septic tank effluent each year. This amount of N is roughly 0.5 percent of all anthropogenic N released each year in Minnesota (Chapter F).

N in wastewater is largely derived from nonfecal toilet flush (40%), but also originates from fecal toilet flush (22%), garbage disposal (9%), clothes wash/rinse (11%), kitchen sink/dishwasher (13%), and bath/shower (5%) (Witt, et al. 1974). It is likely that N in clothes wash is currently lower than stated in the above referenced 1974 study due to the current popularity of disposable diapers and diaper services. If the domestic water source contains elevated NO_3, this NO_3 will contribute to the overall N in the wastewater stream. The type of facility utilizing on-site wastewater treatment systems can also influence the N concentrations entering the septic system (Converse et al., 1984). Facilities with large amounts of food waste or processed wastes high in N will have greater N concentrations in septic system effluent than typical households.

Nitrogen Transformations and Loss Mechanisms

Typical N transformations from the waste source to ground water are shown in Figure I-2. Influent wastewater serving single households was found to be comprised of 78 percent organic N, 21 percent NH_4, and 1 percent NO_3 (Witt et al., 1974). The septic tank is ineffective in N removal. However, much of the organic N is converted to NH_4 (ammonification) in the septic tank by anaerobic bacteria. As a result, the predominant form of N entering the soil absorption system from a conventional septic system is NH_4-N. Once released to the absorption bed and underlying soil, the remaining organic-N in the effluent is mostly converted to NH_4 in either aerobic or anaerobic conditions. About 10 percent of the total N in raw sewage is removed via sludge in a septic tank (Hardisty, 1974).

Anaerobic conditions will normally prevail immediately below a soil absorption system. This was found by Walker et al. (1973a) to be due to the presence of an impeding layer, a "crust" or "biomat", at the boundary between the gravel bed and adjacent soil. In this anaerobic state, the N will remain in the NH_4 form and be readily adsorbed onto soil particles. While adsorption is usually the major mechanism for retaining NH_4 in the soil, other processes including incorporation into microbial biomass or uptake by plants will also minimize NH_4 leaching. However, it is possible under anaerobic conditions for the cation exchange sites in the soil beneath seepage beds or trenches to become equilibrated with cations in the effluent, resulting in leaching of NH_4 (Sikora and Corey, 1975). Ammonium movement to ground water was noted in studies where a high water table[1] was found to cause anaerobic conditions immediately beneath the seepage bed (Walker et al., 1973b).

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[1]These systems would not conform with Chapter 7080.

In most soil absorption systems, the saturated or anaerobic conditions will prevail only a short distance below the system. Under unsaturated (aerobic) conditions, the NH_4 released from septic systems is readily converted to NO_3 (Walker et al., 1973a; Reneau, 1979; Canter and Knox, 1985). This conversion has been shown to occur within a couple inches below the crusting zone of the seepage bed in a well aerated soil. Finer textured soils that restrict movement of water will allow for less nitrification to occur.

In most properly sited septic absorption systems, most of the N will have been converted to NO_3 within a few feet below the seepage bed. Where septic tanks are used without soil treatment, it is likely that much of the N will remain in the NH_4 form near the point of release.

As percolating effluent moves below the rooting zone, denitrification is the only means of reducing the N content. Biological denitrification results in the reduction of NO_3-N to N gas. Denitrification requires:

1) The presence of an anaerobic zone following nitrification,

2) An adequate carbon source for denitrifying bacteria in the anaerobic zone, and

3) Sufficiently warm temperatures for the process to readily occur (> 10°C).

In a properly functioning soil absorption field, denitrification is often limited by the absence of reducing (anaerobic) conditions following nitrification and by the lack of an available carbon source (Lamb et al., 1987).

Denitrification is possible where restricting or semi-restricting layers of substrata are encountered by percolating effluent. Reneau (1979) believed denitrification was responsible for a reduction in NO_3 moving in a saturated zone towards a tile line. Significant denitrification is unlikely to occur in well-aerated sandy subsoil (Walker, et al. 1973 1b). However, there is some evidence that denitrification can occur in a soil that is overall aerobic but contains saturated or nearly saturated aggregates (Smith, 1980; Rodgers, 1980). Within these aggregates, anaerobic microsites may exist making denitrification possible.

In general, the fate of N and occurrence of denitrification is very complex and is difficult to accurately assess and quantify due to the number of mechanisms involved in transformations from one form of N to another (Biswas and Warnock, 1985).

<u>**Soil, Climate, and Vegetation**</u>

As previously noted, soil characteristics can greatly affect the major N transformations. Of great importance is the oxygenated state of the soil, which is largely affected by the soil texture, structure and landscape position. In Delaware, 480 water table aquifer wells were sampled under varying land uses including residential areas utilizing septic systems (Ritter and Churnside, 1984). Residential areas developed in poorly drained soils had much lower NO_3 then residential areas developed in more well-drained soils. Denitrification was believed to be a reason for seeing the lower NO_3 in the poorly drained soils region. Miller (1972) found in a Delaware housing development utilizing septic systems that NO_3-N concentrations in ground water were higher below well drained soils (5-30 mg/l) as compared to a second area of varying permeability and higher seasonal water table (0.01 to 11.3 mg/l). While NO_3 is the usual N end product below a septic system, Sikora and Corey (1975) stated that some N is likely to be in the NH_4 form in finer textured soils.

Soil conditions for maximum denitrification losses would be a permeable soil below the drainfield underlain by less permeable soils. This would allow maximum nitrification in the unsaturated zone and a saturated zone for denitrification.

Starr and Sawhney (1980) found annual precipitation to greatly affect N transformation. In a year of above normal rainfall, little of the NH_4 N from septic system effluent was converted to NO_3. As a result, NH_4-N moved without apparent loss to greater depths. Nitrification rates are also reduced when temperatures drop below 10°C (Lamb, et al. 1987). Therefore, less NH_4 will be converted to NO_3 in the late fall to mid spring months in Minnesota.

Percolating rainfall can dilute the N concentration of septic tank effluent. Dilution will be most effective in areas/times of greater precipitation and where the infiltration capacity of the soil is high.

Roots from trees and grasses can intercept percolating rainfall/effluent resulting in N uptake. Plant uptake of N will primarily occur from about May to November in Minnesota. Septic system drainfields are often placed very shallow into the soil in order to obtain some N loss through grass roots.

<u>Type of System</u>

The N transformations and loss mechanisms previously described are typical for conventional on-site wastewater treatment systems. N loss through denitrification under conventional systems can vary from about 0 to 40 percent (Eastburn and Ritter, 1984; Laak, 1986). Lamb et al. (1987) found a 0 to 6 percent reduction of N under conventional septic systems. This loss will depend on the suitability of the underlying soil for denitrification which should include successive aerobic/anaerobic conditions and suitable soil temperature and carbon content of the soil and effluent.

A system that is commonly used in areas of high water tables, shallow bedrock or less permeable soils is a mound system. A mound system will often create conditions suitable for denitrification in the soil below the absorption field so that N losses of 40 to 70 percent are possible (Harkin et al., 1979). Other systems have been developed that create conditions for denitrification to occur before the effluent is discharged into the soil absorption system. Alternative systems which promote denitrification will be further described in the Best Management Practice Section of this report.

It should also be noted that in systems where dosing can be controlled, denitrification potential can be increased through proper dosing rates. More research is needed to understand the most appropriate dosing rate for specific soil types.

THE EFFECTS OF SEPTIC SYSTEMS ON GROUND WATER NITROGEN CONCENTRATIONS

Overview

Many factors can affect ground water N concentrations in areas where septic systems are in use. The average N content of percolating water over a housing development with septic systems is primarily dependent on the following factors:

1) Amount of wastewater produced per dwelling;

2) N concentration of the wastewater effluent;

3) Rate of "natural" ground water recharge;

4) N concentration of the percolating "natural" recharge;

5) Denitrification in the vadose zone;

6) Density of dwellings with septic systems; and

7) Other upgradient sources of N.

Assuming denitrification to be negligible,[1] the average NO_3 concentration of the water recharging the aquifer (effluent plus natural recharge) can be calculated using a weighted mean.

For example, if one house on an acre of land uses 180 gal/day (681 liters/day) of water with an average effluent N concentration of 50 mg/l, and six inches of precipitation having an average N concentration of 1 mg/l recharges the underlying aquifer during a year, then the net average annual NO_3 concentration reaching ground water (assuming complete nitrification) over this area would be 15 mg/l. If denitrification takes place, this concentration would be reduced. However, if natural recharge was less than six inches and/or "natural recharge" had a higher NO_3 concentration, the average NO_3-N concentration reaching ground water would be greater than 15 mg/l.

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[1]Under certain conditions, it is possible for significant N losses to occur through denitrification in soils between the septic system and water table.

Once percolating water reaches ground water, dilution and dispersion will occur. The calculated average concentration in the above example was 15 mg/l. Since mixing is not uniform, the NO_3-N concentration would be expected to be much greater below the drainfield and in the effluent plume and be much less than 15 mg/l outside of the plume. The N concentration of the diluting ground water will influence the effectiveness of ground water dilution. (e.g. A high NO_3 ground water will be much less effective in diluting NO_3 from a septic effluent plume than low NO_3 ground water.) Ground water dilution is also dependent on the depth below the water table to which mixing takes place (vertical flow and dispersion) and the horizontal velocity of ground water flow. Also, if the ground water flow velocity is low, the dilution capacity of the aquifer will be low (Magner et al., 1987). An aquifer that has a high hydraulic conductivity, high horizontal and vertical hydraulic gradients, and low N in the inflowing ground water will be able to dilute a large volume of percolating septic system effluent.

In summary, the impacts of septic systems on ground water will primarily depend on the N loading to the aquifer and the diluting capacity of the aquifer. Another factor is the potential for denitrification to occur within the aquifer. Dilution by ground water is often the most important factor for lowering NO_3 concentrations, and therefore, the location and depth below the water table of a domestic well in relation to a septic tank are very important variables for explaining observed ground water NO_3 concentrations. Because of dilution or the narrowness of the effluent plume, a given well may not noticeably be affected by an individual septic system. However, as more and more septic systems contribute NO_3 along the ground water flow path, the dilution capacity of ground water is reduced. In the following sections of this report, the impacts from both individual on-site wastewater treatment systems and the collective impacts of numerous systems studied in Minnesota and other states will be described.

Impacts from Individual Systems - Case Studies

After a review of the literature from seven states, Harken, et al. (1979) concluded that seriously elevated NO_3-N levels are rarely encountered in ground water impacted by septic systems, and that whenever high NO_3 occurred it was when the well was in the immediate vicinity of the seepage area or located in the water table aquifer just downgradient of the septic system. In studies of ground water NO_3 levels in Wisconsin, Minnesota, Missouri, Illinois, California, Kansas and Oregon, rarely could septic systems be clearly implicated as sources of excessive NO_3 pollution. Walker, et al. (1973b) stated that the only potential problem with NO_3 in ground water from scattered dwellings would be local contamination of shallow wells downgradient of systems.

Ground water monitoring wells have been installed around septic system drainfields for several studies conducted in Northern United States and Southern Canada. Results of these studies are represented in Table I-1 and are described in more detail in the following pages.

TABLE I-1: **Studies where ground water nitrate concentrations were determined below and around individual on-site wastewater treatment systems.**

LOCATION / REFERENCE	# SYSTEMS STUDIED / GEOLOGY	# OF MONITOR. WELLS PER SITE	BACKGROUND NO3-N CONCENTRATION	AVERAGE G.W. NO3-N DIRECTLY BELOW SYSTEM DRAINFIELD	CONCLUSIONS
Cambridge, Ontario Robertson, 1989	1 Outwash/lacustrine	Many	27 mg/l	33 mg/l in plume	• Downward movement of contaminants was observed. • A long thin plume of impacted ground water resulted in this low dispersivity aquifer.
Throughout Wisconsin Dudley and Stephenson, 1973	11 9/11 sites in outwash sands	3-8	--	15 mg/l	• Nitrate-N was > 10 mg/l at all sites less than 25 feet downgradient of the system. • Nitrate-N was < 10 mg/l and ammonium was < 1 mg/l at all sites 50 ft. downgradient of the system.
South Central Wisconsin Alhajjar et al., 1987	17 Surficial sand aquifer	--	< 3 mg/l	22 mg/l	• Average nitrate-N decreased to 7 mg/l 20 feet downgradient from the drainfield
Wisconsin Walker et al., 1973b	4 varied	18	--	10-15 mg/l at 5 ft. depth >> 109 mg/l at 1 ft. depth	• High nitrate was restricted to the upper part of the aquifer. • Ammonium can be the major nitrogen source reaching ground water where soils are saturated. • Nitrate concentrations decreased downgradient but were still greater or equal to 10 mg/l 100 feet downgradient at 2 sites.
Minnesota MPCA (1991)	9 varied	1-7	usually < 3 mg/l		• Nearby wells had varied conc's. Highest median conc. was 23.5 mg/l. 4/9 sites had no clear impacts.
Wisconsin Bouma et al., 1972	5 varied	–	–	15-30 mg/l	• High ammonium was found in ground water at 1 site.
New York State Chen, 1988	17 all near lakes	2-3	very low	Nearby wells had max. of 3.7 mg/l.	• Only 3 wells had nitrate-N > 1 mg/l. • Ammonium-N was above 1 mg/l in 10 wells at 6 sites. Two wells had ammonium-N > 10 mg/l.
Wisconsin High School Drainfield Polkowski et al., 1970	1	several	< l mg/l	Max. of 21 mg/l	• Nitrate was never observed to be > 10 mg/l 100 feet downgradient of the drainfield. • Nitrate was over double background at 265 feet from the drainfield.
Braceridge, Ontario Robertson, 1991	1 fine sands overlying granite	250	< 10 mg/l	30 mg/l in center of plume	•Sharp lateral and vertical plume boundaries were evident.

Study 1 - Two systems in Ontario, Canada

An extensive monitoring network was installed near Cambridge, Ontario in a surficial sand aquifer to delineate a zone of ground water impacted by a typical domestic septic system (Robertson, et al., 1989; Robertson et al., 1991). The site was located on a flat lying sand plain where glacio-lacustrine and outwash sand occurs to a depth of 13 to 26 feet and overlies a silty till of low permeability. The water table was 6.5 to 8 feet below ground surface in the area of the tile bed (absorption field). A family of four persons had lived at the site for eleven years at the time of study. The septic system consisted of a holding tank and a weeping tile bed of about 100 square meters. The horizontal rate of ground water flow in the area of the tile bed was found to be on the order of 60 feet per year. Calculated flow rates downgradient of the tile bed were about 130 feet per year. Most wells were screened 10 to 40 feet below the water table.

Nitrate concentrations were much lower in the deeper wells, and were the highest within the upper 10 to 15 feet of saturated thickness. The plume from the septic system had average NO_3-N concentrations of 33 mg/l. Since background NO_3-N concentrations were about 27 mg/l, dilution would not be expected to greatly lower NO_3 concentrations. The plume approached the confining till layer in the area of the tile field showing downward movement of contaminants. **The study concluded that a single household septic system located in a sandy unconfined aquifer is shown to produce an extremely long (> 430 ft.) thin (< 33 ft.) plume of impacted ground water if the dispersive capacity of the aquifer is low.**

Another site studied in Ontario (Robertson, et al. 1991) is located on the edge of the Muskoka River near Bracebridge, Ontario. The trenched tile bed septic system has served two adults since 1987 and discharges effluent to fine sand overlying granite bedrock. Depth to water at this site was 10 feet. Data from over 250 sampling points showed a plume of NO_3 contaminated water that extended 66 feet to the river and about eight feet below the water table. Nitrate-N concentrations in the center of the plume were over 30 mg/l, with background concentrations less than 10 mg/l. The dispersive nature of the aquifers at this site was found to be weak, and as a result sharp lateral and vertical plume boundaries were evident.

Study 2 - Eleven sites in Wisconsin

Ground water quality surrounding 11 septic systems in different regions of Wisconsin was studied from 1971 to 1973 (Dudley and Stephenson, 1973). At least three monitoring wells were installed at each site, with many sites having seven or more wells. The wells were generally placed three to five feet below the water table. Nine of the sites had local geology consisting of outwash sands and a shallow depth to water table (< 15 feet). Each well was sampled at least seven times between 1971 and 1973 for NO_3-N, NH_4-N, organic N, total-N and other nutrients. High NO_3-N or total-N (>10 mg/l) was observed at all 11 sites in wells less than 25 feet from the discharging effluent.

Average NO_3 in ground water directly below the systems installed in sand was about 15 mg/l. No wells had NO_3-N concentrations greater than 10 mg/l or NH_4-N concentrations greater than 1 mg/l at a distance of 50 feet downgradient from the points of release. However, elevated NO_3 concentrations (5-15 mg/l) were common at a distance of about 30 feet from the points of effluent discharge. Nitrate concentrations varied significantly between sampling dates and between neighboring wells. Many wells near absorption fields had relatively low NO_3-N (1 to 5 mg/l), whereas nearby wells were often more greatly impacted (NO_3-N 15 to 40 mg/l).

Significant alteration of natural ground water flow direction was observed at 3 of the 11 sites due to artificial recharge from drainfields or absorption pits. A larger zone of contamination occurred as contaminated ground water flowed away laterally in all directions below the drainfields.

Study 3 - Seventeen Sites in Wisconsin

Eight septic systems receiving wastes from households using phosphate-built detergents and nine systems receiving wastes from households using carbonate-built detergent were studied in five counties in south-central Wisconsin (Alhajjar et al., 1987). All systems were new and located in coarse textured soils over shallow aquifers. Wells were installed adjacent to the edge of the drainfields and at 10 and 20 feet downgradient of the drainfields. All wells were placed near the surface of the water table. While background total N concentrations were less than 3 mg/l, total ground water N concentrations one foot downgradient averaged about 22 mg/l (ranging between 0.1 and 170 mg/l) and decreased to an average of about 7 mg/l when 20 feet downgradient from the drainfield.

Study 4 - Four Sites in Wisconsin

Eighteen ground water observation wells were installed in the immediate vicinity of each of four septic tank effluent soil disposal systems in Wisconsin (Walker et al., 1973b).

System 1: Ammonium and NO_3 concentrations were high near the absorption field in the upper 30 cm of ground water. At 100 feet downgradient of the absorption field, NO_3 concentrations were 10 mg/l. Wells screened 5 feet below the water table near the system had NO_3-N concentrations of 15 mg/l decreasing to between 1 and 4 mg/l further downgradient.

System 2: Saturated soil and a resultant lack of nitrification caused NH_4 to be the major N species in the ground water. Ammonium-N concentrations decreased with increasing distance from the system because of NH_4 adsorption to soil colloids.

System 3: Water percolating from the seepage bed in this system perched above a clay layer 8 m below the bottom of the seepage bed. Water in this perched system had very low NH_4 concentrations and NO_3 concentrations exceeding 10 mg/l.

System 4: Nitrate-N concentrations in system 4 were up to 40 mg/l in the upper 30 cm of the aquifer near the drainfield, but decreased to approximately 20 mg/l at 235 ft. downgradient. Samples taken 5 feet below the water table near the system had almost 15 mg/l NO_3-N.

Walker et al. (1973b) noted that high NO_3 contents are usually restricted to the upper part of the aquifer. Nitrate contributions from septic systems in sands was stated in Walker, et al. (1973b) to be approximately equal to those from natural sources if one dwelling with a septic tank is found on 6 acres of land.

Study 5 - Five Sites in Wisconsin

Five septic systems were analyzed and intensively monitored in Wisconsin (Bouma et al., 1972). High N contents (80 mg/l) were found in the septic tank effluent ponded in the seepage beds, with about 85 percent of the N in the organic form. Depending on the oxidative condition of the soil, the NH_4 was usually converted to NO_3. One system had perched water 30 feet below the seepage bed with NO_3-N concentrations of 15 mg/l. Ground water below another system had NO_3-N concentrations averaging 30 mg/l after 12 feet of percolation. In the absence of nitrification, ground water below another septic system had fairly high NH_4. Bouma et al. (1972) concluded "There is no doubt that septic tank absorption beds in sandy soil introduce NO_3 into the ground water if the depth of unsaturated soil below crusted seepage beds is more than 3 feet." The less the depth of unsaturated soil below the seepage bed, the more likely the chance of introducing NH_4 to ground water.

Study 6 - Seventeen Sites Around Eight Lakes in New York State

Forty-three test wells were installed around 17 septic systems near eight lakes in New York State (Chen, 1988). Information on system design, placement, and loading was not provided. Three wells had NO_3-N above 1 mg/l, with the highest NO_3-N concentration being 3.7 mg/l. Ammonium-N was above 1 mg/l in ten wells at six sites, with the highest concentration being 12 mg/l. Total inorganic N (NH_4 + NO_3) exceeded 10 mg/l in two wells (12.0 and 14.7 mg/l). Both of these wells were located within 40 feet of the septic system discharge point.

Study 7 - Nine Large Drainfield Systems in Minnesota

Permits are required by the Minnesota Pollution Control Agency for drainfield systems designed to handle more than 10,000 gallons per day. These drainfields treat wastes from a variety of types of developments, including condominium and townhouse developments, mobile home parks and small towns. Often more than one drainfield is used to discharge waste material. A limited amount of ground water monitoring has been required around the drainfields since the mid-1980's. Anywhere from one to seven monitoring wells have been installed around the drainfields, with an average of four wells per site which were sampled three to four times each year. In most cases, at least one well has been placed upgradient of the drainfield and at least two wells placed in an attempt to intersect the plume. The wells are screened at the top of the water table.

Water quality data from wells around the permitted drainfields were examined for this report. Data were available for nine sites. At four out of the nine sites, NO_3-N concentrations in at least one well were found to be above 10 mg/l during at least one sampling event. At two of the sites, the wells with NO_3-N in excess of 10 mg/l were located more than 100 feet from the drainfield. Ammonium concentrations were measured at six of the sites. While the NH_4 concentrations were usually less than 1 mg/l, occasional concentrations above 5 mg/l were reported.

Some sites had no clear evidence of ground water NO_3 from the large drainfields. The site with the worst NO_3 contamination had wells 50 and 125 feet downgradient of the drainfield having median NO_3-N concentrations of 16 and 23.5 mg/l, respectively. The apparent background NO_3-N concentration at this site was less than 0.5 mg/l. It should be noted that the well depth, distance of the well from the drainfield, and the location of the well with respect to ground water flow directions are all important variables that can help to explain the water quality around drainfields. It can be quite difficult to intercept the drainfield effluent plume while utilizing only a couple of monitoring wells. It is also difficult to quantify background NO_3 concentrations with only one monitoring well placed upgradient of the system.

Study 8 - A High School Drainfield in Wisconsin

Monitoring wells were placed surrounding a drainfield serving Wisconsin Heights High School (Polkowski, et al., 1970). Nitrate-N was measured at concentrations up to 21 mg/l in ground water at a distance of 15 feet from the tile field. The NO_3-N concentration was found to be 2.4 mg/l and 2.0 mg/l in May 1967 and 1968, respectively, at a distance of 265 feet from the tile field when the background level was between 0.8 and 1.0 mg/l for corresponding periods. Dilution appeared to be the factor responsible for reducing NO_3 concentrations. Nitrate was not observed to be greater than 10 mg/l at any time at distances greater than 100 feet from the lower edge of the absorption field.

Impacts Under Housing Developments Using Septic Systems - Case Studies

In the last section of the report it was shown that a single on-site wastewater treatment system does contribute to ground water NO_3 and may contribute to ground water NH_4 but that through dilution and dispersion there is often minimal degradation of ground water quality at points several yards away from the systems.

While the first several systems in an area may not create obvious increases in well water NO_3 concentrations, the cumulative impact of tens or hundreds of systems may be more noticeable as the dilution capacity of the aquifer becomes overwhelmed. The degree of impact on well water will also depend on well depth and construction. Existing case studies of ground water quality in developments utilizing septic systems will be described in the following pages. Results of six of the most pertinent studies are represented on Table I-2.

Specific information about the types of systems installed and construction methods were not provided in most of the studies and it is possible that many systems in these studies would not meet current design and construction codes in Minnesota. Systems properly constructed in Minnesota should result in near complete conversion of NH_4 to NO_3 in the soil below the system. The N would then be in a form which could be lost through denitrification in subsequent anaerobic soil zones.

Case 1 - Eau Claire County, Wisconsin (Tinker, 1991)

Domestic wells in five unsewered subdivisions were sampled for NO_3 in Eau Claire and La Crosse County, Wisconsin. The subdivisions were chosen so that little or no known agricultural sources of NO_3 were upgradient. The soils, depth to water, well depth, number of systems, mean lot size, number of samples and NO_3 concentrations at each of the sites are listed in the table below:

Subdivision	Soils	Mean Depth To Water Table (ft)	Mean Depth of well	# Systems	Mean Lot Size (acres)	Average# Samples Taken	Average Nitrate-N mg/l
Sandy Knolls	Sand Loamy Sand	64	93	50	0.5	15	5.7
Pine Grove-Deer Park	Sand Sandy Loam	30	61	70	1.1	42	3.7
Oak Park	Sandy Loam Loamy Sand	75	120	128	0.6	37	5.7
Lowes Creek	Loamy Sand	34	93	33	1.3	19	1.7
Briarwood	Loamy Sand	33	74	45	1.2	15	2.0

While NO_3-N concentrations were in excess of 10 mg/l in several wells from the Sandy Knolls and Oak Park development, very few to no wells exceeded 10 mg/l NO_3-N in the other three subdivisions. Nitrate concentrations were generally found to increase from the upgradient side to the downgradient side of each subdivision. A direct relationship was also found between mean lot size and the highest NO_3 concentration. According to the relationship observed, the highest N concentration would exceed 10 mg/l when the minimum lot size is less than 1.1 acres.

Case 2 - Lake St. Croix Beach, Minnesota (unpublished)

Eleven wells in Lake St. Croix Beach, Minnesota were sampled by the Minnesota Pollution Control Agency in July and November 1987 for several parameters, including NO_3. Several hundred homes exist in this community with an average density of 2.3 houses per acre, each with its own septic system and well. The underlying surficial sand aquifer has a relatively shallow water table, with many wells that are only 25 to 40 feet deep. Many of the homes and septic systems in the area are over 25 years old. Nitrate-N concentrations in the sampled wells ranged from 0.01 to 10 mg/l, with both a mean and median concentration of about 4.2 mg/l. Ammonium plus organic N was less than 0.3 mg/l in all eleven wells. At this location, background NO_3-N concentrations

TABLE I - 2: Studies of the impact of residential development septic systems on ground water nitrate concentration.

LOCATION/ REFERENCES	GEOLOGY	# HOMES IN AREA	AVERAGE LOT SIZE	# OF WELL SAMPLING POINTS	NITRATE IN GROUND WATER	CONCLUSIONS
Lake St. Croix Beach, MN MPCA, 1987	Surficial sand aquifer	> 200	0.4 Acre	11	Range 1-10 mg/l Mean 4.2 mg/l Median 4.2 mg/l	• Nitrate-N gradually increased to 10 mg/l 2,000 feet downgradient from the edge of development. • Ammonium plus organic-N was less than 0.3 mg/l in all wells.
Roscoe, Illinois Wehrman, 1983	Surficial sand aquifer	4,523	0.6-0.7 Acre	300	Median 5.7 to 6.9 mg/l, spring to fall.	• During most months, 4 to 7 % of wells > 10 mg/l nitrate-N. • Rarely did any wells exceed 14 mg/l nitrate-N. • Background nitrogen was difficult to determine due to nearby agricultural land uses.
Yarmouth, Massachusetts Nelson et al., 1988	Surficial sand, 45 ft. saturated thickness	1,800	0.3 Acre	Downgradient public supply wells	Average of 4.2 mg/l.	• Nitrate in public supply wells downgradient of the homes has been increasing over the last 25 years due largely to septic systems.
Portland, Oregon 30 sq. mile area Quan, et al., 1974					< 1 mg in sewered area. 4-12 mg/l in unsewered area.	• Nitrate concentrations were much higher in unsewered residential areas than an adjacent sewered area. 80% of homes in unsewered area were served by cesspools.
Eau Claire, Wisconsin Tinker, 1991	Mostly sand and loamy sand soils over outwash aquifers.	# Systems A) 50 B) 61 C) 128 D) 33 E) 45		# Samples A) 15 B) 42 C) 37 D) 19 E) 15	Mean NO3-N A) 5.7 B) 3.7 C 5.7 D) 1.7 E) 2.0	• Highest concentration reported was 21.6 mg/l NO3-N. • No wells exceeded 10 mg/l NO3-N in 3 of the 5 subdivisions. • Lot sizes should be greater than 1.1 acre to reduce likelihood of nitrate in downgradient exceeding 10 mg/l.
Portage County, Wisconsin Harmsen, 1989	Surficial sand aquifer	A) 64 B) 136	A) 0.6 acre B) 0.4 acre	A) 97 B) 93	A) Upgradient side 3 mg/l. Downgradient side 8-14 mg/l. B) High background N No clear impacts	• Location of wells in relation to effluent plumes is important. • Septic systems resulted in higher nitrate on downgradient side. • Upgradient N sources can overshadow septic system impacts..

appeared to be less than 0.3 mg/l. It appeared from this monitoring effort that NO_3-N concentrations increased downgradient and approached 10 mg/l at a distance of around 2,000 feet downgradient from the edge of development.

Case 3 - Roscoe, Illinois (Wehrmann, 1983)

A study of ground water NO_3 concentrations as affected by septic systems above outwash terrace deposits was conducted in and around the Village of Roscoe, Illinois (Wehrmann, 1983).

Roscoe and Rocton township villages had 13,521 people living in 4,523 dwelling units at the time of the study, with an average housing density of 1.5 lots per acre. The city of Roscoe had the highest housing density in the area with an average of 1.8 lots per acre. Each home maintains a private on-site water well and septic system.

Over 1,100 ground water samples from over 300 wells were analyzed for NO_3-N in the spring and fall during 1982. The water table in the hydrogeologically sensitive study area can be found at depths from 25 to 30 feet. Average NO_3-N concentrations varied from a low of 5.8 mg/l in the spring to a high of 6.9 mg/l in the fall. Median NO_3 concentrations were similar to the means. During most months of sampling, four to seven percent of wells had NO_3-N in excess of 10 mg/l and very rarely did concentrations exceed 14 mg/l. In an area with several high NO_3 wells, an abandoned well was located 3.5 feet from a septic field line. Nitrate concentrations began to decline after the well was plugged.

Background NO_3 was thought to be around 3 mg/l, but was difficult to determine due to nearby agricultural influences and septic influences. Nitrate concentrations were found to be highest immediately after heavy rains following a dry period. With continued rain, the ground water became diluted and NO_3 concentrations decreased.

Modeling showed that with background NO_3-N of 3 mg/l and housing densities of 2 to 3 homes per acre, the drinking water standard for NO_3 would be exceeded in less than one mile down gradient from the edge of the development. The study concluded that under present development the <u>average</u> NO_3 concentration should remain below the drinking water standard.

Case 4 - Two Subdivisions in Portage County, Wisconsin (Harmsen, 1989)

Two subdivisions in Portage County, Wisconsin were studied to determine the nature of ground water contamination from septic systems. Both domestic wells and multilevel sampling wells were used to assess the flow system and ground water impacts.

Jordan Acres Subdivision

The Jordan Acres subdivision began in the late 1960's and consisted of 64 homes with an average lot size of 0.6 acres at the time of study. The average depth to the water table was 25 feet. Sand and coarse sand aquifer materials are found at the site. The estimated seepage velocity of the upper 25 feet of the aquifer ranged from 1 to 2 feet/day. Background NO_3-N concentrations were usually below 3 mg/l. Most of the domestic wells were screened 5 to 15 feet below the water table.

Nitrate-N concentrations on the upgradient side averaged about 3 mg/l. Nitrate-N concentrations were noticeably higher in the downgradient half of the subdivision with several wells exceeding 10 mg/l. Great variability in NO_3 was found which appeared to be related to the location of wells in relation to effluent plumes. Nitrate distribution was quite variable vertically within the aquifer and also quite variable with time in certain wells.

Village Green Subdivision

The Village Green Subdivision also began in the late 1960s and consisted of 136 homes with an average lot size of 0.4 acres at the time of study. The Village Green subdevelopment has mostly irrigated agriculture upgradient involving mostly potato production. Since NO_3-N concentrations upgradient of the development were very high (averaging 20 mg/l) no clear impact was observed from the septic systems.

Case 5 - Yarmouth, Massachusetts (Nelson et al., 1988)

A mass balance analytical N loading model was used in Massachusetts to simulate historical land development patterns and the migration of N to public supply wells. The public supply wells have shown a steady increase in NO_3-N concentration over their 25 year history, with 1984 concentrations at 4.2 mg/l. Eighty percent of the land upgradient of the wells is developed (1,800 housing units) with an average lot size of 0.3 acres, each with a private septic system. The average saturated thickness of the underlying aquifer is approximately 45 feet.

The model considered N from septic systems, lawn fertilizer, precipitation and road runoff and was calibrated with historical land management information. The model predicted that under current land use conditions the NO_3 concentration at the public supply wells would not exceed 5 mg/l for at least 30 years. In other words, NO_3 concentrations in the ground water in this region have nearly reached equilibrium conditions with current N in recharge.

N loading parameters determined by the modeling effort were used to determine the minimum lot size required to site a new house with a septic system and maintain total expected N concentrations in ground water recharge below 5 mg/l. For a 3 bedroom home with a 5,000 square foot fertilized lawn where storm water recharged on-site, the minimum lot size needed would be 1.7 acres assuming 2 people/bedroom.

Case 6 - Portland, Oregon (Quan et al., 1974)

Ground water quality as affected by subsurface sewage disposal was studied in a 30 square mile area near Portland, Oregon. The 30 square mile area was unsewered, with an estimated 80 percent of all wastewater disposal systems being cesspools. Natural recharge is reduced in this area due to paving. Nitrate contaminated recharge remained near the surface of the water table where it moved laterally until eventually discharging into the Columbia Slough South Arm.

Water samples from wells adjacent or upgradient of the unsewered area and from deeper aquifers within the unsewered area generally had NO_3-N concentrations less than 1 mg/l. Shallower wells and springs in the unsewered areas had NO_3 concentrations ranging from 4.7 to 11.9 mg/l in 1974. The South Arm Slough, whose primary water source is shallow ground water, was also found to be high in NO_3, especially in areas downgradient from the unsewered area.

Case 7 - Tacoma, Washington (Dewalle et al., 1980)

Nitrate analyses of 98 wells were obtained over a 35 year period from both sewered and unsewered areas in a populated river basin south of Tacoma, Washington. The results from these analyses showed significantly higher ground water NO_3 concentrations in areas served by septic tanks and drainfields than in sewered areas. Nitrate concentrations in the unsewered areas have been increasing with time.

Case 8 - Houston County, Texas (Brooks and Cech, 1979)

In Houston County, NO_3 concentrations were measured to determine sources of NO_3 in drinking water. The implications were that the primary factors associated with NO_3 contamination were depth of the wells, the type of well constructed (drilled vs. dug) and the relative location of the well from sources of organic wastes. Wells placed within 16 feet of septic tanks often had NO_3-N in excess of 10 mg/l. The relationship between the distance to sources of human pollution such as septic tanks and the level of NO_3's was found to be statistically significant ($P < 0.05$).

Case 9 - Twin Cities Metropolitan Area (Metropolitan Council, 1979)

During the period of 1959 to 1964, the Minnesota Department of Health surveyed NO_3 concentrations in domestic well supplies of several communities in the Twin Cities area that used septic systems. Metropolitan Council (1974) reviewed the results of this survey and concluded that in areas of highly permeable soils, developments on lots averaging less than one acre in size have a greater incidence of NO_3 contamination than those developments with larger lots.

SEPTIC SYSTEM BEST MANAGEMENT PRACTICE OPTIONS

Best Management Practices (BMPs) for minimizing the impact of NO_3 contamination of ground water from septic systems generally fall into two major categories:

1) Using systems that promote significant denitrification to occur within the treatment system, and

2) Siting restrictions that ensure adequate dilution of the septic effluent by natural recharge and the ground water itself.

Other BMPs could include promoting vegetative nutrient uptake of septic effluent and use of holding tanks instead of on-site treatment systems. **The following discussion will only refer to BMPs specifically for minimizing NO_3 impacts on ground water.** Many other BMPs exist for septic system design, construction and siting that will not be referred to in this report (refer to MPCA Rules Chapter 7080).

Alternative Systems for Nitrogen Management

Several types of systems have been reported in the literature which have shown a potential of greatly reducing N in septic system effluent. Some systems promote denitrification before the effluent is released to the drainfield (Piluk and Hao, 1989; Laak, 1986; Sandy et al., 1987; Lamb et al., 1987). One or more tanks are added to a conventional system where the nitrification and subsequent denitrification take place. The carbon source needed for denitrification can be supplied by the wastewater in certain systems, and in other systems an additional carbon source such as methanol can be added to aid in denitrification.

One example of a denitrification promoting system is a RUCK system described in Laak (1982, 1985, 1986). The RUCK system separates plumbing drainlines into two waste streams. The N rich stream (toilet wastes) is treated using a septic tank followed by an underdrained aerobic sand filter where nitrification occurs. All or part of the remainder of the wastewater, called graywater, contains an abundance of organic carbon and is treated in a separate septic tank to remove settleable solids. The two streams are brought back together in an anaerobic upflow rock-filled tank where biological denitrification occurs. The rock filter effluent is disposed in conventional seepage trenches.

The state of Wisconsin funded a study to evaluate many different types of alternative systems shown to have potential for N removal (Ayres and Associates, 1991). Each system was evaluated and ranked based on its N removal efficiency, effluent quality, reliability, construction and operation costs, owner acceptance, frequency and complexity in maintenance, installation requirements, and its current stage of development. In addition to review of published information, interviews of researchers and site visits were made in order to best evaluate the systems. Authors of the ensuing report concluded that the technology for N removal as applied to on-site systems is relatively untested and little data exist regarding N removal efficiency, reliability, consistency and costs. However, several systems demonstrated nitrogen removal efficiencies between 60 and 90 percent. Costs of most alternative systems were found to generally be between two and four times the cost of a conventional system.

The specific systems evaluated are listed below in the order of highest overall ranking score to lowest (from Ayres and Associates, 1991):

1. Peat filter
2. Recirculating sand filter
3. Recirculating sand filter with an anaerobic filter
4. Recirculating sand filter with an anaerobic filter and carbon source
5. Recirculating sand filter and rock storage filter
6. RUCK
7. Non-Water carriage toilets
8. Blackwater holding tank
9. Extended aeration package plants
10. Package trickling filter plant

A description of each system and associated references are provided in Ayres and Associates (1991). The septic tank/peat filter ranked the highest of the alternative systems. This system, which needs more work in developing design criteria, consists of an in ground peat bed placed between the septic tank and soil infiltration system. As wastewater moves through the peat bed, the wastewater is converted to NO_3 and then lost through denitrification. The second ranked system, a recirculation sand filter system with an anaerobic upflow rock filter was stated to be reasonably well established. This system consists of a septic tank, submerged anaerobic rock filter, recirculation tank, and sand filter. Successive anaerobic and aerobic conditions exist in the different tanks and filters resulting in denitrification of the effluent.

While not ranked in the evaluation process, the study concluded that ion exchange appears to be a promising N removal mechanism, but has not been tested for wastewater applications. Reverse osmosis is another potential N removal process which needs further study.

A type of system commonly installed in Minnesota that has shown potential for N removal is a mound system. Mound systems are modified versions of conventional septic tank/soil absorption systems. A mound of permeable soils is built above the native soil to increase the amount of good soil available to absorb and treat the septic tank effluent. The effluent is released in a bed within this mound of soil. Mound Systems are commonly installed in Minnesota, especially in areas of shallow bedrock, shallow water table, and/or fine textured soils. While not usually installed for the purpose of reducing NO_3 leaching to ground water, significant denitrification has been found to occur below mound systems.

In the sand fill under mound systems, aerobic conditions usually exist and NH_4 is readily converted to NO_3. Over 30 mound systems were evaluated in Wisconsin (Harkin et al., 1979). Forty-four percent of the NO_3 formed in the sand fill and underlying soil denitrified as it passed through the natural soil surface and about two feet into the natural soil. Denitrification takes place due to the saturated (anaerobic) conditions often found at the original soil surface. Average NO_3-N concentrations in the unsaturated sand fill was 45 mg/l. At a depth of about 2 feet below the original soil surface average NO_3 concentrations were 20 mg/l. Harkin et al. (1979) concluded that dose volumes of mound systems should be calibrated to dose 2-4 times daily to minimize NO_3 entering ground water.

Mound systems have been shown to be effective in treating other contaminants as well. Mound systems generally cost 20 to 80 percent more than conventional septic systems varying with a number of factors.[1] Denitrification under mound systems would probably be reduced in conditions of very coarse topsoil, where saturated-anaerobic soils at the natural soil surface are much less likely.

An alternative to denitrification systems is to separate out the high N content toilet wastes from the lower nitrogen content shower, laundry, and kitchen sink wastes. The toilet wastes can be stored in a tank where they are periodically removed and delivered to a municipal waste water treatment plant. The other water is treated in a conventional septic system. To minimize the volume of wastewater in the holding tank, low volume flush toilets or nonwater carriage toilets are recommended. The costs of pumping and transporting wastes from holding tanks can be prohibitively expensive for permanent residences, but may be a viable alternative for less frequently used dwellings. However, the less frequently used dwellings would not generally be major contributors of N to ground water.

Siting Limitations to Allow for Dilution

Gold et al. (1990) found NO_3 loading of ground water in Rhode Island from septic systems placed at a density of 2 per acre to be equivalent to losses below urea-fertilized silage corn with an effective rye cover crop. In the same study, minimal NO_3 leaching was found below forest and fertilized home lawns (flow weighted concentrations below 1.7 mg/l NO_3-N).

One of the most important siting limitations for minimizing NO_3 impacts on ground water from septic tanks is the density of systems in an area. There are two general approaches to the issue of density:

1. Place septic systems far enough apart so that dilution from local natural recharge alone is sufficient for keeping average concentrations of NO_3 at safe levels, or

2. Place septic systems far enough apart so that both dilution from local natural recharge and dilution from underlying ground water is sufficient for keeping NO_3 at safe levels.

The preferable concept for water quality protection, especially in areas of potentially large developments, would be to consider the water table as the lower boundary of the treatment system. Where external sources of NO_3 exist and/or where numerous septic systems are found, the eventual higher NO_3 concentration of the ground water (moving downgradient) will reduce the effective dilution capacity of the ground water. **Therefore, we cannot always rely on dilution from the underlying ground water to keep NO_3 concentrations at a safe level.** Some N contamination of ground water is unavoidable using conventional septic systems in coarse textured soils above shallow aquifers and no matter what lot size are used, there will likely be plumes of elevated NO_3's.

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[1]Personal communication with Dave Gustafson, University of Minnesota

However, if septic systems are placed far enough from other systems, dilution from natural recharge can prevent NO_3 concentrations from becoming excessive throughout the aquifer.

As previously discussed, there are numerous factors, other than housing density, that affect the amount of N movement to ground water. The net average N concentration of percolating water (effluent/natural recharge) was calculated for various amounts of natural recharge with the following assumptions:

1. A typical home in an area (average of three people per home) utilizing septic systems will discharge 135 gal/day of septic system effluent.

2. The N concentration in the effluent averages 50 mg/l,

3. The average N concentration in percolating "natural" recharge on the lot is 1 mg/l, and

4. No denitrification takes place between the drainfield and the water table.

Given the above assumptions, Table I-4 can be used in areas of sandy soils to estimate lot sizes needed to keep net average NO_3 concentrations percolating through a residential development area below 10 mg/l under various amounts of "natural" recharge.

The amount of recharge over an area can be quite complex and variable depending on precipitation patterns and timing, evapotranspiration from the growing plants, the type of soil, macropore development, and several other factors. The United States Geologic survey has estimated recharge rates from well hydrographs in many drift aquifers throughout the state. The recharge amounts derived by several of the USGS studies are summarized in Table I-3 below:

TABLE I-3 Recharge Rates Determined from Well Hydrographs for Various Sand Plain Aquifers by USGS.

Aquifer	County(s)	Years of Study	Number of Wells	Range in Recharge (in/yr)	Average Recharge (in/yr)	References
unspecified sand plain	Kanabec/Pine Carlton	1981	56		5.9	Myette, 1986
Buffalo aquifer	Clay/Wilkin	1977-78	18	2.4-8.8	4.7	Wolf, 1981
Pelican River Ottertail	Becker	1979-80		3.1-6.1		Miller, 1982
Pomme de Terre/ Chippewa River	Swift/Pope Stevens, Grant	1973-80		3.4-8.5	5.6	Soukup et al., 1984
sand plain	Stearns				8	Lindholm, 1980
sand plain	Stearns	1982-84	15	2.6-16.5	10.7	Delin, 1988
sand plain	Stearns	1980-82	12	1.2-15.1	6.0	Delin, 1986

Recharge rates based on USGS work, while quite variable from well to well in a given aquifer, appear to generally be in the range of about 3 to 10 inches per year in sand plain aquifers with an average recharge rate from all studies of 6.6 in/year. Delin (1986 and 1988) found that recharge through glacial till into buried drift aquifers was much less than in sand plain aquifers. Leakage rates in two studies ranged from 0.06 to 3.4 inches per year.

In unpublished work, Palen[1] calculated annual recharge rates from observation well hydrograph records through 1985 for 104 sites in 31 counties throughout the state. The average calculated recharge rate for all 104 wells was 7 inches (Standard Deviation of 2.3 inches). Similar to the USGS study results, the recharge rates calculated by Palen range between 4 and 10 inches for most wells. From the data, it appeared that there was on the average slightly less recharge in western Minnesota compared to eastern Minnesota, as might be expected with generally more precipitation in the eastern half of the state.

Assuming the state sand plain average recharge rate of 7 inches, the recommended minimum lot size for homes with septic systems is about 1.15 acres. **This sized lot is generally consistent with the other monitoring and modeling studies previously discussed, which suggested that lots sizes no less than 1 to 2 acres are needed to keep pervasive aquifer NO_3-N conditions below 10 mg/l.** It was determined from a modeling effort in Olmsted County that 2 to 2.5 acre lots are needed to keep overall N concentrations below 5 mg/l (Olmsted County, 1990).

TABLE I-4 Recommended minimum lot size for coarse textured soils needed to keep average N concentrations of recharging water below 10 mg/l for various amounts of natural recharge rates. Assumes 1 mg/l NO_3-N in natural percolation, 50 mg/l NO_3-N in septic effluent, 4 people/home and no denitrification.

Natural Recharge Amount (Per Year)	Recommended Minimum Lot Size (Acres)
3 inches	2.7
4 inches	2.0
5 inches	1.6
6 inches	1.35
*7 inches	1.15
8 inches	1.0
9 inches	0.9
10 inches	0.8
11 inches	0.75

* average recharge for Minnesota in sand plain aquifers.

[1] Personal communication with Barb Palen, Minnesota Geological Survey. October 1990.

Even with very large lot sizes, the upper part of the aquifer near the drainfield will likely have NO_3-N concentrations in excess of 10 mg/l. Long plumes of high NO_3 water can be found in ground water in areas where septic systems exist (Robertson et al. 1991). Therefore, a ground water flow dependent strategy for the placement of wells and septic systems in a subdivision is important to minimize septic system impacts on neighboring wells.

Recharge rates, and thus dilution from percolating precipitation, will be much less than 7 inches for medium and fine textured soils. However, there is a greater potential for denitrification to occur in the finer textured soils. In addition, there are fewer reported incidences of high nitrate in ground water from septic systems in medium and finer textured soils. **All new residential developments planning to use septic systems should be zoned to allow enough space on the lot to install a replacement system, which would effectively increase required lot sizes.** This will have benefits of greater dilution of N and is a long-term approach for treating septic system effluent.

If the average number of residents per home is greater than three, then lot sizes for coarse textured soils should be larger than stated in Table I-4. If N fertilizers are used extensively over the lots, the lot sizes would need to be somewhat larger in order to ensure average percolate less than 10 mg/l. When systems that allow for denitrification are used, lot sizes could be reduced and still keep the average effluent NO_3-N concentration less than 10 mg/l. If the denitrification promoting alternative systems prove to be feasible, NO_3 contamination could be kept to a minimum, even with relatively small lot sizes. **The use of denitrification promoting systems is a preferable approach for minimizing NO_3 compared to regulating lot sizes. Larger lot sizes encourage urban sprawl and reduce the options for hooking up to city sewer systems at some future date.**

Other siting considerations for septic systems include:

1. Do not locate a significant number of septic systems within community well head protection areas.

2. Septic systems should be located downgradient of domestic drinking water wells. The Minnesota Department of Health requires new wells to be placed at least 50 feet from all septic tanks and septic system drainfields. A 100 foot setback distance from the drainfield is required if there is less than 50 feet of casing and no impervious material at least 10 feet thick is penetrated. Robertson, et al. (1991) concluded that typical minimum permissible distance-to-wells regulations (75-115 ft) should not be expected to be adequately protective of well-water quality for sandy aquifers.

Use of Vegetation for Nitrogen Uptake

Native woody vegetation in a variety of habitat types has some ability to increase N uptake in the presence of excess N from septic drainfields (Ehrenfeld, 1987). However, the reduced ability to harvest native woody vegetation and a shorter growing season contribute to smaller N uptake as compared to grass systems.

SEPTAGE

Septage is generally defined as the liquid and solid material pumped from a septic tank or cesspool during cleaning. Over a period of time, sludge and scum can build up to a point where it occupies from 20 to 50 percent of the total septic tank volume. Septage characteristics vary widely from one location to another. From a number of documented studies EPA (1984) found that the amount of septage generated per person per year to be generally between 50 and 70 gallons. The EPA calculated mean concentration for total N in septage is 677 mg/l. Therefore, according to the EPA figures, an average amount of N per person going to septage each year is 0.34 lbs. (assuming 60 gal/year at 677 mg/l).

With approximately 1,200,000 people using septic systems in Minnesota for waste treatment/disposal, the total amount of N released in septage N per year would be 408,000 lbs calculated from the above figures. Much of the septage pumped in the seven county metropolitan area is dumped into the sewer system and is then treated with all other metropolitan waste. Outside of the metropolitan area, a great majority of the septage generated is applied by the septage pumpers to vegetated fields per an agreement between the landowner. When not injected, much of the N (in the ammonia form) in septage will volatilize during and following surface application to the land. Because of the losses into sewer systems and volatilization, the amount of N from septage that moves into the soil in the state is likely to be less than 100,000 pounds per year.

Despite the fact that the total amount of N in the state from septage application is relatively low, its application to fields can potentially pose localized ground water NO_3 problems. Often the land owner that allows septage application has no knowledge of the amount of septage N applied to the fields. Also the septage is often unevenly applied in the fields, with some areas receiving large amounts of septage-N and other areas receiving little or none. Therefore, it is difficult for the farmer to account for this N and the field is often fertilized as if no other N was applied.

Septage applicators need to be aware of proper application techniques, set back distances, and nutrient contributions from septage. This information is taught at MPCA Land Application of Septage workshops. However, not all counties require certification of septage pumpers. Best Management Practices for septage disposal are discussed at the MPCA workshops and are discussed in detail in EPA's handbook of Septage Treatment and Disposal (EPA, 1984).

CURRENT POLICY REGARDING SEPTIC SYSTEMS AND NITROGEN

The minimum standards and criteria for the design, location, installation, use, and maintenance of individual sewage treatment systems are set in Minnesota Pollution Control Agency Chapter 7080. Counties have the option of adopting Chapter 7080. It was recommended in the Minnesota Nonpoint Source Ground Water Strategy (MPCA, 1989) that Minn. Rule Ch. 7080 be made mandatory statewide by 1992, with a change in statute and rule. Ch. 7080 is currently mandatory in shoreland areas.

While the adoption of Chapter 7080 will minimize ground water contamination from several on-site wastewater treatment pollutants, it does not adequately address prevention of NO_3 ground water contamination. However, if systems are sited according to Chapter 7080, most of the NH_4 should be converted to NO_3 and subsequently little NH_4 should reach ground water. Other than mound systems, no systems (or very few) designed for significant N losses are being installed in Minnesota, and it is up to each individual county to set minimum lot sizes where septic tanks are utilized. However, to the author's knowledge, few counties have set restrictive lot size requirements for new developments.

As previously discussed, permits are required for septic system drainfields designed for over 10,000 gallons per day. While permitting of these systems does not ensure protection of ground water, the MPCA assistance provided through the permit review process will help minimize adverse impacts on **drinking water supplies** that large on-site systems could cause.

Other than providing training opportunities and technical assistance to septage pumpers, the state does not have a policy with regards to septage disposal. The U.S. EPA has proposed federal rules for proper disposal of septage. The final rule is to be issued in January 1992.

SUMMARY

Over 400,000 Minnesota households dispose of wastewater into septic tanks. Septic systems are not designed to remove N, and NO_3 is usually the contaminant of greatest concern below a properly designed, sited, and constructed septic system. On average, for each person using a septic system about 45 gallons/day of wastewater with a total N concentration of about 50 mg/l will be released into the subsoil (7 lbs N per person annually). For Minnesota, it is estimated that between 6.6 to 10.2 million pounds of N are released annually in septic system effluent.

In oxygenated soil conditions, NO_3 is the form of N that will move through the soil below the soil absorption system. Where very moist soils exist below the system or where improperly constructed systems are found, NH_4 may move through the soil towards ground water. Denitrification is possible in the soil under certain conditions. The soils, climate, and vegetation are important variables that can affect the form of N, N losses, and dilution of effluent.

Impacts of septic systems on ground water will primarily depend on the N loading to the aquifer, diluting capacity of the aquifer, and the potential for denitrification in the soil. The diluting capacity of an aquifer is reduced when numerous septic systems exist in an area.

From 66 individual septic systems monitored for ground water impacts in numerous studies in Northern U.S. and Canada, the following generalizations can be made about the nature of N contamination from individual on-site wastewater treatment systems:

1) Nitrate-N concentrations are often between 10 and 40 mg/l at the surface of the water table directly below septic absorption systems.

2) Nitrate concentrations are highest at the water table surface near the points of effluent release and decrease substantially with depth.

3) Dilution and dispersion result in a decrease in ground water NO_3 concentrations downgradient so that NO_3-N is usually below 10 mg/l within 50 to 100 feet from the absorption field. Often times minimal N contamination is observed less than 20 feet from the system. In aquifers with a low potential for dispersivity, long narrow plumes can result with sharp lateral and vertical boundaries.

4) Highly elevated NH_4 can be found in ground water below septic systems, especially where saturated conditions are found immediately below the drainfield.

While the first several septic systems in an area may not create obvious increases in well water NO_3 concentrations, the cumulative impact of tens or hundreds of drainfields in a housing development are more noticeable. Aquifer NO_3-N concentrations between 5 and 15 mg/l often exist in the downgradient side of such developments when background NO_3 is low. One of the critical factors affecting NO_3 concentrations is average lot size. Average lot sizes less than 1 to 2 acres have a great potential to result in some high NO_3 wells in housing developments utilizing on-site systems. Hydrogeological controls and other nearby sources of N greatly affect NO_3 concentrations in such developments.

Several different types of systems have been developed which promote denitrification, resulting in N losses of between 50 and 95 percent. These systems are currently two to four times more expensive than conventional systems and further testing and evaluation of these systems is needed.

Reducing the chance of NO_3 impacted wells from septic systems can be accomplished by installing systems in lots which are large enough to allow adequate dilution from percolating natural recharge. In order to keep the net average percolating recharge/effluent NO_3-N concentrations below 10 mg/l, the suggested minimum lot size is generally between 1 and 3 acres, depending on the amount of annual recharge (precipitation, soil, slope) and N movement to ground water from other sources on the lot. Even with large lots, consideration of ground water flow direction in relation to septic system placement is important to reduce the chance of well interception of effluent plumes.

The amount of N in septage generated in the state is estimated to be about 360,000 pounds. Less than one-third of this amount ends up moving below the soil surface. While the total contribution of N statewide from septage is very small, localized ground water N problems can result from improper application or when not accounted for by the field operator. By developing state rules along with requiring training and certification of septage applicators, the risk of ground water contamination from septage can be reduced.

RECOMMENDATIONS

1. **It is recommended that the MPCA and University of Minnesota, (working with the Individual Sewage Treatment Advisory Committee and State of Wisconsin), further evaluate, test, and develop denitrification promoting systems.** Influent and effluent water quality should be monitored in each test system installed. Each system should be evaluated for costs of installation and maintenance, N reduction, other pollutant reduction, and overall system performance. Based on the results of this work and on alternative system testing in other states, recommendations should be made regarding the feasibility of using these systems on a more widespread basis in Minnesota.

2. **Maximum septic tank densities for new housing developments should be set by each county so that the septic tank effluent is adequately diluted from natural recharge.** Based on the figures obtained and derived in this study, the minimum lot size for coarse textured soils should be between one and three acres for conventional systems in order to keep the net average concentration of effluent and recharge below 10 mg/l. All new residential developments planning to use septic systems should be zoned to allow enough space on the lot to install a replacement system. If an alternative denitrification system is found to be effective and feasible from the recommended future study (#1), minimum lot sizes could be set substantially less than one to three and a half acres when such denitrification systems are installed.

3. Replacement systems in lots less than one acre should be of the type that is recommended following the research in recommendation #1. Upon further testing, denitrification systems should be required for large drainfield systems.

4. Standards should be set for NH_4 in ground water. In order to minimize NH_4 and other pollutants from reaching ground water it is recommended that **Chapter 7080 be made mandatory statewide.** It is recommended that existing septic tanks without soil absorption systems be reconstructed to allow for greater soil treatment of waste.

5. **Certification of site evaluators, designers, installers, pumpers and inspectors should be mandatory statewide.** Pumpers should be required to report to the land owners and county officials the amount of N applied and the extent of coverage from land applied septage. Counties should set aside land that could be used exclusively for septage application.

6. Ground water flow directions should be determined in new developments using septic systems. This information should be considered in the placement design of lots, wells, and septic systems so as to minimize well intersection of contaminant plumes.

REFERENCES

Adam, J.W.H. 1980. Health aspects of nitrate in drinking water and possible means of denitrification. Water SA. 6:2:79-84.

Alhajjar, B.J., J.M. Hartkin, and G. Chesters. 1987. Chemical tracers of pollution plumes from septic systems receiving two types of detergents. Proceedings of the FOCUS Conference on Northwestern Ground Water Issues. May 5-7, 1987. Portland, Oregon.

Alhajjar, B.J., S.L. Stamer, D.O. Cliver, and J.M. Harkin. 1988. Transport modelling of biological tracers from septic systems. Wat. Res. 22:7:907-915.

Ayres and Associates 1991. On-site nitrogen removal systems research/ Demonstration Project. Phase I report prepared for Wisconsin Department of Industry, Labor and Human Relations. Owen Ayres and Associates, Inc. Madison, Wisconsin. 34 pp.

Bauman, B.J. 1985. Ground water impacts of septic systems and their regulation in Montana. From Bauman, B.J. and M. Schafer. 1986. Estimating Ground Water Quality Impacts from On-site Sewage Treatment Systems. ASAE Pub. 7-85. St. Joseph, Michigan, pp. 285-294.

Bennett, Edwin R. and K. Daniel Linstedt. 1975. Individual home waste water characterization and treatment. Colorado State University, Ft. Collins, Colorado.

Biswas, N. and R.G. Warnock. 1985. Nitrogen transformations and fate of other parameters in columnar denitrification. Wat. Res. 19:8:1065-1071.

Bouma, J., W.A. Ziebell, W.G. Walker, P.G. Oleatt, E. McCoy and F.D. Hole. 1972. Soil absorption of septic tank effluent - a field study of some major soils in Wisconsin. University of Wisconsin - extension and geological and natural history survey information circular number 20.

Brooks, Diana and Irina Cech. 1979. Nitrates and bacterial distribution in rural domestic water supplies. Wat. Res. 13:33-41.

Canter, Larry W. and Robert C. Knox. 1985. Septic Tank System Effects On Ground Water Quality. Lewis Publishers, Inc., Chelsea, Michigan. 333 pp.

Chen, Min. 1988. Pollution of ground water by nutrients and fecal coliforms from lakeshore septic tanks systems. Water, Air, and Soil Pollution 37:407-417.

Cogger, Craig. 1988. On-site septic systems: The risk of ground water contamination. Journal of environmental health. 54:1:12-16.

Cohen, S. and Wallman, H. 1974. Demonstration Of Waste Flow Reduction From Households. EPA report 670/2-74-071.

Converse, James C., Robert L. Siegrist, and Damann L. Anderson. 1984. Onsite treatment and disposal of restaurant wastewater. University of Wisconsin - extension small scale waste management project paper 10.13. Madison, Wisconsin. 38 pp.

Delin, G.N. 1986. Hydrogeology of confined-drift aquifers near the Pomme De Terre and Chippewa Rivers, Western Minnesota. U.S. Geological Survey Water-Resources Investigations Report. 86-4098. 90 pp.

Delin, Geoffrey N. 1990. Geohydrology and water quality of confined drift aquifers in the Brooten-Belgrade area, west central Minnesota. U.S. Geological Survey Water-Resource Investigation Report 88-4124. 138 pp.

DeWalle, Foppe B., Russell M. Schaff, and Jack B. Hatlen. 1980. Well water quality deterioration in central Pierce County, Washington. American Water Works Association. September 1980. p. 533-536.

Dudley, John G. and David A. Stephenson. 1973. Nutrient enrichment of ground water from septic tank disposal systems. An Inland Lake Renewal and Shoreland Management Demonstration Project Report for the Upper Great Lakes Regional Commission. 131 pp.

Eastburn, R.P. and W.F. Ritter. 1985. Denitrification in on-site wastewater treatment systems - a review. Proceedings of the Fourth National Symposium on Individual and Small Community Sewage Systems. ASAE. December 10-11, 1984, pp. 305-313.

Ehrenfeld, Joan G. 1987. The role of woody vegetation in preventing ground water pollution by nitrogen from septic tank leachate. Wat. Res. 21:5:605-614.

Gilliom, Robert J. and Clayton R. Patmont. 1983. Lake phosphorous loading from septic systems by seasonally perched ground water. Journal of the Water Pollution Control Federation. 55:10:1247-1305.

Gold, Arthur J., William R. Reagan, W. Michael Sullivan, and Jewell L. Lemunyon. 1990. Nitrate-nitrogen to ground water from rural and suburban land uses. Journal of Soil and Water Conservation: 45:2:305-310.

Hardisty, D.M., 1974, of nitrogen and phosphorus considerations in on-site wastewater disposal. From Laak, R 1982. On-site Soil Systems, Nitrogen Removal. In A.S. Eikun and R.W. Seabloom (eds) Alternative Wastewater Treatment 127-143. Copywright 1982 by D. Reidel Publishing Company.

Harkin, John M., Charles F. Fitzgerald, Colin P. Duffy, and David G. Kroll. 1979. Evaluation of mound systems for purification of septic tank effluent. Technical report WIS WRC 70-05. University of Wisconsin, Madison, Wisconsin. 170 pp.

Harmsen, Eric W. 1989. Siting and depth recommendations for water supply wells in relation to on-site domestic waste disposal systems. Thesis for doctor of philosophy. University of Wisconsin-Madison. 329 pp.

Hickock, E.A. and Associates. 1978. Effects of wastewater stabilization pond seepage on ground water quality. Prepared for the Minnesota Pollution Control Agency. 183 pp.

Jones, R. Anne, and G. Fred Lee. 1979. Septic tank wastewater disposal systems as phosphorus sources for surface waters. Journal of the Water Pollution Control Federation, 51:11:2764-2775.

Kanivetsky, Roman. 1979. Regional approach to estimating the ground water resources of Minnesota. Minnesota Geological Survey Report of Investigations 22. 134 pp.

Laak, Rein. 1982. On-site soil systems, nitrogen removal. In A.S. Eikun and R.W. Seabloom (eds) Alternative Wastewater Treatment, 129-143. Copywright 1982 by D. Reidel Publishing Company.

Laak, Rein. 1985. A passive, on-site advanced treatment system removes phosphorus paper presented at a Conference on Management Strategies for Phosphorus in the Environment, Lisbon, July 1-4, 1985.

Laak, Rein. 1986. Wastewater Engineering Design for Unsewered Areas. Technomic Publishing Company, Inc., Lancaster, Pennsylvania. 171 pp.

Lamb, B., A.J. Gold, G. Loomis, C. McKiel. 1987. Evaluation of nitrogen removal systems for on-site sewage disposal. Fifth national symposium on individual and small community sewage systems, American Society of Agricultural Engineers.

Ligman, K., N. Hutzler, and W.C. Boyle. 1974. Household wastewater characterization. ASCE J. of the Environmental Engineering Division. Paper number 10372. 100:EE1:201-213.

Lindholm, G.F. 1980. Ground water appraisal of sand plains in Benton, Sherburne, Stearns and Wright Counties, central Minnesota. U.S. Geological Survey open file report 80-1285.

Magner, J.A., D.B. Wall, W.J. Zaadnoordijk, and R.A. Piper. 1987. Predicting the ground water flow field under large soil absorption systems using a simple two dimensional, analytical model. ASAE Publication 10-87. pp. 309-318.

McCassland, M., N.M.Trautman, K.S. Porter, and R.J. Wagenet, 1985. Nitrate: health effects in drinking water. Cornell cooperative extension. Fact sheet page 400.02.

Metropolitan Council. 1974. The relationship of septic tank absorption systems to soils. 45 pp.

Miller, J.C. 1972. Nitrate contamination of the water-table aquifer in Delaware, Delaware Geological Survey report of investigations. No. 20.

Miller, R.T. 1982. Appraisal of the Pelican River sand-plain aquifer, western Minnesota. U.S. Geological Survey open-file report 82-347. 44 pp.

Minnesota Pollution Control Agency. 1984. High Rate Absorption (HRSA) Task Force Final Report. November 1, 1984.

Minnesota Pollution Control Agency. 1989. Nonpoint source ground water strategy. Issues and recommendations. 33 pp.

Myette, C.F. 1986. Hydrogeology of the sand plain aquifers in Carlton, Kanabec and Pine Counties, east-central Minnesota. WRI 85-4334.

Nelson, M.E., S.W. Horsley, T.C. Cambarerum, M.D. Giggey, and J.R. Pinnette. 1988. Predicting nitrogen concentrations in ground water - an analytical model. Proceedings of the FOCUS Conference on Eastern Regional Ground Water Issues. September 27-29, 1988. Stamford, Connecticut. pp. 179-202.

Olmsted County. 1990. Technical review committee report on individual sewage treatment systems. December, 1990. 85 pp.

Otis, Richard J. and William C. Boyle. 1976. Performance of single household treatment units. ASCE Journal of the Environmental Engineering Division. 12:1:175-189.

Perkins, Richard J. 1984. Septic tanks, lot size and pollution of water table aquifers. J. of Environmental Health. 46:6:298-304.

Piluk, R.J. and O.J. Hao. 1989. Evaluation of on-site waste disposal system for nitrogen redcution. J. Environmental Engineering. 115:4:725-739.

Polkowski, L.B., W.C. Boyle. 1970. Ground water quality adjacent to septic tank - soil absorption system. University of Wisconsin Engineering Experiment Station. 75 pp.

Powelson, David K., James R. Simpson, and Charles P. Berba. 1990. Virus transport and survival in saturated and unsaturated flow through soil columns. J. Environ. Qual. 19:396-401.

Quan, E.L., H.R. Sweet, and Joseph R. Illian. 1974. Subsurface sewage disposal and contamination of ground water in east Portland, Oregon. Ground Water. 12:6:356-367.

Reddy, M.R. and S.J. Dunn. 1984. Effect of domestic effluents on ground water quality: A case study. The science of the total environment, 40 (1984) pgs. 115-124. Elsevier Science Publishers. B.V. Amsterdam, Netherlands.

Reneau, R.B., Jr., 1979. Changes in concentrations of selected chemical pollutants in wet, tile-drained soil systems as influenced by disposal of septic tank effluents. J. Environ. Qual., 8:2:189-196.

Ritter, William F. and Anastasia E.M. Chirnside. 1984. Impact of land use on ground water quality in southern Delaware. Ground Water. 22:1:38-47.

Rodgers, G.A. 1980. Evaluation of potential substrates to monitor respiratory nitrate reductase activity in soils. Soil Science. 31:387-395.

Robertson, N.D., E.A. Sudicky, J.A. Cherry, R.A. Kapaport and R.J. Shimp. 1989. Impact of a domestic septic system on an unconfined sand aquifer. Proceedings initial symposium on contaminant transport in ground water, Stuttgart, Germany. April 1989.

Robertson, W.D., J.A. Cherry, and E.A. Sudicky. 1991. Ground-water contamination from two small septic systems on sand aquifers. Ground Water. 29:1:82-92.

Sandy, A.T., W.A. Sack, S.P. Dix. 1987. Enhanced nitrogen removal using a modified recirculating sand filter. On-site Wastewater Treatment Proceedings of the Fifth National Symposium on Individual and Small Community Sewage Systems. Dec. 14-15, 1987. Chicago. ASAE. pp. 161-170.

Siegrist, R.L., D.L. Anderson, J.C. Converse 1984. Commercial wastewater on-site treatment and disposal. Proceedings of the Fourth National Symposium on Individual and Small Community Sewage Systems. New Orleans, LA. ASAE publication 07-85.

Siegrist, R., M. Witt, and W.C. Boyle. 1976. Characteristics of rural household wastewater. Journal of the Environmental Engineering Division. 102:3:533-548.

Sikora, L.J. and R.B. Corey. 1975. Fate of nitrogen and phosphorus in soils under septic tank waste disposal. American Society of Agricultural Engineers Winter Meeting, Chicago, Illinois. 19 pp.

Smith, K.A. 1980. A model of the extent of anaerobic zones in aggregated soils, and its potential application to estimates of denitrification. Journal of Soil Science. 31:263-277.

Soukup, W.G., D.C. Gillies, and C.F. Myette. 1984. Appraisal of the surficial aquifers in the Pomme De Terre and Chippewa River Valleys, Western Minnesota. U.S. Geological Survey Water Resources Investigations Report 84-4086. 63 pp.

Starr, J.L. and B.L. Sawhney. 1980. Movement of nitrogen and carbon from a septic system drainfield. Water, Air, and Soil Pollution. 13:113-123.

Tevis, Cheryl and Betsy Freese. 1988. Water treatment systems. Successful Farming, December 1988.

Tinker, John R., Jr. 1990. Volatile organic chemicals beneath unsewered subdivision. J. Env. Health, 53:2:26-28.

Tinker, John R., Jr. 1991. An analysis of nitrate-nitrogen in ground water beneath unsewered subdivisions. Ground water monitoring review. 11:1:141-150.

Walker, W.G., J. Bouman, D.R. Kenney, and P.G. Olcott. 1973a. Nitrogen transformations during subsurface disposal of septic tank effluents in sands: I. Soil transformations. J. Environ. Quality. 2:4:475-480.

Walker, W.G., J. Bouman, D.R. Kenney, and P.G. Olcott. 1973b. Nitrogen transformations during subsurface disposal of septic tank effluent in sands: II. Ground water quality. J. Environ. Quality. 2:4:521-525.

Wehrmann, A. 1983. Potential nitrate contamination of ground water in the Roscoe area, Winnebago County, Illinois. Illinois State Water Survey, Division Contract, Report 325.

Witt, M., R. Siegrist, and W.C. Boyle. 1974 rural household wastewater characteristics. Proc. Nat. Home Sewage Disposal Symposium, American Society of Agricultural Engineers. Chicago Illinois. Dec. 9-10, 1974. pp. 79-88.

Wolf, R.J. 1981. Hydrogeology of the Buffalo Aquifer, Clay and Wilkin counties, west-central Minnesota. U.S. Geological Survey Water-Resources Investigations. 81-4. 83 pp.

U.S. EPA. 1977. The report to congress - waste disposal practices and their effects on ground water. U.S. EPA. Washington, D.C.

U.S. EPA. 1984. Septage Treatment and Disposal Handbook. 300 pp.

MUNICIPAL AND INDUSTRIAL WASTE

Within this chapter the amount of nitrogen (N), number of N releasing sites, and potential ground water N impacts from Municipal and Industrial Waste producing facilities is discussed. Since the treatment processes are similar for industrial and municipal wastes, these sources are discussed within the same chapter. The chapter is divided into four sections 1) land application of wastewater, 2) sludge application, 3) leaking ponds, and 4) discharge into streams.

LAND APPLICATION OF WASTEWATER

Overview

Land treatment is defined as the controlled application of wastewater onto the land surface to achieve a designed degree of treatment through natural physical, chemical and biological processes. The two most common types of land treatment systems in Minnesota are spray irrigation (slow-rate release) and rapid infiltration basins. Seepage basins and overland flow are rarely used for land application of wastewater in Minnesota. When properly sited, designed, constructed and operated, land application systems are a reliable, cost effective, environmentally acceptable method of treating wastes.

Spray irrigation is the application of wastewater to a vegetated land surface. After going through primary and secondary treatment, the applied wastewater is further treated as it flows through the plant-soil matrix. Nitrate removal occurs primarily by crop uptake, which varies with the type of crop grown and the crop yield. The crop should be harvested for effective N removal. Nitrate can also be lost through denitrification and ammonia volatilization. Denitrification losses are typically in the range of 15 to 25 percent of the applied N, but vary greatly with site conditions (U.S. EPA et al., 1981). Ammonia volatilization is another N loss mechanism associated with land application of wastewater. In designing a slow rate system, N losses by denitrification, volatilization and storage in the soil should not be expected to be over 25 percent.

In municipal systems, the most common factor limiting hydraulic loading rate is soil permeability and crop water requirements. Industrial application rates are more likely to be based on soil permeability and N content of the wastewater. Off-site surface runoff of the applied water is generally avoided in design. A properly managed spray irrigation site is the most effective wastewater disposal alternative for treatment of nutrients.

The rapid infiltration wastewater treatment method (RI) is a process where most of the applied wastewater percolates through the soil (in a basin), eventually reaching ground water. RI Systems are often installed when phosphorus in the effluent is too high for discharge to surface waters and when irrigation systems would be too costly. RI basins are usually located in areas where the ground water discharges to nearby surface water. The pre-treated wastewater is applied to moderately and highly permeable soils by spreading in basins or by sprinkling. The wastewater is further "treated" as it moves through the soil matrix. Vegetation is usually not planted, but may grow naturally as grasses and weeds. Therefore, there is often little consumptive use of N by plants.

The primary N removal mechanism in RI systems is nitrification-denitrification. First, the ammonium-N in wastewater must be converted to NO_3 (nitrification), and then nitrate may be converted to N gas (denitrification). Nitrification of the applied wastewater is essentially complete when appropriate hydraulic loading occurs. Warm soils (e.g., > 50°F) with pH greater than 5.5 are most conducive for N removal. Also, alternating aerobic and anaerobic conditions are necessary for significant N removal.[1] The bacteria responsible for denitrification require organic carbon (2 mg of TOC are needed to denitrify 1 mg of N), which can be found in the wastewater. For greater N reduction, primary pre-application treatment of wastewater is preferred in order to leave an adequate carbon supply in the wastewater for the denitrifying bacteria (U.S. EPA et al., 1981). EPA et al. (1981) reviewed several studies across the United States, concluding that N removal in a Rapid Infiltration System is commonly around 50 percent.

Overland flow, which is another wastewater treatment method, is only practiced at a couple sites in Minnesota and will not be discussed in this report.

Under State Policy (Minn. Stat. § 115.03, subd. 1, para e; 116.07, subd. 4) all facilities discharging wastewater are required to apply for a written permit from the MPCA. The MPCA may choose not to require a permit for low volume dischargers. No rules exist for land application of wastewater. The specifications within each permit may vary, but are generally the same for similar waste types being disposed in a similar manner. For spray irrigation of wastewater, the following conditions are written into most permits currently issued:

1) prolonged saturated soil conditions must not develop as a result of spraying;

2) wastewater application shall be discontinued after the first killing frost of each season;

3) no surface runoff may result from the land application site; and

4) a cover crop shall be maintained on the sprayfield during the entire application season;

5) cover crop must be harvested;

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[1]Alternating loading and resting of RI basins and ensuring that the ground water mound is maintained at a minimum of three feet below the bottom of the basin will help to accommodate the nitrification/denitrification processes.

The cover crop chosen at a spray irrigation site is important for N removal considerations. Further evaluation is needed to determine the most appropriate cover crops for removing nutrients from municipal and industrial wastewater.

Ground water monitoring is not required for all permitted industrial/municipal sites. Monitoring wells are sometimes required when the area is geologically sensitive, improper operation is suspected, or high contaminant concentrations are found in the waste. Lack of comprehensive ground water monitoring information is a barrier to an accurate assessment of the impact of wastewater systems on the ground water quality.

Nitrate application rate limitations have only recently (since 1989) been added into municipal permitting conditions. Most existing permits for industrial wastewater application do not have a N rate provision yet included. However, since a permit is issued for five year periods, there could be an opportunity to revise the permits in the near future to include a N provision. The ground water monitoring requirements are sometimes waved after two years of operation if water quality shows little change from the initial levels. It is possible that the complete NO_3 impacts on ground water would not be evident during that two-year period, or a change in operations could increase concentrations.

Another problem with the monitoring requirements is that there is currently insufficient MPCA staff to routinely review the monitoring results and take early appropriate follow-up action or to enforce the permit requirements for monitoring. No wastewater application rules exist to restrict N application rates if ground water NO_3 concentrations exceed 10 ppm below land application sites. The state's nondegradation policy as described in Chapter 7060 is the governing rule that would apply to N in ground water from land application of wastewater. Application of this rule for wastewater discharge is used inconsistently. Chapter 7060 is currently under revision.

Municipal Wastewater Land Application in Minnesota

The table below (J-1) lists the number of permitted municipalities and private domestic complexes (e.g. mobile home parks, condominiums) that land apply wastewater.

Table J-1. Number of municipalities and private domestic complexes permitted for land application of wastewater.

	Municipalities	Private Domestic Complexes
Spray Irrigation	37	7
Rapid Infiltration	9	1

In most cases, the amount of N going onto a given field from spray irrigation or RI sites is unknown. Effluent monitoring at five sites showed ammonia and ammonium and organic-N concentrations between 0.1 and 6.6 mg/l (24 analyses). Assuming per capita wastewater generated is 75 gallons containing about 7 mg/l total N, a city of 1000 would release around 1600 pounds of N per year in spray irrigation water. Since the N applied through wastewater is relatively low compared to the crop's N needs and wet conditions during the summer would prevent irrigation, field managers usually do not account for or rely on N from municipal spray irrigation.

Monitoring wells are required at municipal spray irrigation sites only when there is a high proposed application rate or the site may impact wells used for drinking water. Ground water N data reported to MPCA from below seven permitted sites show most wells with concentrations ranging between 2 and 25 mg/l. Historic NO_3 concentration data are not available at these sites to allow determination of the impact of spray irrigation alone on ground water NO_3. Elevated ammonium (approaching 10 mg/l at times) in some of these monitoring wells is likely originating from the municipal wastewater.

There are ten permitted municipal and domestic wastewater RI's in Minnesota. While the ten municipal RI's certainly do not pose a significant statewide or regional threat to ground water N, it is likely that some N is moving to ground water from these basins. If proper hydraulic loading procedures are followed, the N concentrations reaching ground water should pose very little health risk. However, inspections of the RI sites suggests that proper hydraulic loading (resting/loading cycles) procedures are not adhered to.[1] Ground water monitoring around five RI's shows elevated NO_3-N at three sites and elevated ammonium-N concentrations at one of these sites. However, high NO_3 at most of these sites is believed to be in part due to previous or surrounding land uses. Nitrate concentrations at some of the sites have been decreasing with time.

Industrial Wastewater Land Application in Minnesota

Most industrial land application treatment sites in the state consist of spray irrigation. Table J-2 shows the number of industries permitted for land application of wastewater containing N.

Table J-2 Number of industries permitted for spray irrigation.

	Canneries	Dairies	Others (silage, poultry, other food processing)
Spray Irrigation	16	5	9

The N concentrations in industrial wastewater are often much greater than in municipal wastewater. By examining MPCA files from permitted discharging facilities, the average N in effluent from seven canneries was found to be 150 mg/l, ranging from 30 to 300 mg/l. Average N from three dairy wastewater sites was 24 mg/l. Silage wastewater can have extremely high N, measured at over 600 mg/l at one site. The volume of wastewater applied varies greatly between facilities.

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[1]Personal communication with Neal Wilson, MPCA.

Four of the industrial wastewater application sites have monitoring wells that are regularly sampled. The NO_3-N concentrations in these wells have been reported to be below 10 mg/l. Many other facilities are monitored with pressure vacuum lysimeters, which are used to sample soil water before it reaches the saturated zone. Results from lysimeters usually indicate elevated NO_3 throughout much of the growing season and low levels upon maturation of the cover crop. Lysimeter monitoring may not be as effective in understanding the ground water impacts from industrial wastewater application as monitoring wells. Additionally, ground water standards do not apply to samples obtained from lysimeters. As permits are re-issued the MPCA industrial permit section is attempting to introduce intervention limits and begin requiring ground water monitoring.

LAND APPLICATION OF SLUDGE

Overview of Municipal Sludge

Municipal sewage sludge, the solids removed from sewage during wastewater treatment, is disposed of in Minnesota primarily through incineration or spreading on agricultural lands. Most of the sludge generated in the Twin Cities metropolitan area and Duluth is incinerated, while most of the sludge produced in the rest of the state is applied to agricultural lands. A very small amount goes into landfills. Potential pollutants of concern from sludge include organic solids, pathogens, N, phosphorus, heavy metals, and persistent organic chemicals from household products.

Nitrate is the macronutrient which maximum sludge application rates are usually based on. Current philosophy in Minnesota regarding sludge application is to design an application system based on sound agronomic principles, so that sludge utilization poses no greater threat to ground water resources than current agricultural practices. The concentrations of organic N, NH_4, and NO_3 in sludge are affected by the type of sludge treatment and handling processes. Table J-3 shows the mean and median total N for different sludge types in eight states, primarily in the Midwest.

Table J-3. Total Nitrate as a percent composition on a dried solids basis for various treatment operations (Sommers, 1977)

Sludge Type	Number	Range	Median	Mean
Anaerobic	85	0.5 - 17.6	4.2	5.0
Aerobic	38	0.5 - 7.6	4.8	4.9
Other	68	<0.1 - 10.0	1.8	1.9
All	191	<0.1 - 17.6	3.3	3.9

The mean total N in anaerobic and aerobic digested sludge is about five percent of the sludge on a dry weight basis. Eighty to ninety percent of this N is organic N, which will slowly decompose after application to soils resulting in release of NH_4 (Dunn et al. 1985). The amount and rate of conversion from Organic N to Ammonium is affected by the extent of sludge processing within the treatment plant, temperature, water content, soil pH, and the carbon to N ratio in the soil.

Inorganic N in sludge will usually be over 90 percent ammonium, unless aerobic conditions prevailed during sludge treatment. Dewatering of liquid sludges will subsequently lower the ammonium content, resulting in a sludge with less than ten percent of the total N being present as NH_4 (EPA, 1983). When liquid sludges are applied to the soil surface rather than being injected, more than half of the NH_4 can be lost to the atmosphere.

Plant available N in sludge is the total of 1) all NO_3 in the sludge, 2) ammonium in the sludge minus volatilization losses, 3) the fraction of the organic N present in the sludge that is mineralized during the first year, and 4) mineralized N from previous years of application. Twenty to fifty pounds of N per dry ton of sludge will be available for plants each year. The University of Minnesota has found that crop yields on sludge applied land in Rosemount have been slightly higher than commercially fertilized control areas within the same watershed (Cheng, et. al., 1989). Results from the Rosemount Study also indicate that grass was more efficient than corn in nutrient removal.

Municipal Sludge Application in Minnesota

One hundred fifty-two communities regularly apply sewage sludge in Minnesota. Nearly 33,000 dry tons of sludge were applied to 9035 acres of cropland in Minnesota during 1989. This figure was significantly higher than the 1988 application of 19,453 dry tons. Much of the difference between 1988 and 1989 can be explained by the actions of one city. This city, which typically applies 245 tons/year, emptied its lagoons for repair work in 1989, releasing 7600 tons of sludge to the land.

Assuming 25,000 tons of sludge (dry weight basis) to be a typical amount applied in Minnesota per year, five percent of which is N, then the total N annually applied is 1,250 tons (2,500,000 lbs.). About 250,000 to 625,000 lbs. of this N will be converted to plant available forms of N (ammonium or NO_3) in a given year. Since sludge N is applied to 9,035 acres (1989 figure), about 28 to 70 pounds of plant available N is released per acre throughout Minnesota where municipal sludge is applied.

Current Policy Regarding Municipal Sludge

The Minnesota Legislature recognized the potential impact of improper landspreading of sewage sludge. In 1980, the Waste Management Act was passed, requiring the MPCA to develop standards for land spreading of sewage sludge. Sewage sludge management rules, Chapter 7040, became effective in May 1982 (Minn. Stat. § 116.07, subd. 4). The rules state under 7040.1802 that "Sewage sludge application rates, combined with other known N sources, shall supply no more N than the amount required by the vegetation to be grown at the site." The rules are essentially what could be described as mandatory best management practices. The determination of sewage sludge application rates are based on crop N requirements and are described in 7040.4600. Sewage sludge application rates as outlined in Chapter 7040 are to be based on soil texture, crop N requirements and yield goals, sewage sludge N availability, carry-over N supplied by past sewage applications, and available N added by manures or fertilizers. More specific guidelines are provided in Chapter 7040.

The rules stipulate that all wastewater treatment facilities which landspread sludge must have permitted or approved sites or facilities. Facilities (dedicated land for spreading of sludge) may be required to conduct monitoring programs to evaluate any impacts that may be occurring.

It is recommended that the MPCA periodically review Chapter 7040 to make sure that the procedure to determine sewage sludge application rates based on crop N requirements are consistent with the most recent University of Minnesota research results.

Prevention of sludge N moving into ground water is further addressed in Chapter 7040 with requirements including choosing appropriate sludge application sites based on the soils and geology, analysis of sewage sludge for N on a regular basis, minimum separation distances between application sites and wells and surface water bodies, and reporting requirements to the MPCA.

Certification is required for all waste disposal facility operators and inspectors under Minn. Stat. § 116.41, subd. 2. The rules for certification (chapter 7048) state that all operators must complete training offered by or approved by the MPCA, meet educational requirements, and have work experience related to waste disposal. Sludge Application is a part of the MPCA training that must be completed.

Monitoring Results At Sewage Sludge Application Facilities

Rosemount Study

A long-term study by the University of Minnesota, USDA Agricultural Research Service, and the Metropolitan Waste Control Commission at Rosemount has provided a vast amount of information regarding environmental and agricultural analysis of sludge application to land. It was shown in this study that excessive fertilizer and liquid sludge application rates resulted in high NO_3 concentrations in the underlying shallow ground water (Cheng, 1989). At the sludge application plots, NO_3 concentrations have been determined in three wells under reed canary grass. Average annual NO_3-N concentrations increased from less than 10 mg/l in 1974 to over 50 mg/l in the early to mid 1980's (nine to eleven years after initial sludge application). Following similar application rates to corn, NO_3-N in three wells below the corn plots were over 80 mg/l. Nitrate concentrations in the ground water have decreased following reduced N application rates were reduced. During 1989, NO_3-N concentrations averaged 40 mg/l in the three wells under corn and 21 mg/l in the three wells under the grassed plot. Application of high rates of commercial N fertilizer to a control area from 1985-1988 resulted in shallow ground water NO_3 concentrations similar to those in the sludge treated corn area.

The Rosemount Study concludes that sludge is a very good soil amendment, supplying the nutrient needs and maintaining the quality of field crops. However, as with any excessive use of fertilizer, excess sludge application can result in high NO_3 concentrations in ground water. Proper management of the cropping systems and total N can reduce the potential of significant NO_3 movement to ground water below sludge application sites.

Monitoring Records at MPCA

Ground water monitoring is required by MPCA below seven of the 152 permitted sludge application sites in Minnesota. Average NO_3-N concentrations exceed 10 mg/l at four of the seven sites and at two of these sites there is a large increase in NO_3 between upgradient and downgradient wells.

Industrial Sludge Application

Solid wastes from some food processing operations is returned to fields. The MPCA is not aware of the amount of industrial sludge that is returned to fields in Minnesota. This waste is likely insignificant in the statewide picture, but can pose a potential threat to ground water N in areas surrounding large processing facilities, especially if the N content is not considered when applying fertilizer. Generally, solid wastes from food processing contains about one percent N (U.S. EPA, 1991).

LEAKING WASTEWATER TREATMENT PONDS

Newly constructed wastewater treatment ponds and existing ponds undergoing upgrades in Minnesota are required to assess the seepage rate of the pond seal by the Minnesota Water Balance Test. This test was evaluated by a Water Balance Task Force in 1987 and is described in the resulting report entitled "Report on Evaluation of Minnesota Water Balance Test" by MPCA and Consulting Engineers Council of Minnesota. This test was reported to be adequate to determine if newly constructed ponds are properly sealed and achieve a seepage loss less than 500 gal/acre/day (0.018 inches/day) with a 95 percent confidence interval of plus or minus 1000 gal/acre/day. A leaking rate of 500 gal/acre/day (the limit for newer systems) would be equivalent to the volume of effluent from septic systems on one-third acre lots. However, the N concentration in septic systems is generally higher than wastewater treatment ponds. When ponds are thought to leak at an excessive rate, a ground water monitoring study is required. Currently there are about 15 cities that are monitoring ground water quality below excessively leaking ponds. There is likely other ponds in the state that leak excessive amounts that have no monitoring wells. One of the reasons for the number of potentially leaking ponds is that previous to 1975 pond seal requirements were considerably less stringent than today, with allowable seepage rates of 1/8 inch per day (3394 gal/acre/day). Professional inspectors are now required to conduct soil tests daily as new pond seals are constructed. When an inadequate seal is made, the problem is required to be fixed before wastewater is placed in the basin.

Hickock, E.A. and Associates (1978) conducted a leakage study from five municipal wastewater treatment stabilization ponds and found two ponds leaking at rates of 44,300 and 22,000 gal/acre/day. Ground water near both of these sites had elevated N. Two other cities, one with ponds leaking 200 to 7400 gal/acre/day and the other leaking at a rate of about 4100 gal/acre/day had no appreciable increases in ground water N concentrations after over 14 years of use. The fifth city leaked at a rate of 1400 to 3000 gal/acre/day, with some fecal coliform, chloride and hardness problems associated in the ground water, but with no mention of elevated N.

Other suspected leaking ponds are currently under investigation by the MPCA. One recent investigation showed excessive leakage in ponds treating waste from a south central Minnesota town. Ammonium concentrations in three nearby monitoring wells were between 10 and 20 mg/l during two sampling events in 1990. Nitrate-N concentrations were less than 0.2 mg/l in all wells. Due to the severe leakage, anaerobic conditions probably exist below the pond, thereby preventing the conversion of ammonium into NO_3.

WASTEWATER DISCHARGE INTO SURFACE WATERS

Most conventional treatment plants are not designed to remove total N from the effluent and usually remove no more than 30 to 40 percent of the N (EPA, 1991).

A majority of wastewater treatment systems end up discharging the treated effluent into streams or ditches. Because this water could end up back in ground water via losing streams or by pumping of wells adjacent to streams, wastewater discharge into surface water must be considered a potential source of ground water N.

While the contribution of municipal and industrial waste is a minor N contribution to the land compared with other sources, these sources contribute a higher fraction of N to surface waters.

During 1991, 575 municipal and 317 industrial facilities were permitted for discharge of treated wastewater into surface water in Minnesota. Nitrogen concentrations in discharged effluent often are about 25 mg/l, with the dominant forms of N being dependent upon the type of treatment process used (EPA, 1991). Where the ratio of stream discharge to effluent discharge is less than 10 to 1, ammonia concentrations in the effluent are required to be reported to the MPCA. From 1990 sampling results, 37 municipalities and 13 industrial facilities reported effluent ammonia plus ammonium (NH_3+MH_4-N) concentrations (see Table J-4).

Table J-4. (NH_3+MH_4-N) concentrations in wastewater effluent before surface water discharge from all sites where concentrations were reported to MPCA during 1990.

(NH_3+MH_4-N) in Effluent (number of sites)

	0-1 mg/l	1-5 mg/l	5-10 mg/l	10-20 mg/l	>20 mg/l	Total
Municipal	17	6	5	6	3	37
Industrial	5	2	2	0	4	13

It is evident from Table J-4 that there are many sites that discharge treated effluent which have fairly high (NH_3+MH_4-N) concentrations. The mean and median (NH_3+MH_4-N) concentrations from municipal wastewater discharge were 6.2 and 2.2 mg/l, respectively. Unionized ammonia, which can be quite toxic to aquatic life, is usually a very small fraction of the ammonia plus ammonium concentration and varies with water temperature and pH.

When (NH_3+MH_4-N) concentrations in the receiving streams exceed water quality standards, the municipalities are required to reduce the levels. Most of these plants reduce (NH_3+MH_4-N) levels by converting (NH_3+MH_4-N) to NO_3. Currently, no municipalities are required to reduce total N levels as a result of their permit.

MPCA has conducted effluent monitoring surveys since 1976 at 52 municipal wastewater treatment facilities with mechanical treatment. Most of the sites have been sampled for all major N compounds two to eight different times. Total N concentrations in the effluent were between 10 and 30 mg/l at most sites, averaging 17.4 mg/l. From 130 analyses at the 52 sites, the maximum and minimum total N concentration were 61.9 and 1.3 mg/l. The forms of N found at the different sites varied greatly. Organic N concentrations generally ranged between 1 and 5 mg/l and averaged 3.2 mg/l. Ammonium concentrations generally ranged between 2 and 20 mg/l, averaging 7.2 mg/l. However, some sites had at least one analysis showing ammonium-N concentrations in the 20 to 30 mg/l range. Nitrite-N concentrations were less than 1 mg/l, with only a few exceptions. Nitrate-N concentrations generally ranged between 0.1 and 25 mg/l, averaging 7.8 mg/l.

SUMMARY

Land application of wastewater is permitted for 8 private domestic complexes, 46 municipalities and 30 industrial facilities. Limited data shows that land applied municipal wastewater generally has relatively low N concentrations. Industrial wastewater can have very high N concentrations, often exceeding 100 mg/l. Little is known about industrial effluent N concentrations or the ground water N levels below fields receiving industrial wastewater.

One hundred fifty-two communities regularly apply sewage sludge in Minnesota on a total of about 9035 acres of cropland. About 28 to 70 lbs. of plant available N is released from sludge per acre throughout Minnesota where municipal sludge is applied. Municipal sewage sludge application is regulated by the MPCA and application rates are usually based on crop N needs. University of Minnesota research has shown that excessive sludge application can result in elevated ground water NO_3 concentrations. With proper management of cropping systems and total N, sewage sludge can be a good soil amendment with minimal adverse ground water N impacts.

Excessively leaking wastewater treatment ponds have been shown to cause elevated N levels in ground water. Criteria for pond design has become more stringent in recent years, thereby decreasing the likelihood of excessive leakage from newly constructed ponds.

During 1991, 575 municipal and 317 industrial facilities were permitted for discharge of treated wastewater into surface water. Total N concentrations in this wastewater are often over 25 mg/l. The primary surface water concern with N is unionized ammonia which is toxic to fishes. During 1990, 37 municipalities and 13 industrial sites reported effluent NH_3+NH_4 concentrations. Nine municipalities and four industrial sites had NH_3+NH_4 over 10 mg/l. MPCA sampling of 52 municipal wastewater treatment facilities with mechanical treatment indicated a mean effluent total N concentration from all facilities of 17.4 mg/l. Ammonium, nitrite, and organic-N in discharged effluent has the potential to convert to NO_3 in the stream and subsequently be drawn into aquifers. While this potential exists, surface discharge of treated wastewater effluent is not believed to significantly affect ground water NO_3 concentrations in most areas of the state.

From a statewide perspective, municipal and industrial waste is not a large part of N input to ground water. However, improperly designed, constructed or managed treatment systems do represent potential localized ground water N threats and N from all sources should be properly accounted for when managing cropping systems.

Recommendations

- Nitrogen from municipal spray irrigation systems should be accounted for as part of the crops N needs.

- Monitoring well installation and N monitoring should be required for wastewater application sites that have high total N concentrations in wastewater (e.g. exceeding 30 mg/l) or are in close proximity to drinking water wells.

- It is recommended that the MPCA and University of Minnesota periodically review Chapter 7040 to make sure that the procedure to determine sewage sludge application rates based on crop N requirements are consistent with the most recent research results. Changes to 7040 should be made if necessary.

- It is recommended that better information be generated on the N content of treated wastewater and its impacts on surface and ground water N.

- An automated ground water data base should be used to routinely screen for problematic sites.

- Long-term funding should be allocated to provide adequate training of wastewater treatment system operators, facilitate construction upgrades and writing and enforcing permits.

REFERENCES

Cheng, H.H. 1989. Utilization of sewage wastes on land. Department of Soil Science - University of Minnesota. Soil series. 131.

Dunn, Art, Clarence Manke and Steve Stark. 1985. Land application of sludge. Minnesota Pollution Control Agency. 89 pp.

Hickock and Associates, 1978. Effects of wastewater stabilization pond seepage on ground water quality. Report completed for MPCA, January 1978. 183 pp.

Sommers, L.E. 1977. Chemical composition of sewage sludges and analysis of their potential as fertilizers. J. Environ. Qual., 6:225-239.

U.S. Environmental Protection Agency. 1983. Process design manual for land application of municipal sludge. Cincinnati, OH.

U.S. Environmental Protection Agency. 1991. Nitrogen action plan. Draft. March 1991. 160 pp.

U.S. Environmental Protection Agency, U.S. Army Corps of Engineers, U.S. Department of Interior, and U.S. Department of Agriculture. 1981. Process design manual for land treatment of municipal wastewater. October 1981. Published by U.S. EPA.

FATE OF NITROGEN FERTILIZERS APPLIED TO TURFGRASS

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INTRODUCTION

The goal of a sound fertilization and pesticide management program is to maintain a healthy, vigorous turf which is aesthetically pleasing to the eye and capable of withstanding high traffic. Proper N nutrition is essential. Nitrogen (N) deficiency causes slow growth, yellowing, spindly stems, and thin stands. Excess N results in excessive shoot and leaf growth, reduced root growth, low carbohydrate reserves, increased disease susceptibility and poor tolerance to stress (Taylor et al., 1989). Environmental considerations must also be included as an important component of the goal. Fertilizer applications has been proposed to be a major source of NO_3 contamination in urban areas where turf is a major land use (Flipse, 1984). With the exception of water, N is the most common factor limiting turf growth in Minnesota and annual application rates between 0 to 160 lb/A (0 to 3.6 lb/1000 ft^2) would not be considered unusual.

A sound understanding of the nature and composition of turfgrass must be established to understand the behavior of N and other fertilizers within its environment. Rates, sources, timing of applications, physical and chemical soil properties, irrigation management, and placement are all important considerations as well as agronomic considerations such as variety selection, plant density, rooting depth and density, and thatch development have a significant effect of N fate.

General trends seem to indicate that homeowners are more conscience about the appearance of their lawns than in the past. This is inferred from the tremendous growth in the turfgrass industry (Spectrum Research, 1990). While the public desire for well maintained turf is increasing, there is a parallel concern about the environmental ramifications from increased usage of fertilizers and pesticides on lawns. Public concerns have been elevated by the presence of the lawn care industry. This industry is highly visible. Possibly because of the frequency observed at their clientele, the lawn care industry has been subject to a great deal of negative public perception (Skogley, 1988). The public commonly associates high maintenance and management directly with water quality degradation. These concerns have been transferred to all turfgrass production, including homeowner applied materials.

MINNESOTA TURFGRASS ACREAGE AND N INPUTS

Turfgrass acreage estimates in the U.S. range from 20-25 (Spectrum Research, 1990) to 32 (EPA, 1991) million acres in the U.S. National estimates on non-farm N use are about 4% of the total N fertilizer use (EPA, 1991). Non-farm use is defined as application to golf courses, home lawns, and commercially owned turf.

Rosen et al. (1991) estimated the number of Minnesota residences at approximately 1.3 million. Based on this estimate and the assumption that the average size lawn is 0.2 acres, these authors estimated turf coverage at roughly 260,000 acres. No information relating N use and management on Minnesota turf was available for this review.

RECOMMENDED N APPLICATION RATES

Often the amount of N released from the organic matter within the soil and thatch is not sufficient to maintain vigorous growth throughout the growing season. Supplemental applications are generally required. Recommended yearly applications range from 1 to 4 lb/1000 ft^2 (44 to 175 lb/A) depending upon a high or low maintenance lawn and if grass clippings are removed (Taylor et al., 1989). See Table K-1 for recommended N application rates and timing strategies.

Strategies for fertilization of different portions of golf courses has been summarized by Spectrum Research (1990). Putting greens are intensively managed to maintain plant density, recuperative potential, color and adequate growth. Nitrogen fertilization rates typically range from 3 to 6 lb/1000 ft^2 (131 to 261 lb/A) and small applications at one to three week intervals are recommended. Similar rates are applied to the tee areas. Extensive damage is caused by divots and sufficient N must be applied to recuperate the turf. Tees that receive extensive activity may require up to twice as much N. The greens and the tees occupy approximately 4% of the total course area. The fairways and roughs are generally fertilized with rates similar to private lawns.

Table K-1. Annual N requirements and application timing for Minnesota lawns.[1]

Maintenance Practices	Nitrogen (N) lbs.N/1000 ft^2	Nitrogen (N) lbs.N/acre	Timing of applications[2]
<u>High maintenance lawn</u>			
(Irrigation, clippings removed)	4	174	Aug., Sept., Oct.-Nov., May-June
(Irrigation, clippings not removed)	3	131	Aug., Oct-Nov., May-June.
<u>Low maintenance lawn</u>			
(No irrigation, clippings removed)	2	87	Aug., Oct.-Nov.
(No irrigation, clippings not removed)	1	44	Sept.

1. Modified from Taylor et al., 1989.

2. Assuming 1 lb N/1000 ft^2 of quickly available nitrogen is applied at each application.

LEACHING OF FERTILIZER N APPLIED TO TURFGRASS

Petrovic (1990), in a thorough review of United States management, concluded that N movement has been generally only studied as a series of components rather than a holistic approach. Most studies are limited to a narrow time range (often months) under a limited set of physical conditions. Regardless of these limitations, there are a number of excellent studies from the cooler portion of the United States.

Nitrogen Rates

Similar to agronomic systems, rates are the primary factor contributing to water resource degradation. On fine sandy loams in Maryland, the effects of high N rates (200 lb/A) were compared to unfertilized plots (Gross et al., 1990). Application rates were split into five applications with the heaviest loading in the late fall. Average NO_3-N concentrations of the soil solution at 30" below the fertilized plots ranged from 0 to 3 mg/L over the 24 month study. Check plot NO_3-N levels averaged 0.3 mg/L. This study demonstrated that acceptable concentrations under intense N management can exist. Leaching losses, due to the dense root zone and efficient nutrient use, were kept to a minimum under properly managed and judiciously fertilized established turf. Under similar soil conditions, rates of 0, 90, and 220 lb/A/yr were found to have an insignificant effect on N percolation losses and concentrations when proper irrigation management was utilized (Morton et al., 1988). Using tagged ^{15}N material, Starr and DeRoo (1981) tracked NO_3 movement in a sandy loam under Kentucky bluegrass-red fescue turf. Nitrate-N concentrations at depths of 70-94" ranged from 0.3 to 10 mg/L under an annual rate of 160 lb/A. These authors concluded that at this annual rate, there was minimal ground water contamination.

Conversely, water quality was rapidly affected under golf greens constructed with sand media after a N application (Brown et al., 1977). This media would be considered coarser than most home lawns and proportionately represents only a small portion of an entire golf course. Drainage waters exceeded the drinking water standard for several days when using a 85 lb/A rate. Significant NO_3 leaching losses were measured in Michigan under worst case scenario conditions. Rieke and Ellis (1974) found that under high rates of N (260-350 lb/A) in single spring applications of an extremely mobile source ($NH4NO_3$) on irrigated coarse textured soils that substantial losses did occur. Much lower leaching losses under more realistic conditions indicated that N can be managed on sensitive soils with a careful fertilizer management plan. The study suggests the following practices: annual rates of less than 260 lb/A; lower and more frequent N applications; selection of correct N source; and the utilization of lower N requiring grass selection.

Exner et al. (1991) also determined that turfgrass fertilization contributed significant leaching losses in Nebraska sandy loam soils. Ammonium nitrate (34-0-0) at rates of 0, 89, 134, 179, and 214 lb/A were applied to bluegrass/fescue stands in late August. Excessive irrigations were applied during the next 34 days; a total of 25" was applied which exceeded evapotranspiration (sum of plant needs and evaporation losses) by a factor of three. Soil coring information showed that as much as 95% of the applied NO_3 leached below the turfgrass root zone. Excessive NO_3 leaching losses were also observed under the check plots; NO_3-N concentration of the irrigation water was 8 mg/L which was probably sufficient N to satisfy plant needs without any additional fertilization. Irrigation methods used in this experiment were believed to represent practices common in Sidney as well as much of the semiarid Midwest.

Gold et al. (1990) compared NO_3 losses during a 2-year study in Rhode Island under corn, forests, septic systems, unfertilized and fertilized lawns. Nitrate concentrations (flow weighted) were 13, 0.2, 68, 0.2 and 0.9 mg/L and annual leaching losses of 59, 1.3, 43, 1.2, and 5.0 lb/A for the respective sources. The fertilized lawn, despite receiving N rates similar to the corn, had ten times less NO_3 loss.

Nitrogen source

Rate of release or conversion of N to plant available forms (NH_4 and NO_3) is extremely variable in common turfgrass formulations. Taylor et al., (1989) categorized the soluble and slow-release forms of N (Table K-2). Within slow release source's, conversion times to inorganic N is dependent on each source's unique characteristics (Hummel and Waddington, 1984; Landschoot and Waddington, 1987), and will be subject to variation due to soil temperature and moisture conditions. From an agronomic perspective, agriculture is limited to three main N sources which dominate 90% of the national N sales (anhydrous ammonia, urea, and liquids). These sources would be considered moderate to highly soluble materials. Other sources with higher solubilities such as ammonium nitrate have been phased out due to low N composition or due to mobility problems. Options in source selection of turf fertilizers are numerous. A number of valid options for turfgrass would be economically unfeasible in production agriculture. Some slow release fertilizers are extremely expensive compared to many of the water soluble products. For many urban lawns and golf greens, this may not be a major stumbling block. However, to maintain fairways, parks and large lawns, the additional cost may be considered too expensive. The cost to fertilize an average size urban lawn can vary from $4 to $117 per year (Table K-2).

Many products available to the homeowner are mixtures of slow release and highly soluble forms of N. In Minnesota, the product must contain a minimum of 15% N in the slow release form to be registered by the Minnesota Department of Agriculture as a slow release product. Selections of an N source or combination of sources should be dependent on quickness and duration of desired response, rate and frequency of applications, and economic aspects (Landschoot and Waddington, 1987) along with environmental considerations.

Slow release N fertilizers (ureaformaldehyde and IBDU) and applications of sludge seldom resulted in soil NO_3-N concentrations higher than the check plots in Michigan coarse-textured soils (Rieke and Ellis, 1974). Percolation through Maryland sandy loams under rates of 200 lb/A were monitored comparing liquid urea and dry urea applications (Gross et al., 1990). Nitrate-N concentrations for the respective sources were 1.0 and 0.9 mg/L. Leachate and runoff losses were monitored from golf greens constructed from a variety of soil media (Brown et al., 1982) using a number of N sources. Percentages of applied inorganic N losses from NH_4NO_3, 12-12-12, Milorganite, ureaformaldehyde, and IBDU were 19, 7, 5, 1.4 and 0.9% respectively. The high loss from the NH_4NO_3 source was most likely due to the fact that 50% of the N is in the soluble NO_3 form. In a unique study under actual golf course conditions, Cohen (1990) studied NO_3 and pesticide concentrations under five Cape Cod courses. Although some elevated NO_3 concentrations were originally detected, acceptable concentrations did result when lower rates or slow release N, or both were utilized. Cohen concluded that reasonable changes in management practices minimized NO_3 problems.

Table K-2. Typical turfgrass N fertilizers and approximate costs.[3]

Nitrogen Sources	Approximate Cost per Lb. of N
Soluble materials	
Ammonium nitrate	$0.29
Ammonium sulfate	$0.42
Urea	$0.21
Urea solution	$0.39
Slow or Controlled Release	
IBDU (Isobutylidene diurea)	$1.72
Sulfur Coated Urea	$0.83
Ureafromaldehyde (such as Nitroform)	$0.95
Milorganite	$2.00
Sustane	$3.77
Formulene (Liquid 30-0-1)	$0.85

Timing of Nitrogen Applications

Timing of the N application can have a profound effect on the amount of N lost to leaching and can alter the physiology of the turf. Fall has been identified as a highly beneficial time for N fertilization of cool-season grasses (Street, 1988; Taylor et al., 1989; Wehner et al., 1988). During the late fall, top growth is minimal but soil temperatures are still warm enough for substantial N absorption which stimulates carbohydrate accumulation and spring root growth. Turf fertilized during this optimal time period will green-up in the spring without stimulating excessive shoot growth, maintain higher carbohydrate reserves during the spring and summer months, and have less problems with summer diseases. Heavy spring applications will result in a nice looking lawn for a short period of time but actually can delete the plants energy reserves (carbohydrates) which weakens the plant.

Fall has been also identified as a major ground water recharge period. Significant drainage can occur because of the low plant water use and reduced evaporation rates. Cool temperatures also reduce the activity of a number of important components of the N cycle. Timing of the application should consider the characteristics of the N source. Better turf color was observed with IBDU when applied in June/September as opposed to a November application (Wehner et al., 1988). With more soluble materials (urea and SCU), better color resulted the following spring due to a November application although small additional amounts of spring N were required in the urea treatment to produce optimal color.

3. Adopted from Taylor et al., 1989 and data compiled by D.H. Taylor. Prices were quoted on a ton basis and approximate and should be used for comparative purposes only.

The shift for more fall fertilization has the potential for more NO_3 losses if applications are made after the plant uptake period has concluded. Petrovic (1990) summarized some of his earlier findings and reported that 21 to 47% of November-applied urea (85 lb/A rate) was lost to leaching. When sulfur-coated urea was used, losses were reduced to 12%. Soil profile characteristics were important factors to the leaching component. Petrovic concluded "even though the late fall N fertilization principle has many good agronomic benefits, the environmental impact may overshadow the positive factors in ground water sensitive areas".

Split applications with the heaviest rates in the fall appear to be a reasonable compromise between optimum agronomic and minimized ground water effects. Concentrations under high N rates in a split application were kept to a minimal level (Gross et al., 1990). Leaching was significantly reduced when splitting high annual rates (350 lb/A) into three applications (Rieke and Ellis, 1974).

Effects of Physical Properties

Although soil texture is considered important in understanding leaching losses, studies under standardized treatments with soil texture as the main effect are rare. Concerns are commonly focused on golf greens due to the coarse media used in their construction. Greens commonly represent about 2% of the course area. Brown et al. (1982) studied the ground water impact from various soil textures utilized in golf green construction. The United States Golf Association specifications state that greens are to be constructed with 93% sand (maximum), 3% silt (minimum) and 5% clay. Infiltration must be at least 2"/hr. The following combinations were studied: "sand greens" consisting of 90% sand and 10% peat moss; "mixed greens" consisting of 80-85% sand, 5-10% clay and 10% moss; and "soil greens" constructed with 100% sandy loam soil. Leachate at 12" and runoff were monitored. When soluble forms of N (12-12-12 and 34-0-0, both forms of NH_4NO_3) were applied at rates ranging from 130 to 145 lb/A, concentrations from the sand greens commonly exceeded the 10 mg/L drinking water standard during the first 30 days since the application. Across all sources, total inorganic N lost from the "sand greens", "mix greens", and "soil greens" averaged 9, 8, and 3% respectively. Nitrogen source selection overshadowed soil texture effects.

Effects of Irrigation Management

Effects of N rate and irrigation scheduling were studied in Rhode Island (Morton et al., 1988). Irrigation scheduling schemes ranged from ideal (scheduling with tensiometers) to a worst case scenario (weekly applications plus rainfall) and N rates were 0, 85, and 220 lb/A. Even under the highest N rate and proper irrigation techniques, inorganic N concentrations were kept below 1 mg/L. Annual inorganic N losses were 1.7, 2.6, and 4.4 lb/A for the 0, 85, and 220 lb/A rates, respectively. Overwatering the plots resulted in losses of 2.5, 12.2 and 28 lb/A for the same respective rates. Mean annual losses ranged from 3 lb/A across N rates for the "scheduled" scheme to 14 lb/A for the "overwatered". The study suggests that proper irrigation management is essential for controlling leaching losses. The authors warn that despite the success of the irrigation scheduling, the average homeowner could not be expected to perform this level of scheduling due to the lack of knowledge regarding soil moisture stress. In sensitive areas, homeowners should be encouraged to limit the quantity and frequency of waterings.

Effects of irrigation scheduling and N sources were evaluated over a 2-yr period in a Florida sand (Snyder et al., 1984). Parameters such as plant color, tissue N, and concentration below the root zone were monitored under a rate of 45 lb/A/month. Nitrate-N concentrations, resulting from either fertigation or sulfur coated urea, were commonly less than 4 mg/L when irrigation events were triggered by soil moisture sensors. Nitrogen source selection was much more important when a wetter soil profile was maintained. These results strongly indicate that NO_3 in ground water can be minimized, even in coarse textured soils, by combining reasonable irrigation practices with a controlled release N source or through fertigation. This data also suggests that soil moisture sensors to trigger lawn irrigations would be an extremely beneficial management tool. Professional turf managers have both the economic and management incentives to control the frequency and quantity of irrigations. This level of control is not commonly observed on home lawns, commercial and industrial sites which are not under daily professional supervision. Automatic soil moisture sensors would greatly aid homeowners in making irrigation management decisions. This research also points out that with sound irrigation management that it was possible to use cheaper water soluble material such as urea with minor impact on leaching or runoff.

Effects of Turfgrass Selection

Nitrogen use is significantly affected by turfgrass selection particularly in the Kentucky Bluegrass varieties. Some varieties such as Merion need to be fertilized heavily to maintain its luxurious look (Anonymous, 1991). Plant breeders are now developing bluegrass cultivars that maintain good vigor and color with lower amounts of N. Some of the bluegrass varieties well-suited to low maintenance situations are: Aquilla, Monopoly, Newport, Park, Rugby, and South Dakota Certified (Taylor et al., 1989). Some of the newer tall fescues are being utilized more because of their better water use efficiency and lower nutrient needs. Mixtures of Kentucky bluegrass, fine fescue, and perennial rye grass can also produce a suitable low maintenance lawn.

Effects of Nitrogen Placement

The importance of the thatch layer must be considered when attempting to determine the fate of applied N. Thatch is defined as the intermingled layer of living and dead stems, leaves and roots of turfgrass. It develops between the green vegetation and the underlying soil surface. Where substantial thatch has formed, it can be the primary growth medium for turf. Characteristics of thatch are a highly porous structure, low cation exchange capacity (expressed on a volume basis) and low moisture holding capacity are somewhat physically analogous to sand. The presence of thatch and the accompanying microorganism populations can affect N transformations (Rieke and Ellis, 1974). Nelson et al. (1980) determined that slow release sources of N were more effective than urea when placed directly into the thatch. Less leaching, volatilization and more residual N was available when using isobutylidene urea (IBDU) compared to urea in either thatch or soil but the differences were much more profound in the thatch. Application of urea resulted in 2.5 times as much leaching in the thatch as compared to soil only. Where IBDU was utilized, leaching from the thatch was reduced from 81 to 5%.

In many homeowner situations, applied N is subjected to some volatilization losses when applications are not followed up by an irrigation event. Gaseous losses were found to be significant even on acidic soils and strongly influenced by source and N rates (Torello et al., 1983). Losses from sulfur coated urea were 0.2 and 2.3% of the applied N at rates of 85 and 260 lb/A, respectively. Prilled urea (small uniform pellets of urea) losses averaged 10% of the applied. Liquid applications of various formulations at lower rates (44 lb/A) reduced losses to 3 to 4.5%. Irrigating immediately following a surface N application is an important management practice. Bowman et al. (1987) found that up to 36% of the applied N was lost to NH3 volatilization if an irrigation event did not follow a liquid urea application. Sixty eight percent of the urea remained in the thatch. Applications of 0.4 and 3.6" of water within five minutes of application reduced gaseous losses to 8 and 1%, respectively.

RUNOFF OF FERTILIZER N APPLIED TO TURFGRASS

Runoff occurs when precipitation, less the interception, exceeds the infiltration rate. Effects of the runoff on nutrient movement is influenced by a number of factors and discussed by Spectrum Research (1990). In summary of that work, fertilizer loss is determined by a combination of factors: 1) volume of runoff; 2) timing, amount, and placement in relation to runoff events; and 3) the magnitude of the various N routes such as immobilization and volatilization. There is only a limited number of studies examining turfgrass runoff. Morton et al. (1988) observed only two runoff events from turf surfaces in a 2-year Rhode Island study. Concentrations of inorganic N ranged between 1-4 mg/L. Similar results were found elsewhere. Nitrate and ammonium concentrations and loads in runoff from both fertilized and non-fertilized plots were extremely low, although ammonium losses were influenced by application rate (Gross et al., 1990). The largest annual losses of NO_3, NH_4, and total N summed 0.9 lb/A. In one large rainfall event immediately after a fertilization, a significant amount of N was lost in runoff. Low NO_3 and ammonium runoff losses were also noted by Brown et al. (1982). Under the criteria for golf green construction, the likelihood for runoff is minimal. Other studies have been summarized elsewhere demonstrating that well maintained, dense turf is extremely effective in reducing, if not eliminating, runoff under most conditions (Gross et al., 1991; Cooper, 1990; Watschke and Mumma, 1989).

Runoff contributions from frozen Minnesota soils should be investigated. This would be practically critical following late fall or early spring N applications.

OTHER FACTORS INFLUENCING N LOSSES

Clipping management is an important consideration in the selection of annual application rates. Rosen et al. (1991) measured clipping yields and N contents of conservatively fertilized turf. Seasonal accumulation of total N was 1.4 lb 1000 ft^2. Actual N recovery by clippings varies with grass species and N management. Recoveries within the first year when utilizing soluble N were commonly 25 to 60% during a single year. Recoveries under rates of 220 lb/A averaged 46 to 59% over a 3-year period (Hummel and Waddington, 1981). Similar recoveries were noted elsewhere (Starr and DeRoo, 1981). Nitrogen tied up within clippings should be viewed as a slow release source and adjustments in application rates should be made.

Due to the significant amount of N within the clippings, it is important to keep them and other organic materials such as leaves out of surface waters. Contributions of N and as well as phosphorus can be high from storm sewers. Some of the organic formulations are being advertised as environmentally safe because the materials are natural. Although these products have been demonstrated to reduce leaching and gaseous losses, they still must be handled with caution. Any form of N, organic or inorganic, must be kept away from direct routes to surface waters.

SUMMARY AND CONCLUSION OF TURFGRASS CONTRIBUTIONS TO NITRATE CONTAMINATION OF WATER RESOURCES

Potential environmental risks associated with turf applied N appears to be minimal if application rates do not exceed the turf's physiological needs. Under intense management such as golf greens and high maintenance lawns, the development of a dense population and the development of the thatch layer reduces runoff to very low levels. Leaching losses are also commonly minimal because of the prolific root development, the increased moisture holding capacity directly below the thatch, and the turf's ability to utilize high rates of N. Because of the coarse soil texture of golf course greens, leaching contributions will more than likely be accelerated but these areas are small in relation to the rest of the land area. Actual research data from the courses indicates that leachate problems can be addressed with minimal changes and still maintain a functional turf. Cooper (1990) suggests that rather than threatening environmental quality, improved turf quality through judicious management can protect water quality compared to a poorly maintained turf or other land use. Skogley (1987) made a similar conclusion by stating "in reality, fertilizer applications maybe more beneficial in protecting ground water than contaminating it".

Leaching rates do not appear to be linearly correlated with N application rates as long as the annual rates do not exceed turfgrasses' physiological needs. Maximum amounts of required N were found to vary with management, residual soil N, and varieties but in general most research suggests that applied rates should not exceed 160 to 175 lb/A/year (3.6-4.0 lb/1000 ft^2). Good N nutrition insured good vigorous top growth and root development serving as an effective filter and a porous protective covering capable of minimizing runoff. Clippings added a significant amount of N and should be considered when selecting an annual application rate.

Runoff volumes under turf tend to be minimal during the growing season. Existing studies have concluded that N runoff losses are small in comparison to other avenues of N loss. Losses could be a potential problem if runoff occurred immediately after a fertilization event.

Timing of the N applications was determined to have an effect on turf physiology and N fate. Proper timing balanced the growth between the roots and the shoots. Lower N rates were used successfully when properly timed and still maintain good lawn color. Optimal timing under Minnesota growing conditions is late summer through mid-fall. The environmental consequences of late fall applications on cold soils should be explored. There is some danger that the homeowner or commercial applicator maybe getting the material on too late in the season. Studies under Minnesota conditions with varying lengths of time before the turf goes dormant should be addressed.

Source selection is extremely important under certain physical conditions and management criteria. Research data indicates that slow release sources should definitely be used in the following situations: applications on coarse textured soils; conditions when the thatch is thick and/or an irrigation immediately after application can not be applied; and when large, infrequent applications are made. For experienced lawn owners or turfgrass managers willing to make multiple fertilizer applications and are capable of making proper watering decisions, the more soluble and cheaper sources of N can be used with minimal effects on the environment. Any source of N, whether it is in the organic or inorganic form, must be handled with common sense and kept from direct contact from any water resources or non-target areas such as driveways and sidewalks.

Effects of irrigation management were determined to have a significant role in leaching losses. Unfortunately, homeowners would not have the essential information such as daily evapotranspiration or soil moisture information to irrigate correctly. Therefore, lawn care providers should limit the amount of lawn waterings. Watering devices triggered by soil moisture status rather than existing schedulers based on time periods would be an effective tool. Soil moisture sensors or portable tensiometers would provide important data but are probably too expensive and time consuming for the homeowner. They should be strongly encouraged for the professional groundkeepers and golf course managers.

Turfgrass types and varieties vary in their N needs. This is particularly true in the bluegrasses where some varieties require very high rates of N. Although the adoptability of the lower maintenance tall fescues in the colder climatic conditions is improving, Minnesota turf industry is still dependent on the cool season grasses. Although most lawn owners do not have the opportunity to select a particular variety nor do most know their existing variety, low maintenance types should be selected when the opportunity arises. These varieties should be promoted by the sod and seed dealers.

Literature focusing on leaching and runoff under turfgrass conditions are limited under actual Minnesota conditions. Despite the lack of such localized studies, the existing information appears relatively universal for the northern United States and it strongly suggests that turfgrass contributions is a very manageable problem. Only in worst case scenarios were there significant losses or elevated nitrate-N concentrations. Current N use and knowledge of its management by turfgrass clientele is extremely limited.

The fate of applied N on turf is not the only environmental considerations. Although the evidence is strong that N fertilization does not adversely affect water resources, the importance of proper management and environmental parameters such as pesticides, phosphorous, or other nutrients must not be overlooked. Although many of the articles reviewed did study some of the other parameters, the literature search within was focused on N. In general, the results for pesticide and other nutrients were parallel with the N conclusions. Also keeping any type of application away from impervious surfaces and from surface waters must be stressed. Clippings and other lawn litter such as leaves must be kept out of any surface waters also.

Literature Cited

Agnew, M.L. 1989. Groundwater, turf management, and public perception. Iowa turfgrass grower. March. Iowa State Univ.

Anonymous. 1991. Lawn Care Industry, Vol 15, No. 3, p.13.

Bowman, D.C., J.L. Paul, W.B. Davis and S.H. Nelson. 1987. Reducing ammonia volatilization from Kentucky bluegrass turf by irrigation. Hortic. Sci. 22:84-87

Brown, K.W., J.C. Thomas, and R.L. Duble. 1982. Nitrogen source effect on nitrate and ammonium leaching and runoff losses from greens. Agron. J. 74:947-950.

Brown, K.W., R.L. Duble, and J.C. Thomas. 1977. Influence of management and season on fate of N applied to golf greens. Agron. J. 769:667-671.

Cohen, S.Z., 1990. The Cape Cod study. Golf Course Manag., p. 26,28,32,34,38,40,42,44. Feb.

Cooper, R.J. 1990. Evaluating the runoff and leaching potential of turfgrass pesticides. Golf Course Manag., p. 6-7, 10, 12, 14, 16. Feb.

Exner, M.E., M.E. Burbach, D.G. Watts, R.C. Shearman, and R.F. Spalding. 1991. Deep nitrate movement in the unsaturated soze of a simulated urban lawn. J. Environ. Qual. 20:658-662.

Flipse, W.J., B.G. Katz, J.B. Lindner, and R. Markel. 1984. Sources of nitrate in ground water in a sewered housing development, central Long Island, New York. Ground Water 32:418-426.

Gold, A.J., W.R. DeRagon, W.M. Sullivan, and J.L. Lemunyon. 1990. Nitrate-nitrogen losses to groundwater from rural and suburban land uses. J. Soil and Water Cons. p. 305-310.

Gross, C.M., J.S. Angle, R.L. Hill, and M.S. Welterlen. 1991. Runoff and sediment losses from tall fescue under simulated rainfall. J. Environ. Qual. 20:604-607.

Gross, C.M., J.S. Angle, and M.S. Welterken. 1990. Nutrient and sediment losses from turfgrass. J. Environ. Qual. 19:663-668.

Hummel, Jr., N.W. and D.V. Waddington. 1981. Evaluation of slow-release nitrogen sources on Baron Kentucky Bluegrass. Soil Sci. Soc. Am.J. 45:966-970.

Hummel, Jr., N.W. and D.V. Waddington. 1984. Sulfur-coated urea for turfgrass fertilization. Soil Sci. Soc. Am. J. 48:191-195.

Landschoot, P.J. and D.V. Waddington. Response to turfgrass to various nitrogen sources. Soil Sci. Soc. Am. J. 51:225-230.

Morton, T.G., A.J. Gold, and W.M. Sullivan. 1988. Influence of overwatering and fertilization on nitrogen losses from home lawns. J. Environ. Qual. 17:124-130.

Nelson, K.E., A.J. Turgeon, and J.R. Street. 1980. Thatch influence on mobility and transformation of nitrogen carriers applied to turf. Agron. J. 72:487-492.

Petrovic, A.M. 1990. The fate of nitrogenous fertilizers applied to turfgrass. J. Environ. Qual. 19:1-14.

Rieke, P.E. and B.G. Ellis. 1974. Effects of nitrogen fertilization on nitrate movement under turfgrass. p. 120-130. In E.C. Roberts (ed.) Proc. 2nd Int. Turfgrass Res. Conf. ASA, Madison, WI. 19-21 June 1972. Blackburg, Va.

Rosen, C., P. Johnson, D. Birong, D. White and K. Ruser. 1991. Recycling of nitrogen from turfgrass clippings. Misc. Publ. 71. Mn. Agric. Exp. Sta., Univ of Mn. St. Paul, Mn.

Skogley, C.R. 1987. Turf-growing doesn't appear to harm the environment. Univ. of Rhode Island Coop. Extension Service.

Spectrum Research. 1990. Environmental issues related to golf course construction and management: A literature search and review. Spectrum Research, Inc., Duluth, Mn.

Starr, J.L. and H.C. DeRoo. 1981. The fate of nitrogen applied to turfgrass. Crop Sci. 21:531-536.

Street, J.R. 1988. New concepts in turf management. Landscape Management 27:38, 40, 44, 46.

Synder, G.H., B.J. Augustin, and J.M. Davidson. 1984. Moisture sensor-controlled irrigation for reducing N leaching in bermudagrass turf. Agron. J. 76:964-969.

Taylor, D.H., C.J. Rosen, and D.B. White. 1989. Fertilizing lawns. AG-FO-3338. Mn. Ext. Ser., Univ. of Mn.

Taylor, D.H., D.B. White, W.C. Stienstra, and M.E. Ascerno, Jr. 1989. The home lawn. AG-MI-0488. Mn. Ext. Ser., Univ. of Mn.

Torello, W.A., D.J. Wehner, and A.J. Turgeon. 1983. Ammonia volatilization from fertilized turfgrass stands. Agron. J. 75:454-456.

Watschke, T.L. and R.O. Mumma.1989. The effect of nutrients and pesticides applied to turf on the quality of runoff and percolating water. A Final Report to the U.S. Dept. Int., U.S. Geol. Sur. ER 8904. Env. Res. Res. Inst. Penn State Univ.

Wehner, D.J., J.E. Haley, and D.L. Martin. 1988. Late fall fertilization of Kentucky bluegrass. Agron. J. 80:466-471.

NITROGEN CONTRIBUTIONS FROM FOREST, PRAIRIE AND MISCELLANEOUS SOURCES

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EFFECTS OF FORESTS AND FOREST MANAGEMENT ON GROUND WATER

Overview

Nitrogen losses have been monitored in a number of forest studies although it appears from the available literature that there is more concern about the cation (calcium, magnesium, and potassium) cycles than N nutrition/degradation. Nitrogen fertilizer applications to Minnesota forest systems are minimal[1]. In general, leaching losses under unfertilized forested conditions appear to be lower than most other land uses. Pionke and Urban (1985) monitored a small Pennsylvania watershed which represented a mixture of land uses (cropland, 57%; forestry, 35%, and pasture, 8%). Nitrate-N concentrations under forest lands did not exceed 4 mg/L and had a mean value of 0.7 mg/L. The mean concentration under cropland was 3.0 mg/l. Over 200 wells were monitored over a 2-year study period to investigate the effects of land use and geologic patterns on Michigan ground water (Richardson, 1979). A classification scheme was devised and divided land use into 9 categories. The three forest categories all ranked low in comparison to any type of agricultural land use. Nitrate-N concentrations of these three groups averaged between 0.3-0.5 mg/L.

Gold et al. (1990) compared NO_3 losses during a 2-year study in Rhode Island under forest, corn, septic systems, and lawns. Nitrate-N concentrations were 0.2, 13, 68, and 0.6 mg/L and annual leaching losses were 1.3, 59, 43, and 3 lb/A for the respective sources. In a Georgia comparative land use study, Beck et al. (1985) determined that NO_3-N concentrations under forests were generally less than 1 mg/L compared to concentrations ranging from 4-6 mg/L under agriculture. Elsewhere, nitrate-N concentrations under forested conditions have been reported below 1 mg/L (Hill, 1982; Weil et al., 1990) below 1.5 mg/l (Ritter and Chirnside, 1987), to less than 3 mg/L (Moody, 1990).

Management/Species Effects

Krajenbrink et al. (1988) investigated the dilution effect of forests which were interdispersed within agricultural lands in the Netherlands. Despite a rather homogeneous environment within the forests, leachate solutions below them were found to be very heterogeneous in terms of N concentrations. Concentrations from ground water and solution samplers were also found to be highly dependent upon tree species. The highest NO_3-N concentrations (21 mg/L) were found under coniferous forests.

Juergens-Gschwind (1989) reviewed the impacts of forestry in Europe. Three independent studies found leaching losses of N to be within the range of 5 to 14 lb/A/year. A key factor in understanding losses under forested conditions is the relationship between soil pH and the nitrification process. Forest litter, particularly under conifers, is acidic and the soil pH continues to decrease as

1.Personal communication with Dr. Sandy Verry, Principle Forest Hydrologist, Nor Central Forest Exp. Station, Grand Rapids, MN.

the forest matures. Bacteria responsible for nitrification become less active under acidic conditions. Any factors which alter nitrification (ie. clear cutting, fires, disease) will ultimately affect leaching losses. Tree species have an influence on N transformation and movement. Within this European review, the literature indicated that leaching losses increased going from a pine to beech to Douglas fir ecosystem. Forest thinning and clearcutting can temporarily reduce water/nutrient uptake and increase nitrification. These authors report nitrification can increase 10 to 40 fold unless hindered by extreme acidity, anaerobic conditions or a high carbon/nitrogen ratio.

Verry[2] found that clear cutting of aspen on Minnesota's mineral soils increased surface runoff by about 30% and estimated that 15 years of regrowth would be required to lower the runoff to preharvest values. Nitrate concentrations remained stable but the total loading to surface waters was increased by the same percentage. No changes in leachate concentrations were observed.

EFFECT OF NATIVE GRASSLANDS ON GROUND WATER

Organic matter content of grassland soils tends to be considerably higher than forest soils. Stevenson (1982) summarized the reasons for this occurrence with the key ones being: 1) larger quantities of raw material of humus synthesis are produced under grass; 2) nitrification is inhibited in grassland soils, therefore preserving N; and 3) inadequate aeration under grassland production, thereby contributing to organic matter buildup. Despite the fact that the total N content of prairie soils (4,500-14,000 lb/A) exceeds that of arable soils (2,700-9,000 lb/A), leaching losses below native prairie are characteristically very low (Juergens-Gschwind, 1989). Losses are similar to those already reported under living alfalfa (Chapter G) and low maintenance turf (Chapter K). The potential for ground water pollution is low due to the general N-deficient status and low N turnover rates (Kenney, 1982). Hallberg (1989) conducted a national literature search and compared ground water effects from different land uses. Nitrate-N concentrations under natural prairie areas were commonly less than 0.1 mg/L.

Effects of land management, soil and geologic features were studied within 30 watersheds in a three state area (Kansas, Oklahoma, and Texas) to determine the net impact on water quality (Smith et al., 1987). Mean NO_3-N and NH_4-N concentrations across three locations which were in native grasses ranged from 0.5-3.6 and 0.04-0.34 mg/L, respectively.

Surface runoff under prairie conditions are also similar to those already discussed in turf (Chapter K). Timmons and Holt (1977) quantified runoff under native prairie in Big Stone County. Flow-weighted concentrations of organic N and inorganic N were 2.6 and 1.3 mg/L. Total N lost annually was 0.7 lb/A and 63-88% of the average nutrient load was transported by snowmelt.

There is a small amount of existing research which indicates that plowing down of the prairie grasslands has a similar effect on ground water as does termination of legume crops. Cameron and Wild (1984) found that about 90 lb/A of NO_3-N had leached below a 3' depth over a two winter period as a result of plowing grasslands. Bergstrom (1987) utilized lysimeters and tile-drained field plots to determine that considerable leaching (>40 lb/A) occurred during a 5 month period following plowing.

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2. Personal communication with Dr. Sandy Verry, Principle Forest Hydrologist, No Central Forest Exp. Station, Grand Rapids, MN.

NATURAL INORGANIC SOURCES

Atmospheric Deposition

Precipitation

Atmospheric precipitation commonly contains NH_4, NO_2, NO_3, and organically bound N (Stevenson, 1982). These additions are normally too small to sustain crop production but are considerably important in mature ecosystems such as undisturbed natural forests and native prairie. Since natural systems are not subjected to large N removal commonly associated with the physical harvesting of cropland, atmospheric deposition is a critical component in the delicate N balance. Actual quantities of "natural" N deposition are only estimates since these types of measurements have only been collected after centuries of burning fossil fuels.

Important sources of atmospheric NH_3 include volitization from land surfaces, fossil fuel combustion and natural fires (Stevenson, 1982). It has been theorized that NO_3 in precipitation occurs from electrical discharge during thunderstorm activity. Other sources, as reviewed by Stevenson, have predicted that only 10 to 20% of the NO_3 in precipitation can be accounted for by electrical discharge.

Researchers dealing in depositional N studies emphasize that there is considerable uncertainty associated with characterizing N concentrations. Concentrations vary seasonally as well as within individual storms. Because of this variability, the most reliable estimates of N input should be calculated for each event, summed over time and then expressed as a loading value (Schepers and Fox, 1989). Demonstrating this point, these authors compared Minnesota data (NO_3 and NH_4 concentrations and loadings) from Morris and Waseca during 1976-78. Although concentrations were similar at the two sites, the net annual loadings[3] were quite different (3.5 and 7.3 lb/A, respectively) due to the large differences in rainfall (14 and 22"/year, respectively). At both sites, the contributions from NO_3 and NH_4 were approximately equal.

Currently there are four active Minnesota sites in the National Atmospheric Deposition Program/National Trends Network (NADP/NTN, 1990). They are: 1) the Marcell Experimental Forest in Itasca County which has been operating since 1978[4]; 2) Fernberg in Lake County (1980); 3) Camp Ripley in Morrison County (1979); and 4) Lamberton in Redwood County (1979). Major cation and anion deposition chemistry has also been collected at these sites.

Nitrate-N and NH_4-N concentrations[5] ranged between 0.2-0.3 and 0.4-0.6 mg/L, respectively, for 1989. Depositions of NO_3-N and NH_4-N for the same time period were 1-2 and 2-2.6 lb/A, respectively. Nationally, the 1989 NO_3-N depositions ranged from 0.2-4 lb/A and NH_4-N depositions ranged from 0.4-3.0 lb/A (See Figure L-1 through L-4).

3.Monitoring time periods were from April through November of each year.

4.Operational dates from personal communication with Gwen Scott, Data Manager of the NADP/NTN Program, Fort Collins, CO.

5.Flow-weighted concentrations.

Earlier Minnesota studies reported heavier depositions of N (Anonymous, 1981) than the NADP report. One important difference was that sampling sites within the earlier investigation were centered within areas dominated by agriculture. Annual N depositions during 1975-78 from Waseca, Lamberton, Morris, and Staples were 10, 12, 5, and 3 lb/A, respectively. Annual inorganic N deposition (1975-78) across the Midwest ranged from 4 to 18 lb/A. Depositions at the Cloquet Forestry Center and the Marcell Experimental Forest during a similar time period were both 4.5 lb N/A/year. Another depositional study estimated inorganic N loading via precipitation in west-central Minnesota to be 6.7 lb/A (Burwell et al., 1975).

Inorganic concentrations were found to vary tremendously during a given rain storm with a rapid decline several minutes after the initial rain (Francis and Schepers, 1987). This data suggests that rainfall has a "scrubbing effect" on the atmosphere. Due to the fact that concentrations are the highest at the start of a rainfall event, most of the N should be infiltrated rather than being carried off in the runoff (Schepers and Fox, 1989).

Nitrogen in precipitation is also difficult to assess on a local level due to the profound effect from large feedlots, power generating plants, and industrial areas (Legg and Meisinger, 1982). Hoeft et al. (1972) found elevated deposition in areas adjacent to Wisconsin barnyards. Total N additions ranged from 12 to 27 lb/A.

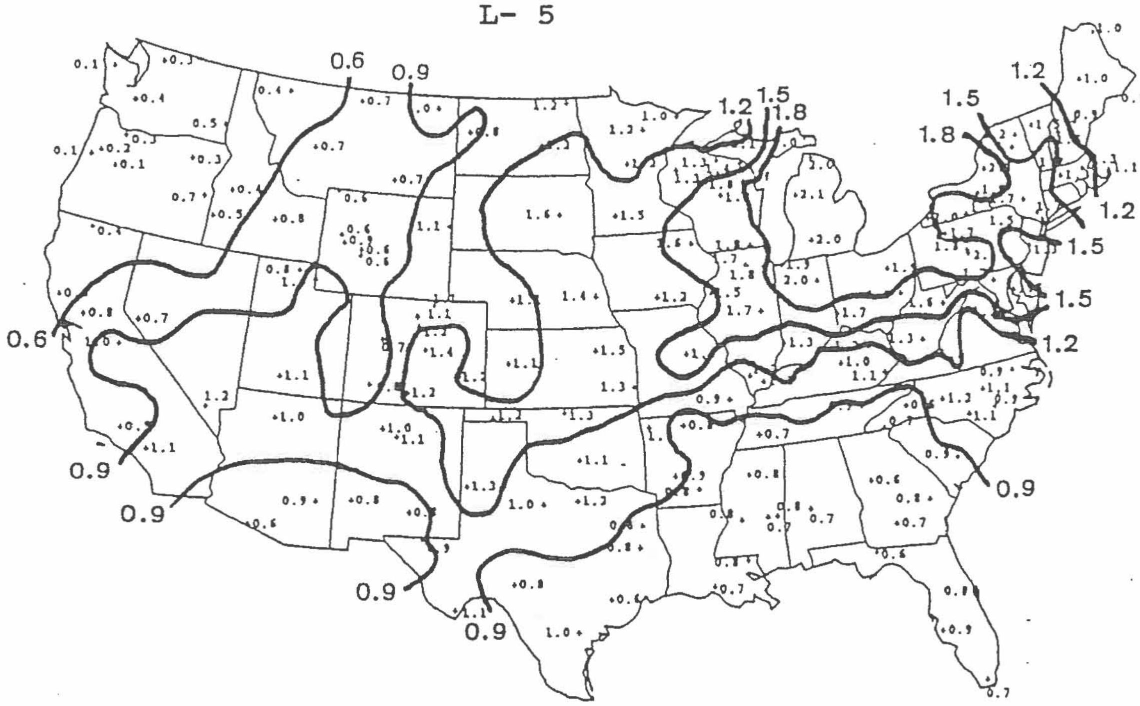

Figure L-1. 1989 annual precipitation-weighted mean NO_3 ion concentrations (mg/L). Please note to divide these values by 4.4 to convert to NO_3-N units. Figure taken from NADP/NTN, 1989.

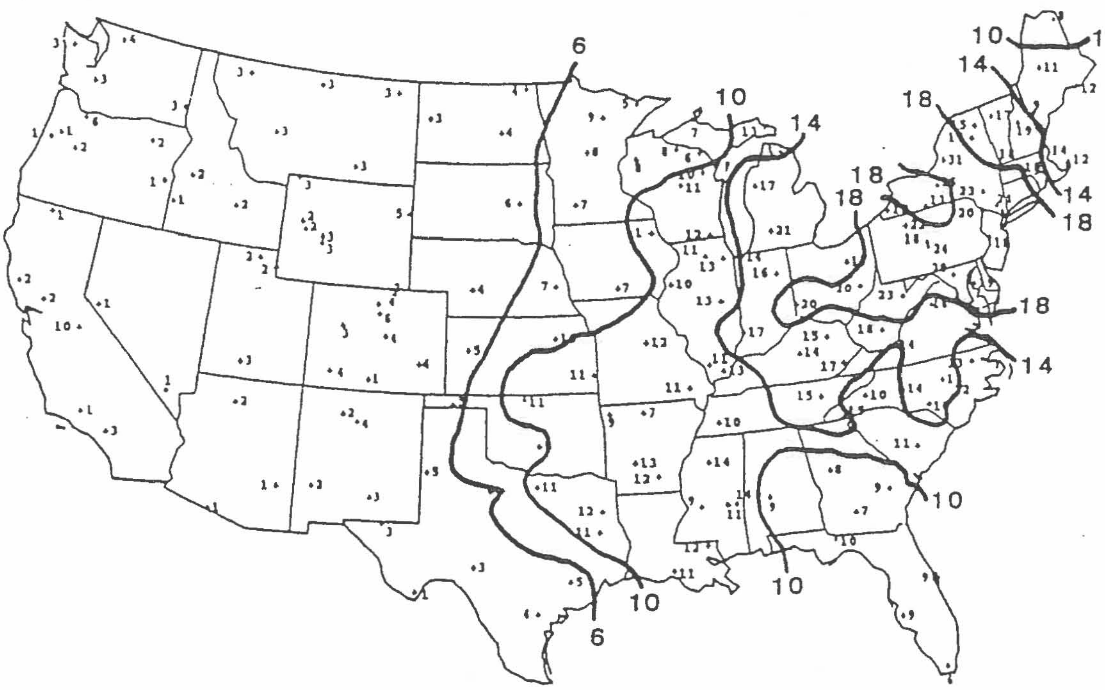

Figure L-2. 1989 annual NO_3 ion deposition (kg/ha). Please note to divide these values 4.9 to convert to NO_3-N deposition in lb/A. Figure taken from NADP/NTN, 1989.

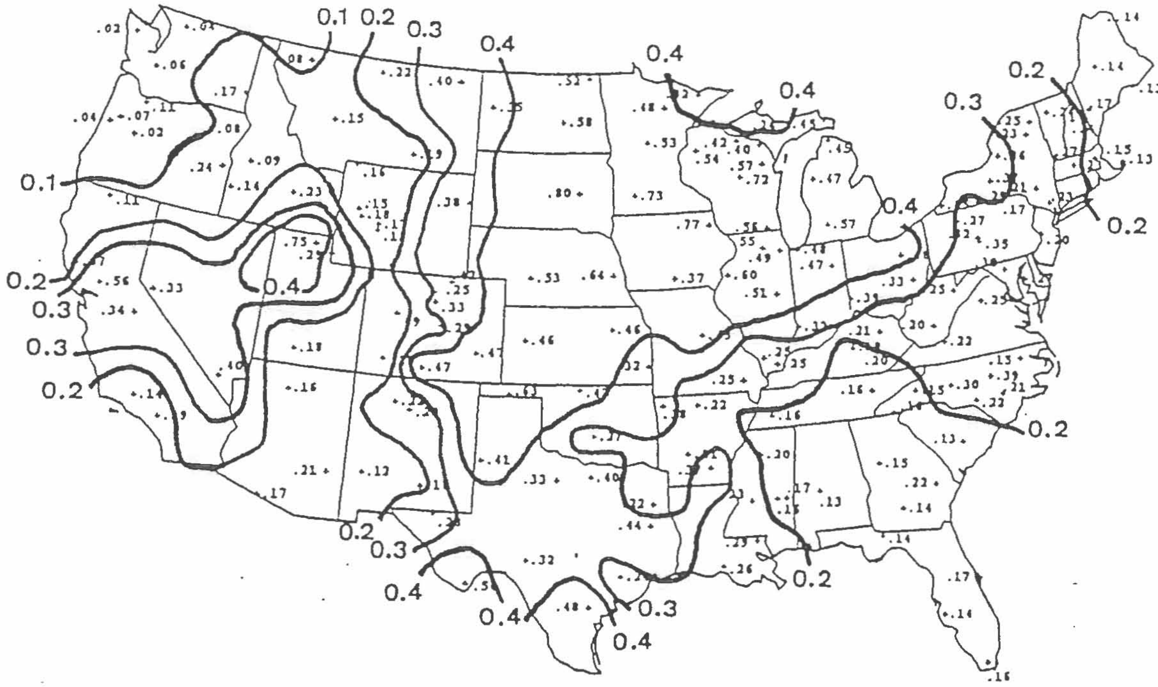

Figure L-3. 1989 annual precipitation-weighted mean NH_4 ion concentrations (mg/L). Please note to divide these values by 1.29 to convert to NH_4-N units. Figure taken from NADP/NTN, 1989.

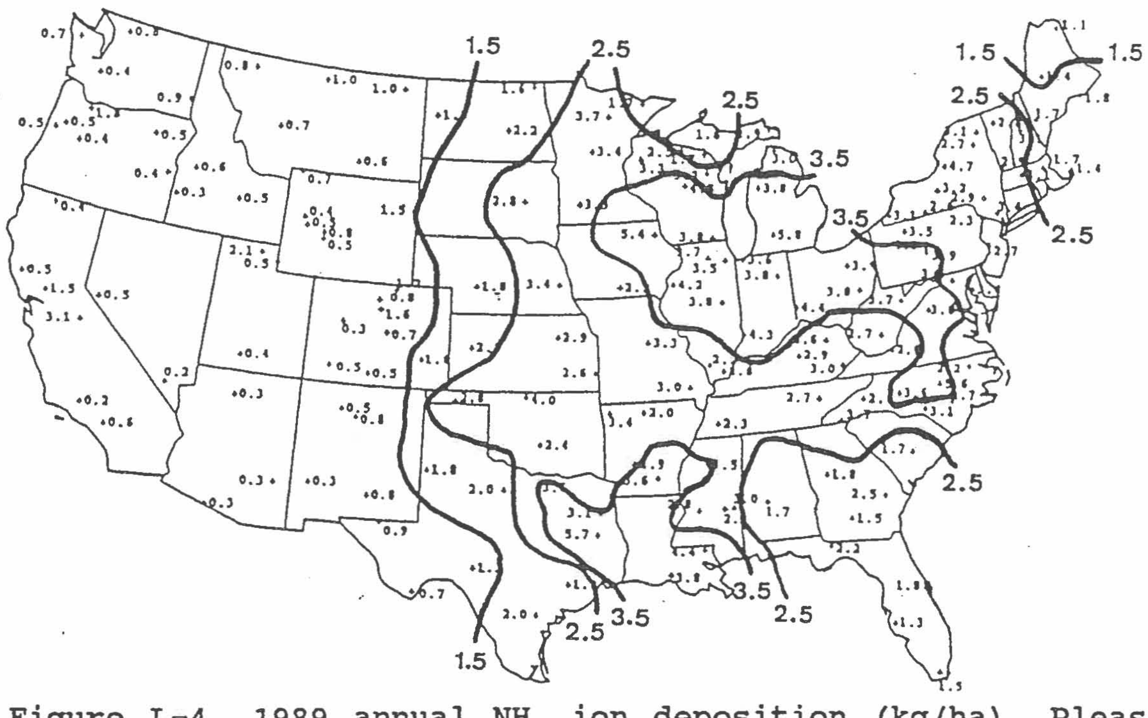

Figure L-4. 1989 annual NH_4 ion deposition (kg/ha). Please note to divide these values 1.44 to convert to NH_4-N deposition in lb/A. Figure taken from NADP/NTN, 1989.

Absorption

Although direct absorption from the atmosphere by soil, water, or plants is not normally considered a significant process in the N cycle, a number of studies have found high absorption contributions in areas where atmospheric NH_3 has been elevated. More information can be found in Legg and Meisinger (1982) although two of these cases will be mentioned here. A New Jersey study found that soil absorption rates were between 20 to 75 lb/A/year in industrial areas of that state. Another study determined that the NH_3 volatilized from nearby feedlots had a significant impact of surface waters. One lake adjacent to a large feedlot absorbed 65 lb NH_3-N/A/year. A number of other studies enforced the concept that the contribution of atmospheric gases, either through precipitation or absorption, to soil and water may be appreciable in areas where the concentration of NH_3 in the atmosphere is greater than normal.

Biological Fixation

Symbiotic Fixation

The word "symbiosis" refers to the relationship of two dissimilar organisms living together in intimate association and the cohabitation being mutually beneficial (Hausenbuiller, 1972). Minnesota's commonly grown legumes (alfalfa, soybeans, and clover) form a symbiotic relationship with the Rhizobium bacteria. Rhizobium converts atmospheric N_2 to plant useable N and, in turn, obtains soluble carbohydrates from the legume host. As mentioned earlier in this report (Chapter G), actual amounts of NH_2 fixed by leguminous crops has been difficult to quantify due to complicating factors as legume species, Rhizobium-host relationships, soil moisture, soil NO_3 levels, and other nutritional factors. Contributions from alfalfa and soybeans were previously discussed. Nitrogen fixation is not limited to only these crops. Other crops, such as the trefoils and vetches, are currently small in Minnesota acreage but may gain in popularity with the renewed interest in legumes.

There are native Minnesota plant species capable of symbiotic fixation; tag alder would be the most common. The related red alder is a pioneer species found in the Pacific Northwest and is capable of fixing greater than 100 lb/A/year of atmospheric N (Van Miegroet and Cole, 1984). Nitrate-N concentrations within the leachate below these N-rich stands commonly exceeded 10 mg/L. Grigal[6] speculated that the amounts fixed by tag alder would be similar to red alder but the fate of N fixed by this species is unknown at this time. Tag alder inhabits low lying areas which are commonly moist to saturated. Wet conditions and high organic carbon amounts are conducive for the denitrification process.

Non-symbiotic Fixation

The process in which free-living microorganisms utilize N_2 is termed non-symbiotic or "free fixing". Some blue-green algae and various anaerobic and aerobic bacteria are capable of this process. Blue-green algae can be found in virtually any environmental setting where sunlight is available for photosynthesis. Environmental effects from blue-green algae and their relationship with the N:P ratio in lakes is reviewed by EPA (1991). Importance of algae to agricultural soils is limited. The general consensus of many soil

6.Personal communication with D.F. Grigal, Department of Soil Science, Universit of Minnesota, St. Paul, MN.

scientists is that no more than 6 lb/A/year are added by the combined activities of non-symbiotic N_2-fixing microorganisms (Stevenson, 1982). Schepers and Fox (1989) reviewed the literature and concluded that more research is required before it is possible to predict the quantity of N provided from this process. Several studies cited high fixation values in temperate and tropical soils which remain wet during most of the cropping season.

Geologic

As mentioned in Chapter A, most of the earth's N is associated with igneous rock. Much of this N is tightly bound within silicate mineral lattices. High NO_3 concentration levels are seldom found in rock and any inorganic releases are extremely limited. Accumulations of NO_3 and NH_4 from most naturally occurring sources (ie. organic matter) are seasonal in many soils but leaching, denitrification and biological uptake generally limit longer accumulations (Marrett et al., 1990). Yet in a few isolated areas of the United States, naturally occurring NO_3-N rich deposits have caused elevated concentrations in ground waters (Moody, 1990; Marrett, 1990). These sites have been limited to the extreme arid areas of the country. Marrett et al. (1990) also identified volcanic deposits being high in NO_3.

Due to subhumid classification in which Minnesota lies, it would be extremely unlikely that geologic deposits would affect the state's drinking water supplies.

Soil Organic Matter Contributions

Introduction

Prior to the dramatic increase in N fertilizer usage (about 1945), crop production became increasingly dependent on the capacity of the soil to provide N (Stanford, 1982). Now a renewed interest in developing improved methods for assessing N availability has been stimulated by the environmental consequences of over-application and economic considerations. Soil organic matter is an important contributor in supplying N to both agronomic crops and native vegetation; contributions of inorganic N are estimated to exceed 670,000 tons/year (Chapter F, Figure F-1). Hausenmiller (1972) illustrated approximations of the N content across United States. Percent N concentrations in Minnesota ranged from: 0.10-0.20% in north central to northeast; 0.20-0.30% in south central to south east; and greater than 0.3% in the western portion of the state. These percentages translate into a range from 4,000 to greater than 12,000 lb/A of organic N in the surface foot of soil.

Despite the great importance of this N pool, factors affecting plant available N are not totally understood by the research community. Due to the reliance of biological processes, the complexities within the entire N cycle are enormous. As a result, soil N availability is more difficult to predict than the other macronutrients. Jansson and Persson (1982) state that "Individual N transformations have been studied extensively within such basic sciences as biochemistry, microbiology, and plant physiology. The result has been the accumulation of considerable information concerning the various transformations, their environmental demands, their mechanisms, the intermediates, and the end products. In contrast, little is known regarding the ecological unity of the transformations-the complete process of plant production on soils."

Mineralization rates and the subsequent conversion of NH_4 to NO_3 are strongly influenced by soil moisture and temperature. Counter balancing some of the mineralization effects is the process of immobilization which was previously defined (Chapter A) as the transformation of inorganic compounds into the organic state. The difference between the two processes is dependent upon the amount of energy available for microbial activity. Generally this is gauged by the carbon to nitrogen (C/N) ratio and is commonly used as a descriptive parameter for both plant and soil materials. This ratio is commonly defined as the weight of organic carbon to weight of total N. A C/N ratio in well-balanced soil biomass should be about 25/1 (Jansson and Persson); the C:N ratio of undisturbed topsoil is generally 10 to 12/1 (Tisdale and Nelson, 1971). A higher ratio generally indicates slow decomposition and N deficiency is more likely due to immobilization. Jansson and Persson (1982) advocate using a energy/N relationship rather than the C/N ratio since the later can be misleading due to the fact that some of the C and N constituents of organic matter undergoing decomposition are not readily available to microorganisms. It is readily apparent that there are large differences in decomposition rates between agricultural crops; these differences may be attributed to both the C/N ratios and chemical composition of the plant material (Waggar et al., 1985; Zielke and Christenson, 1986) as well as residue particle size (Vigil et al., 1991).

Assessment of Nitrogen Availability in Agricultural Soils

A crucial challenge facing N management is establishing methodologies for predicting the potentially mineralizable component of the organic N pool. Early efforts for method development in the post World War II era were short-lived for two reasons according to Stanford (1982): 1) contributions of mineralized N during the cropping season were commonly masked by the presence of varying amounts of residual N from past fertilization; and 2) low fertilizer costs discouraged development and use of effective N soil tests. More economical and environmentally sound management practices could be developed if mineralization could be accurately described and predicted under field conditions (Honeycutt et al., 1988). Useable techniques would be particularly valuable in measuring N supplying capabilities of soils with recent additions of manure or legumes (Fox and Piekielek, 1984).

Stanford (1982) reviewed methods for assessing soil N availability and divided these methods into the following categories: 1) methods for estimating residual mineral N in soil; 2) incubation methods including short term methods under aerobic and anaerobic conditions; and 3) chemical indexes for estimating N availability. Methods for measuring residual soil NO_3-N or estimating mineralized N based on the pre-sidedress nitrate test have already been discussed (Chapter G). Although the test provides useful information, determination of inorganic N does not provide a reliable means of estimating the amount of N released during the cropping season. Various biological and chemical methods currently exist, however none of these tests currently appear to be universally accepted and reliable enough to warrant its routine use in soil testing laboratories (Fox and Piekielek, 1984).

Mineralization rates can be estimated by various incubation techniques under controlled environmental conditions; however, these techniques are very time consuming and often not practical for making N fertilizer recommendations. Chemical indexes, consequently, have been developed as a means for making rapid estimates of N availability. The underlying assumption here is that the various chemical extractants utilized remove certain fractions of relatively easily decomposable soil organic N. Soil type and pH are probably the major factors influencing the effectiveness of any given extractant. Hadas et al. (1986) also found that soil depth, calcium carbonate amounts, and particle surface area all influenced results. Juma and Paul (1984) tested several different techniques and concluded that: the extracted N pool can only partially explain the source of N being mineralized; N is mineralized from several pools; and that there is a remote possibility that a single extractant may eventually be found that can extract the variety of N compounds undergoing mineralization/immobilization. Cabrera and Kissel (1988) found that the method by Stanford and Smith (1972) overpredicted mineralization rates in many cases and speculated that non-representative soil water contents and alterations through soil preparation before incubation both significantly affected results.

There are a number of other methods suggested for mineralization predictions since Stanford's exhaustive review (1982). Several studies have measured CO_2 evolution as a means for predicting net N mineralization (Gilmour et al., 1985; Castellanos and Pratt, 1981). Temperature has a profound effect on C and N mineralization; N mineralization rate constants developed by Stanford et al. (1975) reportedly vary by about twofold for each 10 degrees C in temperature. Honeycutt et al. (1988) found that a heat unit concept model was useful for collectively describing mineralization of C fractions with contrasting decomposability. Relationships have also been found between the quantity of decomposable organic N and enzyme activity (Hadas et al., 1986). Other methods, such as the burial of soils within polyethylene bags (Westermann and Crothers, 1980; Smith et al., 1977) or the burial of ion exchange resin bags (Binkley, 1984), have yielded mineralization information but these techniques appear to be mainly useful for verifying other quicker procedures.

Several assessments of various chemical extraction techniques are currently being made under Minnesota conditions (Vivekanandan and Malzer, 1991; Schmitt et al., 1990). The efficacy of the hot KCl extraction method (Gianello and Bremner, 1986) and the phosphate-borate method (Gianello and Bremner, 1988) were compared to aerobic incubation procedures and field responses on an Esterville sandy loam (Vivekanandan and Malzer, 1991). Crop rotation and tillage influenced the inorganic N level in soils at planting time but not mineralizable N as measured by chemical or biological indices. Schmitt et al. (1990) compared the results of the soil nitrate test, soil inorganic N, and phosphate-borate extractable N for predicting corn yields. Results were dependent upon parent material but overall the phosphate-borate method yielded the highest correlations.

University of Minnesota soil scientists have developed a process oriented computer model called NCSOIL (Molina et al., 1983; Hadas et al., 1987). NCSOIL computes the changes in organic forms of C and N, NH_4, and NO_3 concentrations which result from residue decomposition, mineralization, immobilization, nitrification, denitrification, and nonsymbiotic N fixation. The utility of this model for aiding in making N recommendations is not clear at this time.

Effects of Tillage

Effects of tillage on water quality due to alterations in soil infiltration, soil moisture content, and runoff were previously discussed in Chapter G. Tillage also has a significant impact on soil N mineralization/immobilization relationships. Cultivation of native prairie soils decreases soil organic matter and total N contents. In a 16-year study, Follett and Schimel (1989) found that tillage systems significantly altered total N and microbial biomass in the top 4" of soil. Total N decreased to 73, 68, and 50% in no-till, stubble mulch and moldboard plow treatments, respectively, to that found in native sod. Nitrogen immobilization was the highest in native sod; increased tillage density decreased the ability of soil to immobilize and conserve mineral N. Similar results were determined by Kettler et al. (1991). Elliott et al. (1986) also found that mineralized N under plowed soils was more subject to leaching than mineralized N produced under no-till management. Aulakh et al. (1991) determined that residue type, placement, degree of incorporation and soil moisture are important factors in controlling N availability.

Tracy et al. (1990) observed greater residual NO_3 accumulations under no-till systems in the top 1" surface soil; tillage did not influence N mineralization below the 2" soil depth. El-Haris et al. (1983) studied the interactions between residue types and tillage systems (moldboard plow, chisel, and no-till). The N mineralization potential was greater for the chisel plow and no-till in the top 2" but less than that from the moldboard system in the 2-6" layer. The net result in the entire plow zone was no affect on the N mineralization potential due to tillage or crop rotations. In a Canadian study, Carter and Rennie (1984) found that mineralization-immobilization turnovers were not significantly affect by tillage differences (no-till vs shallow tillage).

Assessment of Nitrogen Availability in Forest Soils

Although it appears that Minnesota forests do not commonly receive N fertilization, the literature indicates that commercial forests of the Pacific Northwest and northeast United States do apply N fertilizers since N is considered the most common limiting factor (Kraske and Fernandez, 1990; Federer, 1983; Myrold 1987). Development of reliable soil tests for estimating available N for tree uptake and growth would be an important forestry management tool. Various biological and chemical tests have been evaluated and generally are similar to those types of tests conducted on agricultural soils (Kraske and Fernandez, 1990; Myrold, 1987).

MISCELLANEOUS MAN-INDUCED N SOURCES

Landfills

A number of items containing N have entered landfills, including such things as feces and urine from diapers and domestic animals, lawn clippings and other yard waste, food scraps, and unused fertilizer. Therefore, leaking landfills would be expected to release some N to underlying soils.

There are 371 permitted solid waste facilities in Minnesota; 55 of these being Superfund sites. About 270 of the 371 sites are active landfills with monitoring wells installed. However, there are only 62 facilities that have wells installed to accurately define differences between ground water quality upgradient and downgradient of the landfill. For this study, NO_3 and NH_4 concentrations at 41 of the 62 sites were reviewed from MPCA files.

Fifteen of the 41 facility files reviewed for N impacts were Superfund sites. Nine of these 15 sites showed appreciable increases in nitrogen between upgradient and downgradient wells. Average increases in NH_4-N ranged from 11 to 58 mg/L, with a total average increase at all sites of 19 mg/L. Nitrate-N increases ranged from 15 to 38 mg/L, averaging 20 mg/L. The other six sites had either no significant nitrogen differences between up and down-gradient wells or had erratic trends with no obvious increases.

Of the 26 non-Superfund facility files reviewed, no sites had NO_3+NH_4-N increases of more than 5 mg/L between upgradient and downgradient wells, and most sites had increases of less than 2 mg/L.

In summary, it appears from this review of existing data that Superfund site landfills often contribute significant amounts of NO_3 and NH_4 to ground water. Other landfills in the state have not been shown to appreciably affect ground water nitrogen levels.

Spills

Spills of N fertilizer products may occur where products are manufactured, stored, handled or applied; this includes large manufacturing sites, co-ops, distributorships and farmland. Spilled liquid N fertilizer products are more difficult to clean up and provide greater potential for ground water contamination than spilled dry products, which must first be solubilized before significant amounts can move within the hydrologic system. There are approximately 450 Minnesota Department of Agriculture (MDA) permitted liquid commercial fertilizer storage facilities, including manufacturing facilities, in Minnesota. Storage capacity at these facilities ranges from a single 5,000 gallon tank to 8,720,000 gallons in seven tanks.[7] Data is not available on liquid storage capacities at private farming operations.

Minnesota Statute 18D.103, subd. 1 (1990) requires that all fertilizer spills, no matter how small, be reported to the MDA. The MDA has maintained a data base on reported spills since 1989. Information that should be submitted upon report of an incident includes product spilled, volume spilled, location of spill, and

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7. Personal communication with John Peckham, Supervisor, Facilities Unit, Agrono Services Division, Minnesota Department of Agriculture, St. Paul, MN.

source or cause of spill. In many cases, the volume spilled is not known and can be only estimated; furthermore, the product spilled is not always fully or correctly identified. In 1990, 55 fertilizer spills and sixteen mixed fertilizer-pesticide product spills were reported. Estimated spill volume ranged from 0.8 pounds N (in urea base mix liquid fertilizer) to 11,800 pounds N (in 28% liquid urea-ammonium-nitrate).[8]

Many fertilizer spills may go unreported, thus the reported number of spills probably underestimates the actual frequency of N fertilizer spills that occur in Minnesota.[9] The percentage of unreported spills is not known. The total N loading in Minnesota from fertilizer spills is, consequently, unknown.

When a spill is reported to the MDA, the Incident Response Unit oversees the clean-up of the spill and, if possible, the recovery of the spilled product. In general, clean up involves excavation of contaminated soils and landspreading of excavated soils on agronomic fields at agronomic rates. This mechanism potentially lessens or eliminates the total N-loading at any given spill site. Spills which are cleaned up with the approval of the Incident Response Unit may be eligible for reimbursement from the Agricultural Response and Reimbursement Account (ACCRA), created under Minnesota Statute 18E.03 (1990) and maintained in the State Treasury. If a site is eligible, reimbursement may be granted for up to 90% of the clean-up costs greater than $1,000 and less than $100,000, and up to 100% of the costs greater than $100,000 but less than $200,000.[10] There are three sources of revenue to ACCRA, a) surcharge fees pesticide registrations, b) surcharge fees on fertilizer inspection fees, and c) surcharge fees on the pesticide applicator and storage license applications. While this program is still developing, preliminary response is positive. It is anticipated that spills will be reported not only because it is required by the MDA, but also because of limited liability, provided statutory violations were not the cause of the incident.

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8. Personal communication with Roger Mackedanz, Consultant, Technical Response Unit, Agronomy Services Division, Minnesota Department of Agriculture, St. Paul, MN.

9. Personal communication with Roger Mackedanz, Consultant, Technical Response Unit, Agronomy Services Division, Minnesota Department of Agriculture, St. Paul, MN.

10. Minnesota Statute 18E.04, Subd. 4 (1990).

Literature Cited

Anonymous. 1981. Nutrient concentrations and accumulations in precipitation over the north central region. North central regional research publication No. 282.

Aulakh, M.S., J.W. Doran, D.T. Walters, A.R. Mosier, and D.D. Francis. 1991. Crop residue type and placement effects on denitrification and mineralization. Soil Sci. Soc. Am. J. 55:1020-1025.

Beck, B.F., L. Asmussen, and R. Leonard. 1985. Relationship of geology, physiography, agricultural land use, and ground-water quality in southwest Georgia.

Bergstrom, L. 1987. Nitrate leaching and drainage from annual and perennial crops in tile-drained plots and lysimeters. J. Environ. Qual. 16:11-18.

Binkley, D. 1984. Ion exchange resin bags: Factors affecting estimates of nitrogen availability. Soil Sci. Soc. Am. J. 48:1181-1184.

Burwell, R.E., D.R. Timmons, and R.F. Holt. 1975. Nutrient transport in surface runoff as influenced by soil cover and seasonal periods. Soil Sci. Soc. Am. Proc. 39:523-528.

Cabrera, M.L. and D.E. Kissel. 1988. Evaluation of a method to predict nitrogen mineralized from soil organic matter under field conditions. Soil Sci. Soc. Am. J. 52:1027-1031.

Cameron, K.C. and A. Wild. 1984. Potential aquifer pollution from nitrate leaching following the plowing of temporary grassland. J. Environ. Qual. 13:274-278.

Carter, M.R. and D.A. Rennie. 1984. Nitrogen transformations under zero and shallow tillage. Soil Sci. Soc. Am. J. 48:1077-1081.

Castellanos, J.Z. and P.F. Pratt. 1981. Mineralization of manure nitrogen-correlation with laboratory indexes. Soil Sci. Soc. Am. J. 45:354-357.

El-Haris, M.K., V.L. Cochran, L.F. Elliott, and D.F. Bezdicek. 1983. Effect of tillage, cropping, and fertilizer management on soil nitrogen mineralization potential. Soil Sci. Soc. Am. J. 47:1157-1161.

Elliott, E.T., P.W. Tracy, G.A. Peterson, and C.V. Cole. 1986. Leaching of mineralized N is less under no-till cultivation. p. 53-54. In Trans. Int. Congr. Soil Sci., 13th, Hamburg, W. Germany. Vol. 6. Congr. Centurum, Hamburg.

Environmental Protection Agency, 1991. Nitrogen action plan (draft). PM-221.

Federer, C.A. 1983. Nitrogen mineralization and nitrification: depth variation in four New England forest soils. Soil Sci. Soc. Am. J. 47:1008-1014.

Follett, R.F. and D.S. Schimel. 1989. Effect of tillage practices on microbial biomass dynamics. Soil Sci. Soc. Am. J. 53:1091-1096.

Fox, R.H. and W.P. Piekielek. 1984. Relationships among anaerobically mineralized nitrogen, chemical indexes, and nitrogen availability to corn. Soil Sci. Soc. Am. J. 48:1087-1090.

Francis, D.D. and J.S. Schepers. 1987. Verification in nitrate and ammonium content in precipitation rainfall events. p. 26 In Agronomy Abstracts. ASA, Madison, WI.

Gianello, C. and Bremner, J.M. 1986. Comparison of chemical methods of assessing potentially available organic nitrogen in soil. Communication in Soil Science and Plant Analysis. 17:215-236.

Gianello and Bremner, 1988. A rapid steam distillation method of assessing potentially available organic nitrogen in soil. Communications in Soil Science and Plant Analysis. 19:1551-1568.

Gilmour, J.T., M.D. Clark, and G.C. Sigua. 1985. Estimating net nitrogen mineralization from carbon dioxide evolution. Soil Sci. Soc. Am. J. 49:1398-1402.

Gold, A.J., W.R. DeRagon, W.M. Sullivan, and J.L. Lemunyon. 1990. Nitrate-nitrogen losses to groundwater from rural and suburban land uses. J. Soil Water Conserv. 2:305-310.

Hadas, A., S. Feigenbaum, A. Feigin, and R. Portnoy. 1986. Distribution of nitrogen forms and available indices in profiles of differently managed soil types. Soil Sci. Soc. Am. J. 50:308-313.

Hadas, A., J.A.E. Molina, S. Feigenbaum, and C.E. Clapp. 1987. Simulation of nitrogen-15 immobilization by the model NCSOIL. Soil Sci. Soc. Am. J. 51:102:106.

Hallberg, G.R. 1989. Nitrate in groundwater in the United States. In Follett (ed) Nitrogen management and ground water protection. Elsevier, New York. p. 35-74.

Hausenbuiller, R.L. 1972. Soil science: Principles and practices. Brown Co., Dubuque, Ia.

Hill, A.R. 1982. Nitrate distribution in the ground water of the Alliston region of Ontario, Canada. Ground Water Vol. 20, No.6:696-702.

Hoeft, R.G., D.R. Kenney, and L.M. Walsh. 1972. Nitrogen and sulfur in precipitation and sulfur dioxide in the atmosphere in Wisconsin. J. Environ. Qual. 1:203-208.

Honeycutt, C.W., L.M. Zibilske, and W.M. Clapam. 1988. Heat units for describing carbon mineralization and predicting net nitrogen mineralization. Soil Sci. Soc. Am. J. 52:1346-1350.

Jansson, S.L. and J. Persson. 1982. Mineralization and immobilization of soil nitrogen. In F.J. Stevenson (ed.) Nitrogen in agricultural soils. Agronomy 22. Am. Soc. of Agron., Madison, WI.

Juergens-Gschwind, S. 1989. Ground water nitrates in other developed countries (Europe)-Relationships to land use patterns. In Follett (ed.) Nitrogen management and ground water protection. pp 75-137. Elsevier Pub., New York, New York.

Juma, N.G. and E.A. Paul. 1984. Mineralizable soil nitrogen: Amounts and extractability ratios. Soil Sci. Soc. Am. J. 48:76-80.

Kenney, D.R. 1982. Nitrogen management for maximum efficiency and minimized pollution. In F.J. Stevenson (ed) Nitrogen in agricultural soils. p. 605-650. Agronomy 22. Am. Soc. of Agron., Madison, WI.

Kettler, T.A., J.W. Doran, J.F. Power, and L.N. Mielke. 1991. Soil N availability and uptake by corn under varying crop rotation and tillage regimes. p. 269. In Agronomy abstracts. ASA, Madison, WI.

Krajenbrink, G.J.W., D. Ronon, W. Van Duijvenbooden, M. Magaritz, and D. Wever. 1988. Monitoring of recharge water quality under woodland. J. Hydrology 98:83-102.

Kraske, C.R. and I.J. Fernandez. 1990. Conifer seedling growth response to soil type and selected nitrogen availability indices. Soil Sci. Soc. Am. J. 54:246-251.

Legg, J.O. and J.J. Meisinger. 1982. Soil nitrogen budgets. In F.J. Stevenson (ed.) Nitrogen in agricultural soils. Agronomy 22. Am. Soc. of Agron., Madison, WI. p. 503-566.

Marrett, D.J., R.A. Khattak, A.A. Elseewi, and A.L. Page. 1990. Elevated nitrate levels in soils of the eastern Mojave Desert. J. Environ. Qual. 19:658-663.

Molina, J.A.E., C.E. Clapp, M.J. Shaffer, F.W. Chichester, and W.E. Larson. 1983. NCSOIL, a model of nitrogen and carbon transformations in soil: description, calibration, and behavior. Soil Sci. Soc. Am. J. 47:85-91.

Moody, D.W. 1990. Groundwater contamination in the United States. J. of Soil and Water Conserv. 2:170-179.

Myrold, D.D. 1987. Relationship between microbial biomass nitrogen and a nitrogen availability index. Soil Sci. Soc. Am. J. 51:1047-1049.

National Atmospheric Deposition Program. 1990. Precipitation chemistry in the United States 1989. National Atmospheric Deposition Program/National Trends Network. Natural Resource Ecology Laboratory, Fort Collins.

Pionke, H.B. and J.B. Urban. 1985. Effect of agricultural land use on ground water quality in a small Pennsylvania watershed. Groundwater 23: 68-80.

Richardson, C.T. 1979. Ground water quality in northern Michigan ecosystems as affected by land use patterns and geologic characteristics. Agro-Ecosystems 5:231-243.

Ritter, W.F. and A.E. Chirnside. Impact of land use on ground-water quality in southern Delaware. Groundwater.

Schepers, J.S. and R.H. Fox. 1989. Estimation of N budgets for crops. In Follett (ed) Nitrogen management and groundwater protection. Elsevier, New York. p. 221-246.

Schmitt, M.A., G.W. Randall, G.W. Rehm, and G.L. Malzer. 1990. Corn production and nitrogen use: are soil tests improving N recommendations? Soils, fertilizer and agricultural pesticides short course. December 12-13, 1990. Minneapolis, Mn.

Smith, S.J., J.W. Naney, and W.A. Berg. 1987. Nitrogen and ground water protection. In Fairchild (ed) Ground water quality and agricultural practices. Lewis Pub., Chelsea, MI. p. 367-375.

Smith, S.J., L.B. Young, and G.E. Miller. 1977. Evaluation of soil nitrogen mineralization potentials under modified field conditions. Soil Sci. Soc. Am. J. 41:74-76.

Stanford, G. 1982. Assessment of soil nitrogen availability. In F.J. Stevenson (ed.) Nitrogen in agricultural soils. Agronomy 22. Am. Soc. of Agron., Madison, WI.

Stanford, G., and S.J. Smith. 1972. Nitrogen mineralization potentials of soils. Soil Sci. Soc. Am. Proc. 36:465-472.

Stanford, G., M.H. Frere, and R.A. Vander Pol. 1975. Effect of fluctuating coefficient of soil nitrogen mineralization. Soil Sci. 115:321-323.

Stevenson, F.J. 1982. Origin and distribution of nitrogen in soil. In F.J. Stevenson (ed) Nitrogen in agricultural soils. p. 67-122.

Timmons, D.R. and R.F. Holt. 1977. Nutrient losses in surface runoff from a native prairie. J. Environ. Qual. 6:369-373.

Tisdale, S.L. and W.L. Nelson. 1971. Soil fertility and fertilizers. Macmillan Company, New York, NY.

Tracy, P.W., D.G. Westfall, E.T. Elliott, G.A. Peterson, and C.V. Cole. 1990. Carbon, nitrogen, phoshorus, and sulfur mineralization in plow and no-till cultivation. Soil Sci. Soc. Am. J. 54:457-461.

Van Miegroet, H. and D.W. Cole. 1984. The impact of nitrification on soil acidification and cation leaching in a red alder ecosystem. J. Environ. Qual. 13:586-590.

Vigil, M.F., J.S. Schepers, and J.W. Doran. 1991. Mineralization-immobilization of N in soils amended with corn residues of various particle sizes. p. 279. In Agronomy abstracts. ASA, Madison, WI.

Vivekanandan, M. and G.L. Malzer. 1991. Crop rotation, tillage and N fertilizer influence on nitrogen availability to corn. p. 304. In Agronomy abstracts. ASA, Madison, WI.

Waggar, M.G., D.E. Kissel, and S.J. Smith. 1985. Mineralization of nitrogen from nitrogen-15 labeled crop residues under field conditions. Soil Sci. Soc. Am. J. 49:1220-1226.

Weil, R.R., R. A. Weismiller, and R.S. Turner. 1990. Nitrate contamination of groundwater under irrigated coastal plain soils. J. Environ. Qual. 19:441-448.

Westermann, D.T., and S.E. Crothers. 1980. Measuring soil nitrogen mineralization under field conditions. Agron. J. 72:1009-1011.

Zielke, R.C. and D.R. Christenson. 1986. Organic carbon and nitrogen changes under selected cropping systems. Soil Sci. Soc. Am. J. 50:363-367.

RESPONSE - WELL OWNER OPTIONS AND GOVERNMENT PROGRAMS

Lead Agency: Minnesota Pollution Control Agency
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WELL OWNER RESPONSE

Options for Communities with Unacceptable Nitrate Levels

In order to protect human health, EPA requires that public water supply systems[1] be monitored for nitrate (NO_3) and other parameters. Public water supply systems with NO_3-N in excess of 10 mg/l must notify residents using the water.

Currently there are seven community water systems that distribute water to residents which have NO_3-N in excess of 10 mg/l. Several cities with elevated NO_3 in the past were able to drill new wells that currently provide water with acceptable NO_3 levels. The cost of drilling new wells varies greatly depending on geologic materials penetrated, well depth, and distance that water must be piped to the distribution system.

A common way for public water systems to provide acceptable water is to blend water high NO_3 with low NO_3 water.

Treatment of community water to remove NO_3 has not usually been considered a feasible option, although one town in SW Minnesota is exploring this option. Another possible option for a community would be to treat only a portion of the water and make this water available to families with infants less than six months old. While this option would be more cost effective than treating all water, the logistics of delivering water to the community would have to be considered. It is also possible that a community could truck in water from another area, or provide low NO_3 commercially bottled water to those most at risk.

Three technologies are recognized by EPA for physically removing NO_3 from water (U.S. EPA, 1991). Reverse osmosis is one approved method that has a 67 to 95 percent removal rate which costs from $1.50/1000 gallons for a community type system (about $150 per year per household) to $5.90/1000 gallons for a small system. Electro-dialysis reversal system will remove 51 to 92 percent of the NO_3 with costs similar to reverse osmosis. Anion exchange systems have a 65 to 99 percent removal rate with treatment costs from $0.77/1000 gallons for a large system to $3.40/1000 gallons for a small system (U.S. EPA, 1991).

Des Moines, Iowa has passed a proposal to purchase a $5 million anion exchange system to treat nitrate (Hubert, 1991). Average household monthly water bills are expected to increase $2.21 ($26.52/year).

Rural water systems are another option being utilized by seven groups in Minnesota. These systems were developed to deal with both water quantity and water quality concerns. A rural water system can be described as a water pumping, treatment, and distribution system delivered to a combination of communities and rural homes.

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[1] A public well is defined as any well providing piped water for human consumption and serving a minimum of 15 service connections or 25 persons daily for 60 days of the year.

One rural water system in southwestern Minnesota serves 9000 people in 14 towns and hundreds of farms. Over 1100 miles of pipes have been installed for this system. In the past, the Federal Government (FmHA) provided 35 to 50 percent cost share grants for qualifying areas to install rural water systems. This funding is becoming much more restricted. Without this cost share money, the hookup costs for any one farm in southwestern Minnesota is estimated to be \$15,000-\$16,000. If the farmers and rural homes cannot afford to get onto the system, costs to the small cities would increase substantially.

In some parts of the state, especially southwestern Minnesota, it is very difficult for cities, farmers and rural homeowners to find an adequate supply of good quality water. One rural water supply system spent over \$100,000 in 1990 trying to locate adequate water quantity/quality throughout a seven county area. Most of the deeper aquifers had aesthetic water quality problems associated with high iron, manganese, sulfate or hardness and that most of the shallow aquifers had NO_3 problems or could not supply an adequate amount of water for the rural water system. Most of the existing wells in the rural water systems are relatively shallow and susceptible to surface contamination, including NO_3. The future potential for rural water systems depends largely on protecting the existing shallow aquifers from degradation.

Researchers are experimenting with new methods for treating high NO_3 water. One such method is to treat NO_3 within the aquifer, which is referred to as in-situ treatment. In-situ treatment was previously discussed (Chapter C).

The preferable long-term solution for community systems is pollution prevention. Implementation of wellhead protection around community wells is advised.

Domestic Water Supplies

Water Testing

All domestic well owners should be made aware of testing services and have the opportunity to have their water analyzed. Health clinics should be promoting water testing to pregnant women and parents of infants. Every parent of infants should know the NO_3 content of their drinking water and the implications of drinking the water.

All new wells must have nitrate analyses performed on the water. The responsibility to assure that this test is performed rests with the well contractor.

Several counties have conducted NO_3 testing programs within the last few years. Some of these programs and the testing results were previously described (in the "Nitrate in Ground Water - Existing Conditions" chapter).

Options for Homeowners

When a private well has elevated NO_3 levels, the owner often has four options 1) drill a new well, 2) buy a treatment system, 3) buy bottled water or obtain water from a different source, or 4) continue drinking high NO_3. Boiling does not remove NO_3, but concentrates it thereby increasing the NO_3 concentration.

Drill New Well

Depending on many different variables, drilling a deeper well may or may not be an option for obtaining water with acceptable nitrate concentrations. This option can be quite expensive. In bedrock aquifers in southeastern Minnesota, the cost of drilling a new well is sometimes over $10,000. Deeper aquifers with low NO_3 may have other problems such as high iron. In other parts of the state there may not be an economically feasible option to drill a deeper well due to underlying igneous and metamorphic rocks.

In some situations, moving a well to a different location on the property and keeping the well at the same depth may be sufficient to obtain adequate water quality. Well relocation is appropriate if the old well is thought to be improperly constructed or the well currently intercepts plumes from NO_3 sources such as septic systems or feedlots. Knowing ground water flow directions is important when relocating a well away from potential sources.

Bottled Water

Obtaining water from a different source (bottled water, city water, etc.) is usually less expensive than drilling a new well. The two major disadvantages with this option would be 1) the inconvenience of hauling in the water, and 2) the possibility of having trouble conducting real estate transactions with the residence in the future. In 1987, Consumer Reports tested 50 brands of bottled water for NO_3 and found no brand with NO_3-N over 10 mg/l.

Treatment Systems for Private Water Supplies

Nitrate concentrations in water **cannot** be reduced by water softeners, boiling, chlorination, or most filtering. Nitrate levels are most often reduced by three methods: 1) reverse osmosis, 2) distillation, or 3) anion exchange. With any treatment system frequent water testing is necessary to determine whether the system is still working effectively and some maintenance is required. Since nitrate can cause acute health effects in newborn infants within one or two days, there is a significant risk associated with failure of a treatment device.[1]

In a **reverse-osmosis system,** pressure forces water through a membrane which extracts impurities. Reverse osmosis (RO) can also reduce arsenic, lead, inorganic mercury, asbestos and radium. A carbon filter can usually be added to an RO system to reduce organic chemicals. Owners must follow the manufacturer's maintenance and replacement recommendations in order for the system to function properly. Filters should be replaced regularly and membranes need replacement every five years. The membrane will plug sooner if the water contains much iron and manganese. Another maintenance chore is to frequently empty the storage tank. An RO system requires at least 40 pounds of water pressure. The price range is usually $450 to $1000 for a unit installed in a home.

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[1] Personal communication with Dan Wilson, Minnesota Department of Health.

A **distillation system** boils water, condenses the vapor, and collects the condensed vapor. Nitrate, in addition to dissolved solids, metals, and minerals remain in the boiling water, while concentrations of these parameters in the condensed water are low. Boiling also kills microorganisms. There is a possibility of increasing volatile organic chemicals with a distillation system. About 1 gallon is distilled every two hours for a cost of about 20 cents per gallon. An acid cleaner must be used after every 20 to 200 gallons to prevent mineral build-up. Costs of the units vary from $250 to $1200.

With an **ion exchange** system (this is not a water softener), NO_3 ions can be exchanged with chloride ions on the surface of resin beads. Anion exchange systems, while relatively inexpensive, are only effective for a short period of time before the unit becomes saturated and needs to be recharged with a concentrated sodium chloride brine (McCasland, et al. 1985). If the water is high in sulfate, the effectiveness of ion exchange systems for NO_3 removal is reduced.

Continue Drinking High Nitrate Water

The fourth option of continuing to drink the high NO_3 water certainly is the least expensive and troublesome option. However, this option is not reasonable if infants are to be drinking the water. Long-term health risks for adults consuming excess NO_3 are possible, but unconfirmed (See Chapter A). In addition, if NO_3 is detected at high concentrations in the well, there is a greater likelihood of other contaminants in the water. Nitrate is often used as an indicator of a well susceptible to other contaminants.

MDH Recommendation

Because of the necessary maintenance of treatment systems and the fact that bottled water and treatment systems are only temporary solutions, the Minnesota Department of Health recommends to homeowners that a new well depth or location is the best option to consider for high NO_3 wells (where low-nitrate water can be obtained by a new well).

FEDERAL, STATE AND LOCAL RESPONSE

Federal Nitrogen Action Plan

In response to a growing national concern about the ecological and health impacts of nitrogenous compounds, the U.S. Environmental Protection Agency (EPA) is developing a nitrogen action plan (EPA, 1991). A draft report was released in March 1991 and will be finalized during the fall of 1991. EPA's intent is to develop a strategy to focus and coordinate activities and help states deal more effectively and efficiently with all sources of Nitrogen in order to limit risks posed by contamination.

The 158 page draft report is divided into an executive Summary, Recommendations, and a technical appendix. The technical appendix is divided into four sections: 1) risk characterization, 2) sources and relative importance of sources, 3) pollution prevention, and 4) remediation and treatment. Much of the technical appendix gives a national perspective of many issues discussed in this report of Nitrogen in Minnesota Ground Water.

Recommendations taken directly from EPA's DRAFT Nitrogen Action Plan report are listed on the following seven pages.

EPA Draft (3/5/91) Recommendations

EPA Nitrogen Action Plan Recommendations

The Nitrogen Action Plan workgroup's recommendations are organized into five categories. 1) develop State Nutrient Management Programs, 2) improve on-farm nitrogen management to protect water quality, 3) improve public and private drinking water quality, 4) increase control of point sources through current regulatory authority, and 5) research in areas of uncertainty. These recommendations are not ranked in any order of importance. They are all equally part of the plan.

EPA would ensure implementation of the recommendations through three basic approaches: direct EPA action, nonregulatory and regulatory; EPA encouraging or requiring state action; and EPA working with the U.S. Department of Agriculture (USDA) and other federal agencies. The Nitrogen Action Plan would be implemented in two phases. Phase I emphasizes using current regulatory authorities pollution prevention techniques, and research. Activities under Phase II would begin if these voluntary efforts and current legal authorities were insufficient.

Direct Agency Action

Under Phase I, EPA would use portions of the Safe Drinking Water Act (SDWA), Clean Water Act (CWA), and Toxic Substances Control Act (TSCA), Coastal Zone Management Act (CZMA) to implement the recommendations. Although the authority is present for most recommendations, additional money or reorientation of current resources is necessary in many programs. Direct EPA action is required to some degree in order to implement recommendations in each of the five categories. Phase II recommendations would all be implemented under increased EPA authority.

State Action

States that rank sources of nitrogen compounds as major sources of ground or surface water contamination in their assessments would develop programs that adequately address those sources both from a pollution prevention and a drinking water remediation perspective. EPA will work with the states through guidance, grant agreements, and technical assistance to implement the Nitrogen Action Plan recommendations.

Many of the actions addressing surface water contamination by nitrogen compounds would be implemented by the states under the §319 Nonpoint Source Program of the CWA. EPA guidance under the Coastal Zone Management Act will include many of these recommendations. §319 grant monies for implementing these actions can be used as incentives for state participation by adding nitrogen management as a rating factor in grant guidance, although this may require additional appropriations. If states have pesticide management plans, this will be coordinated with their nutrient management programs.

The recommendations that address ground water will be implemented under, or in coordination with Comprehensive State Ground-Water Protection Programs (CSGWPPs) and §319 Nonpoint Source Programs.

Other Federal Agencies

EPA will work in close cooperation with the USDA, USGS, and TVA (National Fertilizer and Environmental Research Center) to implement many of the recommendations. Recommendation #2 would be implemented through USDA programs. Under Phase I, we propose that EPA form a workgroup with USDA to develop and implement and programs that will improve fertilizer, manure, and feed management to promoted formally through memoranda of understanding, intra and inter-agency research initiatives. Coordinated research is essential.

EPA also has an interest in working with USDA on drinking water issues in recommendation #3 since Cooperative Extension Service agents work with private well owners and Farmer's Home Administration provides loan guarantees and makes grants to small community drinking water systems.

Phase I

1. **State Nutrient Management Programs**

   EPA will include nitrogen-related problems among those considered for action under Nonpoint Source Programs, State Comprehensive Ground-Water Protection Programs, Wellhead Protection Programs, Pesticide Management Plans, and Coastal Zone Management Plans. To maintain eligibility for EPA grants, states would be required to consider and identify nitrogen-related problems for action under these programs. States would then implement nutrient management activities within these programs to prevent further water quality degradation from nitrates and related compounds. EPA will provide technical assistance documents, guidance on development and implementation of nutrient management programs, and grant guidance documents to the states.

EPA Draft (3/5/91) Recommendations

State Program Elements:

1.1 Identify high risk watersheds through §319 (including locations where ground-discharge significantly affects surface water quality) and vulnerable wellhead areas.

1.2 In high risk areas, require farmers to keep records on yield, yield goals, application rates of fertilizer and manure, perform and keep records on tests for all field applications of nitrogen (soil, tissue, manure, and sludge tests as they become available).

1.3 Require farmers to use practices in #1.2 to better match fertilizer application with supplemental N needs.

1.4 Require specific timing and crop management practices for fertilizer and manure application, where appropriate (e.g., side-dress, fall/winter bans, and fall cover crops) in high risk areas.

1.5 Require best irrigation management practices, including anti-backsiphoning devices and calibration of irrigation to water/nutrient needs.

1.6 Where states have been delegated National Pollution Discharge Elimination System (NPDES) authority, focus on the most problematic feedlots regardless of size (manure management, land application, and storage and/or composting requirements). Increase compliance with NPDES regulations through enforcement and other activities. Use general permits where operations are numerous.

1.7 Develop innovative funding mechanisms (e.g., fees on sources of N) to assist public water systems and domestic well owners with treatment or the development of an alternate source of water. Funds from a State Revolving Fund could also be used to address animal waste BMPs.

1.8 Promote New Source Performance Standards as a pollution prevention technique for new livestock operations and expansion of existing operations (e.g., alternative markets for manure, composting).

1.9 Develop improved guidance for auditing and managing septic systems and wells to support state comprehensive ground water protection programs. The Guidance should address how to site septic systems in a manner that prevents baseflow to surface water of N levels above those acceptable for aquatic life.

1.10 Develop regulations for secondary containment and storage of nitrogen fertilizers.

2. **Farm Nitrogen Management**

Through the Presidents' Water Quality Initiative (WQI), the 1990 Farm Bill, EPA's Agriculture Policy Committee and other forums, EPA will collaborate with USDA to develop and implement voluntary, cost-share best management practices (BMPs) that will improve the efficiency of fertilizer use.

2.1 Assist USDA in its accelerated program to calibrate and implement soil and manure tests.

2.2 Encourage and expand recordkeeping (realistic yield goals, fertilizer application; yields; manure, sludge, food processing residue, N tests); include soil tests when available.

2.3 Expand Water Quality Initiative cost-share monies for appropriate manure lagoon liners and storage facilities, on-farm biomethanation plants, composting systems, manure spreaders, etc. where cost effective. Assess and revise existing SCS specifications to assure efficient use of federal resources in constructing storage facilities.

2.4 Support financial incentives for vegetative filters (CRP, WQIP, ACP).

2.5 Encourage the Soil Conservation Service to modify its national standard for earthen manure ponds to require liners to protect ground water in high risk areas.

2.6 Encourage USDA to offer easements to retire cropping rights within Wellhead Protection Areas by using the Environmental Easement Program in the 1990 Farm Bill.

3. **Remediation and Treatment**

EPA will work with state, federal, and private agencies to improve the quality of public and private drinking water supplies. Some actions can be taken by EPA through current authority under the Safe Drinking Water Act (SDWA). Other recommendations will require collaboration and cooperation, rather than regulation.

Public Water Supply

3.1 Increase federal and state enforcement actions against public water systems with violations of the nitrate standard set under the SDWA.

3.2 Require development of enforceable limits on fertilizer use (an other nitrogen inputs) in wellhead protection areas established under the SDWA and provision of bottled water to infants in return for exemptions from the MCL for small public water systems with nitrate violations.

3.3 Encourage states to develop innovative funding (tax, fee on sources of nitrogen, etc.) to assist public water systems and domestic well owners to treat water, provide an alternate source, or buy easements.

3.4 EPA to enter into a Memorandum Of Understanding with the Food and Drug Administration to require bottled water companies to monitor at the same frequency as Public Water Supplies and for the same contaminants.

3.5 Encourage states to develop wellhead protection programs to protect public wells from all sources of contamination as required under §1428 of the SDWA.

3.6 Pursue adoption of requirements for wellhead protection where public wells are financed by federal or state grants or loans (e.g., FmHA).

Domestic Water Supply

3.7 Encourage states to implement specific action to protect private wells, i.e. well construction codes, well driller certification, well testing requirements, sanitary surveys, financial aid, alternative water (infants and pregnant women), septic system siting, and land use restrictions to protect water quality. Many of these actions will be necessary in order for a state to meet the required elements of a SCGWPP.

3.8 Encourage states/lending agencies to require well testing before real estate transfers and for new wells.

3.9 Encourage states to consider adopting the approaches used under their Wellhead Protection Programs for public water wells to protect densely-settled areas relying on geographically clustered private wells.

4. Point Source Control/Management

This recommendation focuses on using EPA's current regulatory authority under the Clean Water Act (CWA), the Safe Drinking Water Act (CWA), and the Toxic Substances Control Act (TSCA) more effectively to deal with sources of nitrate contamination. Some additional authority would also be required.

4.1 Under the Class V well underground injection control program of the SDWA, require BMPs on cropland and greenhouses that are drained by agricultural drainage wells.

4.2 Require water quality based permits for feedlots and greenhouses. Strengthen NPDES permitting for feedlots regulated under CWA authority. Include a land application and manure storage component protective of surface and ground water in permits.

4.3 Require anti-backsiphoning devices on fertigation systems.

4.4 Revise the CWA to eliminate the point source exemption for irrigation return flows so that EPA can target those categories of flows or geographic areas with the greatest potential for serious environmental damage.

4.5 Under TSCA authorities consider using the product stewardship to require fertilizer manufacturers to develop programs on proper handling and use of fertilizers. Begin a regulatory investigation on requiring fertilizer dealerships to store and handle fertilizer to better protect water quality.

4.6 Move up the timetable in the Water Quality Standards Framework to develop nutrient guidance for water quality standards by 1993.

4.7 Add a requirement to the EPA Operating Guidance that during a state's triennial review, the state adopt numeric ammonia standards for water where designated uses are impaired due to ammonia. Encourage adoption of state-wide standards.

5. **Research**

In order to better understand the risks of nitrogen compounds and effective ways to deal with these risks, research must continue. These recommendations identify key areas where more research is needed either by EPA or through increased coordination with other federal agencies. Adoption of particular recommendations may depend on predictions of future contamination of shallow or deep aquifers.

5.1 Fill in data gaps on health effects using a TSCA test rule or EPA/other federal agency funds.

5.2 EPA along with other federal agencies such as USGS, USDA, and NOAA should work to jointly improve understanding of fate and transport, including aerial deposition, nitrogen soil loadings, and waste load allocations in surface waters. Plans should be developed on a land use by land use basis including intensively managed crop lands and unmanaged forest ecosystems to identify the processes most important in determining the environmental processing of nitrogen. The plans would include the consideration of:

- in situ denitrification rates and mechanisms in the saturated zone and below the root zone in the unsaturated zone.
- fate and transport modeling in the saturated zone and below the root zone.
- estimates of the depth and age of nitrate contamination of the saturated zone and predictions of peak contaminant levels in deep aquifers under several nitrate management scenarios.
- determine the benefits of adding organic matter to the soil.
- fate and transport modeling in soil with emphasis on computing N mass balance and transformation rates.

5.3 Develop new technologies and improve existing technologies for water supply and wastewater treatment.

- Improve efficiency of drinking water treatment to reduce costs, especially for small systems.
- Improve wastewater technology, including use of constructed wetlands.
- Develop/evaluate alternative septic tank designs.

5.4 Research to improve manure management:

- USDA to research cost-effectiveness of innovative manure and septage uses and distribution, for areas in which land area suitable for application is limited.
- Evaluate the nutrient content of manure and how it changes over time in relation to the ability of the plant to take up N.

5.5 Evaluate the effectiveness, i.e., risk communication, economic efficiency, financial impacts, health and environmental efficacy of implementing the Nitrogen Action Plan.

5.6 Work with USDA to evaluate effectiveness of nutrient best management practices for water quality, including ground water discharge into surface water.

5.7 Determine where economically efficient application of fertilizers and manures will still adversely affect water quality.

5.9 Evaluate information on fertilizer use by turf growers and lawn care companies obtained under TSCA to determine the relative importance of non-agricultural fertilizer use as a water pollution source.

Phase II

EPA would implement a second set of water protection activities if Phase I proved insufficient. For example, additional measures would be needed if other agencies fail to adopt voluntary recommendations, if voluntary measures are inadequate, if state enforcement of regulatory requirements is lacking, or if further research reveals that the health or ecological risks associated with nitrate contamination is more severe than current assessments indicate.

Examples:

° Create state revolving loan fund and grant program to assist small PWSs with no other recourse in providing alternative supplies and installing treatment facilities. [Will require a federal infusion of start-up money.]

° Use TSCA and SDWA to limit fertilizer applications in targeted areas.

° Implement a nationwide tax on sources of nitrate and use the proceeds to help contaminated water suppliers, buy easements on highly vulnerable land, cost-share manure storage and composting facilities, cost-share appropriate use of compost, among others.

° Obtain legislative authority to require farmers to develop nutrient management plans in watersheds where nutrients impair or threaten water quality.

° Require farmers to adopt nutrient best management practices in order to be eligible for farm subsidy payments.

State and Local Programs

Overview

There are two major types of approaches for responding to the issue of protecting water quality. The first type of program is designed to respond to problems from specific sources or contaminants. Examples of such programs include the Nitrogen Fertilizer Management Plan, and programs associated with MPCA rules 7020, 7040, and 7080 regarding municipal sludge application, feedlots and septic systems. While the Nitrogen Fertilizer Management Plan is the only program of this type designed to deal specifically with nitrogen contamination, nitrogen contamination is an issue with MPCA rules 7020, 7040, and 7080 and water well construction. These programs are generally driven more by management and design goals rather than achieving specific water quality goals.

The second major type of programs are comprehensive programs that deal with a variety of contaminant sources in an effort to meet specific water quality goals. Examples of this type of program include the Minnesota Clean Water Partnership Program, Wellhead Protection Program and Comprehensive Local Water Planning. These programs are generally focused on achieving water quality goals through implementation of necessary management practices. This second type of program often relies on several source specific programs to meet the resource goals. Protecting drinking water from NO_3 is a major goal in many watersheds, municipalities, and counties associated with these programs.

Other efforts are designed to provide information useful for prioritizing state and local programs. These efforts include geologic sensitivity classification, geologic mapping, research and ground water monitoring and modeling.

Descriptions of major ongoing programs and projects and their association with ground water nitrate are provided in this section of the report. This is not a complete list of existing programs that affect nitrogen contamination of ground water. Many other local projects and efforts are underway.

Source Specific Programs

Feedlots

See "Minnesota Feedlot Program" description in Chapter H.

Septic Systems

See "Current Policy Regarding Septic Systems and Nitrogen," Chapter I.

Municipal and Industrial Waste

See Chapter J.

Nitrogen Fertilizer Management Plan

The following is the executive summary of the Recommendations of the Nitrogen Fertilizer Task Force on the Nitrogen Fertilizer Management Plan. The entire report is included in Appendix B.

The Nitrogen Fertilizer Task Force was established by the Legislature in the 1989 Comprehensive Groundwater Protection Act to

> "...study the effects and impact on water resources from nitrogen fertilizer use so that best management practices, a fertilizer management plan and nitrogen fertilizer use regulations can be developed."

The Commissioner of Agriculture appointed a task force to make recommendations on the structure of the Nitrogen Fertilizer Management Plan based upon review of the effects of nitrogen fertilizer use on the water quality. The task force membership was established by statute to include a diverse group of representatives from agriculture, environmental groups, local government and state government.

The Nitrogen Fertilizer Task Force met ten times and held two public meetings in St. Cloud and Rochester over the period of six months. They reviewed information related to the nitrogen cycle, nitrate contamination of ground and surface water, Minnesota hydrogeologic conditions, crop production, nitrogen management, and nitrogen research. The task force also reviewed programs of other Midwestern states and received an overview of the status of existing state and federal programs.

The Nitrogen Fertilizer Management Plan, as defined in statute, must include components which (a) promote the prevention of contamination of water resources by inorganic nitrogen fertilizer, and (b) develop appropriate responses to the detection of inorganic nitrogen from fertilizer sources in ground or surface water.

The task force, after reviewing information and considering testimony, made recommendations for voluntary Best Management Practices (BMPs) which form the cornerstone of the Nitrogen Fertilizer Management Plan. In addition, it was necessary that the Management Plan complement the statutory language regarding regulatory action when voluntary BMPs are proven ineffective. The Groundwater Protection Act requires that if voluntary BMPs are proven ineffective, the MDA may promulgate rules for the establishment of Water Resource Protection Requirements (WRPRs).

The task force recommends that the Nitrogen Fertilizer Management Plan consist of three phases: (1) Promotion of BMPs, (2) Evaluation of BMP adoption and effectiveness, and (3) Response to the evaluation phase [to include non-regulatory and regulatory components]. These three phases apply at the state, regional or local level.

The task force discussed who should be involved in the implementation of the Nitrogen Fertilizer Management Plan. While the MDA is ultimately responsible for addressing the impacts of nitrogen fertilizer on water resources and has the responsibility to administer and coordinate the Nitrogen Fertilizer Management Plan, the University of Minnesota, other local, state, and federal agencies are crucial to the successful implementation of the plan. The roles and responsibilities of these groups are listed in the report.

The primary goal of the Nitrogen Fertilizer Management Plan is to prevent degradation of Minnesota's water resources by efficiently managing nitrogen inputs to maintain farm profitability. The key prevention component in this plan is the promotion and adoption of voluntary BMPs which are based upon total nitrogen management.

Minnesota's varied farming systems require a flexible format of BMPs developed to be adapted to any farming system. In addition to environmental considerations, the BMPs will have been demonstrated to be economically viable.

The Task Force recommended a three tier system of BMPs for Minnesota. The first tier is a set of state-wide BMPs that are not crop- or region-specific. The second tier consists of five sets of regional BMPs; each is tailored to one of five general regions in Minnesota. The third tier consists of BMPs for special situations that exist across the state and that present a unique set of management concerns.

This three tier system enables the BMPs to be applied to any specific situation or farm. By combining the statewide BMPs with an appropriate regional or special situation BMPs, a specific set of BMPs can be developed for any given field or situation. The specifics of the BMPs can be varied because each field history or management situation is different, yet the process for arriving at a specific set of BMPs for any situation is uniform.

The state-wide BMPs can be considered generic in that they apply to all areas of the state. The eight state-wide BMPs are listed below; a more detailed description of the BMPs can be found in the report.

1) Develop realistic yield goals.

2) Develop and utilize a comprehensive record keeping system to record field specific information.

3) Adjust nitrogen rate according to soil organic matter content, previous crop and manure applications.

4) Use a soil nitrate test when appropriate.

5) Use prudent manure management to optimize nitrogen credit.

6) Credit second year nitrogen contributions from alfalfa and manure.

7) Do not apply nitrogen above recommended rates.

8) Plan nitrogen application timing to achieve high efficiency of nitrogen use.

The second and third tiers tailor the BMPs to a region or situation. Each succeeding tier enhances or refines the previous tier and serves to match the BMPs to the prevailing climatic and soil conditions. The specific BMPs of the second and third tiers are listed in the report.

The second tier consists of regionalized Best Management Practices. The regions are based on generally climatic conditions, soil characteristics and resulting sensitivity to ground water contamination. The regional BMPs refine the prescriptions of the statewide BMPs. Five regions were identified: (1) Southeastern, (2) South Central, (3) Southwest and West-Central, (4) East-Central and Central, and (5) Northwest.

The third tier of BMPs are referred to as Special Situations BMPs. The special situations are a result of certain combinations of management and environmental conditions that may render an area or site more susceptible to ground water contamination than would be predicted by the general characteristics of the surrounding region. The third tier accounts for those management situations or sites which are interspersed throughout the state. The four situations that the task force defined as warranting special BMPs are: (1) Irrigated soils, (2) Coarse textured [non-irrigated] soils, (3) Turf, and (4) Areas near surface water.

The task force determined that the effectiveness of the BMPs needs to be evaluated on two important aspects: implementation of the practices in a voluntary system and effect on nitrate contamination of water resources. If either the implementation of BMPs or the nitrate concentrations are not being positively affected then, those factors need to be modified.

It is also recognized that even excellent and immediate implementation of BMPs may not have immediate effects on nitrate contamination of water resources due to a lag effect. The task force listed a number of methods with which evaluation of both BMP effectiveness and implementation could be accomplished at the state, regional or local level. Both of these factors need to be evaluated because of the potential time lag between implementation of BMPs and actual measurement of the impact in the water resource.

The most difficult issue for the task force to resolve was how to respond to areas where there has been significant degradation of the water resource due to nitrates. The task force reacted to this issue by proposing a mitigation and regulation framework for the MDA. This framework is based upon appropriate response to the extent of the problem and can be applied at the local, regional or state level.

In addition, a specific structure was developed to respond to local conditions where significant nitrate contamination exists and where nitrogen fertilizer practices have been implicated. This structure relies on local units of government and Soil and Water Conservation Districts (SWCDs) working in cooperation with the MDA to resolve the problem. Voluntary BMPs will be applied prior to implementing a regulatory program in this structure. Concurrent with the voluntary efforts, an evaluation will be conducted to identify the potential source(s) of the nitrate problem to adapt mitigation efforts to the source. If the voluntary BMPs are not effective, the MDA will rule development for WRPRs to be applied to the area.

Water Well Construction Code

Prior to 1974 there were no state requirements when drilling and constructing a non-municipal well. As a result, some wells were 1) constructed near pollution such as septic tanks and feedlots, 2) improperly grouted and sealed, allowing surface contaminants to readily move into the aquifer and mover through natural confining layers down into deeper aquifers, 3) constructed in low lying areas at or below grade where surface water could runoff and directly enter the well, and 4) left improperly sealed. As a result of these practices, there is/was a greater potential for NO_3, bacteria, and other contaminants to move into aquifers and to greater depths within aquifers.

During 1974, the Minnesota Department of Health enacted a Water Well Construction Code (Chapter 4725) that required: licensing well drillers, permits for new well construction, isolation distances, sealing of unused wells, specific standards for casing, grouting, sealing, completing, and capping wells, and water samples analyzed for NO_3 and bacteria be taken from newly constructed wells.

Newly constructed wells must be located on a site which has good surface drainage, at a higher ground elevation than, and at a sufficient distance from cesspools, buried sewers, septic tanks, privies, barnyards and feedlots or other possible sources of contamination. Specific isolation distances are outlined in the table M-1.

Table M-1 Isolation distances from potential nitrogen pollution sources for new wells constructed since 1974. The well code is currently under revision.

	If casing > 50 ft or 10 ft of impervious material penetrated	If casing < 50 ft and no impervious material > 10 ft penetrated
Commercial fertilizer storage area	150 ft.	150 ft.
Below grade manure storage	100 ft.	100 ft.
Cesspools, leaching pits and drywells	75 ft.	150 ft.
Septic tank	50 ft.	50 ft.
Septic system drainfield	50 ft.	100 ft.
Outhouse	50 ft.	100 ft.
Animal or poultry yard	50 ft.	100 ft.
Manure storage pile	50 ft.	100 ft.

All wells that are unused must be sealed in accordance with the well code. In 1990, 10,000 wells were reportedly sealed in Minnesota. Few studies have examined the extent or contribution of NO_3 movement into ground water from improperly or unsealed wells. The existence of unsealed abandoned wells poses a potential ground water contamination threat from many different pollutants.

The well code also requires that a water sample be submitted to a MDH certified laboratory before placing the well into service. The sample must be analyzed for NO_3 and bacteria. Currently, if the NO_3-N concentration is found to be in excess of 10 mg/l in a private well, the owner is informed of the problem and available options, but is not required to take correction action. If the well contractor violated the well code in any way, they are responsible for correcting the violation.

While the state well code probably has not resulted in a substantial nitrogen mass loading reductions, it has likely resulted in a reduction of NO_3 found in many individual wells. Because of the well code, as older wells are replaced by new wells, fewer wells in the state should have elevated NO_3 concentrations.

Special Projects

Anoka Sand Plain Demonstration Projects

The Minnesota Extension Service (MES) is actively involved in promoting, demonstrating and researching best management practices related to minimizing nitrogen in ground water. Extension projects are an integral component of the Minnesota Nitrogen Fertilizer Management Plan. MES has been working cooperatively with the Soil Conservation Service (SCS) on several projects. An example of a cooperative effort is the Anoka Sand Plains Demonstration Project. MES and SCS are working with numerous farmers in the Anoka Sand Plain area conducting best management practice demonstration projects. This is a five-year project which began in 1990.

Garvin Brook Rural Clean Water Project

In 1981, Garvin Brook Watershed in central Winona County became one of 21 Rural Clean Water Project areas in the country to evaluate the social, economic, and technical aspects of controlling nonpoint source pollution. Cost sharing for BMPs including nitrogen management and sinkhole treatment have been available through this program since 1985 for land owners in the project area. Many of the contracts for BMP work extend past 1993. A survey indicated that many farmers will continue nitrogen management BMPs after their cost share contract expires.

Comprehensive Resource Focused Programs

The Minnesota Clean Water Partnership Program

Recognizing the seriousness of nonpoint pollution and the need to establish a comprehensive program for its control, the Legislature created the Clean Water Partnership Program in 1987. The program, administered by the Minnesota

Pollution Control Agency (MPCA), provides local units of government with resources to protect and improve lakes, streams, and ground water degraded by nonpoint source pollution.

Clean Water Partnership projects begin with a desire by a local government to improve a water resource that has been polluted by land-use-related activities. Local leadership and expertise, combined with technical and financial resources from the state, create an effective program for controlling pollution and restoring water quality. Project sponsors can receive financial assistance for up to 50 percent of project costs.

Clean Water Partnership funding for local water quality projects is awarded in two phases. The first phase of a Clean Water Partnership project involves the completion of a diagnostic study. As part of the diagnostic study, local sponsors work with the MPCA to collect data and information on the water resource and its surrounding drainage area. This information is used to identify pollution problems and their causes and define water quality goals and objectives. The final step of the diagnostic study is the development of a plan that identifies the combination of education, management practices and other activities needed to restore water quality.

The second phase of the projects involves implementation of the plan recommended in the diagnostic study.

Local units of government eligible for Clean Water Partnership grants include counties, municipalities, lake improvement districts, townships and joint powers organizations established to manage a water quality project.

Status of Program

The Legislature has allocated $2.6 million to the MPCA for grants to local units of government. Thirty projects have been selected out of 90 applications through three application cycles. The 30 projects represent over $5.0 million of state and local efforts for lake, stream, ground water, wellhead protection and wetland restoration projects across the state.

Potential to Minimize Nitrogen Impacts on Ground Water

Of the 30 Clean Water Partnership projects, five projects are studying N impacts on ground water as a major component of their diagnostic study and will be developing plans to protect aquifer(s) from NO_3. These five projects include: 1) Brown/Nicollet and NE Cottonwood counties, 2) Olmsted County, 3) Coon Creek Watershed District, 4) city of Clear Lake (Sherburne County), and 5) Beardsley area. Six additional Clean Water Partnership Projects are addressing nitrogen compounds in ground water to a lesser degree. Each year, additional projects begin diagnostic studies, and potentially many more counties, cities, watersheds and other local units of government could be addressing the issue of N in ground water through the Clean Water Partnership Program.

Several different approaches to addressing this issue will be proposed in the various projects by the local governmental units. Learning experiences from existing projects will need to be communicated to other counties, cities, watersheds, and other local governmental units that are not directly involved with a Clean Water Partnership program. Through this transferring of knowledge and ideas about ways to deal with nitrogen compounds in ground water, the Clean Water Partnership shows promise for helping to minimize nitrogen impacts on Minnesota ground water.

It is recommended that funding for the Clean Water Partnership program be increased to provide additional grant money, administrative personnel and additional staff to increase the level of state technical assistance regarding nitrogen and ensure that information gained through existing projects is transferred statewide.

It is also recommended that all new CWP projects be required to develop a strategy to further promote N Best Management Practices.

Two Examples of Clean Water Partnership Projects Addressing Nitrogen:

1. Brown/Nicollet and NE Cottonwood Counties - Prior to being accepted into Clean Water Partnership, over 3,500 wells were analyzed for NO_3 and bacteria in Brown, Nicollet and NE Cottonwood counties. This project, led by Brown-Nicollet Community Health Services, has 22 contributing sponsors with a major goal of trying to understand the geologic, hydrologic, land use, and other factors affecting NO_3 concentrations in domestic and community water supplies. The project is focusing on two areas with many high NO_3 wells. Once a better understanding of the nature of the NO_3 problem is obtained, a plan for protecting the study areas and entire counties will be developed and implemented.

2. Clear Lake - The city of Clear Lake's municipal well has had NO_3 levels near 10 mg/l for several years. Through the Clean Water Partnership Program, the city is working cooperatively with state and local agencies/groups to study the aquifer system near Clear Lake, determine reasons for the high NO_3 in the city well, and design a program to protect the ground water near Clear Lake from NO_3 and other contaminants.

Wellhead Protection Program

The 1986 Amendments to the federal Safe Drinking Water Act (SWDA) and the Minnesota Groundwater Protection Act of 1989 mandate the development and implementation of wellhead protection (WHP) measures for public wells. A WHP area (WHPA) is defined as the surface and subsurface areas surrounding a public well or wellfield through which contaminants are likely to move toward and reach the well or wellfield. The fundamental goal of WHP is to prevent contaminants that may have adverse effects on human health from entering public wells. WHP is a management process that acknowledges the link between the quality of ground water supplies for drinking water and land-use activities.

The development and implementation of a WHP program in Minnesota presents a significant challenge. The task of establishing WHP measures for approximately 2,500 community and 15,000 non-community public wells is formidable, especially given the diversity of hydrogeologic conditions in Minnesota. Moreover, WHP must be integrated with the goals and objectives of other state and local water programs and then implemented within the resource limitations of those programs. The State program will only be successful to the degree to which there is local participation. Local units of government are in the best position to develop and implement site-specific WHP measures for public wells. The cities of Maple Grove, Clear Lake, Moorhead, Rochester, and St. Peter are conducting WHP projects. These projects are Clean Water Partnership Projects, with much technical assistance being provided by MDH. Nitrate in ground water is of particular concern with the Clear Lake, Rochester and St. Peter WHP projects.

The MDH is responsible for developing the State's WHP Program and for preparing the State WHP Program plan for submittal to the U.S. EPA. Under provisions of the 1986 Amendments to the SDWA, the State WHP Program must address the following seven major elements:

1) The roles of state/local agencies and public water suppliers in WHP Program development and implementation;

2) WHPA delineation for each public well that includes the surface and subsurface areas contributing water to the well;

3) An inventory of all potential human sources of contamination in each WHPA that may have an adverse effect on human health;

4) A program that contains, as appropriate, technical/financial assistance, control measures, education, training, and demonstration projects to protect the water supplies within WHPAs from such contaminants;

5) A contingency plan for the location and provision of an alternative drinking water supply for each public water system in the event of well or wellfield contamination;

6) WHP measures for all new public wells; and

7) Public participation to the maximum extent possible.

The MDH established two workgroups to provide the Department with guidance on technical and policy issues related to the development to the State WHP program. An Ad Hoc Technical Workgroup completed a report (Minnesota Department of Health, 1991) in April 1991 that addressed technical issues related to delineating, mapping and monitoring of Wellhead Protection Areas, rating contamination vulnerability of existing public wells, and prioritizing studies required to support the WHP program. The Ad Hoc Technical Workgroup identified the following research needs in their report to the Commissioner of Health: 1) continued local and regional aquifer studies to acquire baseline aquifer data and maps; 2) assessments of aquifer and well vulnerability to contamination in order to develop effective contaminant source management strategies and monitoring schemes; 3) studies to improve understanding of the vertical movement and contaminant transport characteristics of confining units; 4) development and testing of WHP area delineation methods; and 5) studies to assign and rank contaminant source risk. Not all contaminant sources within a WHP area pose the same level of risk to the public well. It is important to direct limited resources toward those sources posing the greatest risk.

An Ad Hoc WHP Policy workgroup has been meeting since September 1990 to provide MDH with guidance related to numerous policy related issues. This group is still meeting and is planning on releasing their recommendations sometime in early 1992. A number of difficult policy related issues exist pertaining to Wellhead Protection and nitrogen in ground water. Some current questions include:

1) Should differential management techniques such as focusing of resources or more stringent ground water protection measures be implemented within Wellhead Protection Areas? (i.e. should fertilizer management, septic system installation and animal, human and industrial waste management be more protective of ground water in WHP areas than other areas of the State?)

2) Can or should local governmental units have more stringent measures controlling contamination sources within Wellhead Protection Areas than state or federal law requires?

3) Currently about 21 community wells have NO_3-N exceeding the drinking water standard of 10 mg/1 and many other noncommunity public supply wells have high NO_3. What actions should be taken when public water supply wells have NO_3 above or approaching the drinking water standard? At what point do you adopt Water Resources Protection Requirements and over what area?

4) How can owners of private wells (e.g. mobile home parks) control contaminant sources beyond their property boundary?

5) Where will the state and local governmental units obtain the human and financial resources to effectively protect ground water in WHP areas? How should efforts be prioritized to make the best use of existing resources?

The decisions made regarding these and other issues will determine the potential impact that the WHP program will have regarding ground water N. Increased educational and monitoring activities will occur around WHP areas regardless of the decisions made about the above issues. Increased monitoring associated with WHP projects will likely uncover more NO_3 problem areas.

Minnesota Department of Health is required to submit a WHP program plan to EPA by September 1992. It is anticipated that WHP rules will be developed in 1992 and promulgated in 1993. Many ongoing activities are occurring to further develop the state's WHP program. MDH received an U.S. EPA grant to develop a WHP geographic information system and has contracted with the U.S. Geological Survey to construct a computer data base of aquifer properties which is needed to define WHP areas. MDH also received funding from the U.S. EPA to conduct a ground water age-dating study (using tritium) for purposes of assessing well/aquifer vulnerability. MDH is working with several pilot WHP projects funded by the MPCA's Clean Water Partnership program and will be conducting a pilot contaminant source inventory project this summer. Numerous educational and training programs have been conducted for public water suppliers, local/state governmental officials and the general public, and a WHP video was produced with funding from the STEP program.

Minnesota Department of Health will work through the Environmental Quality Board to establish coordination among state agencies at the policy level. MDH will also work with appropriate state agencies to develop memoranda of understanding. Development of such memoranda with the Minnesota Pollution Control Agency and the Minnesota Department of Agriculture are particularly important because these agencies administer most of the contaminant source control programs in the state. The coordination of WHP with local units of government will be conducted largely through the comprehensive county water planning process and through direct MDH interaction with local public water surveyors.

Minnesota Comprehensive Local Water Planning

Program Description

The Comprehensive Local Water Management Act (103B.301) was passed in 1985 with Rules (M.R. Chapter 9300) developed by the Water Resource Board and State Planning Agency in 1986. The Act was passed in response to Minnesota Counties desiring legislative authority to plan for and manage water and water related resources at the local level.

The purpose of the Act is to encourage counties to plan for the protection and management of its water and water-related resources and those land uses, both current and future, which impact water resources. Comprehensive Local Water Planning is a voluntary program for counties outside the Twin Cities seven county metropolitan area and mandatory for the metro area under the Metropolitan Surface Water Management Act (103B.201).

The Local Water Resources Protection and Management Program (LWRPMP, 103B.3369) was created in 1989 with the passage of the Ground Water Protection Act. This program established a State funding program to assist counties, with implementation of State approved Comprehensive Local Water Management Plans (Plan), through a combination of non-competitive base grants and competitive challenge grants. It is currently funded at $2.4 million annually.

In order to receive State approval, a county must prepare a Plan that:

1. Includes an inventory of existing and available natural resources data, a description of existing management programs and their effectiveness, and related background information. This portion of a Plan is based on 55 required data elements, which includes ground water and surface water quality. The data and summaries provide the raw facts about particular problems or issues.

2. Provides detailed assessments of the resource data and their implications for present and future use. Assessments are meant to be an analysis of the data and help lead a county to possible solutions. Eighteen separate assessments and/or implications are required. Implications of ground water and surface water quality and quantity are included.

3. Identifies significant resource issues and prioritizes them based on local needs, attitudes and perceptions. An issue has been defined as a problem or opportunity that has a significant influence on the way a county functions or on its ability to achieve a desired future, and for which their is no agreed-upon response.

4. Includes goals and objectives. Goals and objectives form the logical link between issues and actions.

5. A set of specific actions that the county will take to address its high priority issues.

6. An implementation plan which states how and when the plan will be carried out. The implementation plan must also describe the manner in which the Plan will be amended and how the county will resolve conflicts.

Status of Program

Currently 78 of 80 greater Minnesota counties are involved in Local Water Planning. By January 1, 1992, it is estimated that 65 counties will have approved Plans, the remaining 13 counties are in the planning process and estimate that they will have Plans completed in 1992.

Local Response to Ground Water Nitrogen

Although Comprehensive Local Water Planning began in 1985, it was not until 1988 that counties began the planning process, with the first Plans to be approved and implemented occurring in 1990 and 1991. Overwhelmingly counties have identified the purity of drinking water as their number one priority.

The majority of counties have emphasized through their Plans, that a lack of reliable data is available to determine the ambient condition of ground water quality. Counties have recognized the complex relationships between ground water quality and other factors such as land use, geology, and ground water quantity. Counties have also recognized that the presence of N in ground and surface water may be an indicator that other pollutants (such as pesticides) are impacting the resource.

As a result, the general response by counties to ground water contamination, including nitrogen, has been the formulation of action plans which incorporate the following general components:

1. Development of ambient ground water monitoring programs, utilizing certified laboratories, to establish long-term reliable data from which future assessments and program and political decisions will be based.

2. Development of local water well screening programs that are coupled with education and outreach programs. Education of residents is a high priority as it builds a local base of support for future decisions and programs.

3. Through local SWCD and Minnesota Extension Offices, encourage and promote the use of N, manure and pesticide BMP's.

4. Local efforts to inventory and map potential contaminant sources and rank existing or potential threats. Sources include feedlots, abandoned wells, storage tanks, dumps and landfills, and individual sewage treatment systems.

5. Local efforts to adopt and enforce programs such as the MPCA feedlot and on-site sewage treatment programs, and the MDH well code.

Following are some specific examples of implementation activities that have been identified in county Plans that address nitrogen contamination and nitrogen management.

Olmsted County - Selected Implementation Initiatives

** Develop, adopt, and implement a county feedlot ordinance.
** Institute a surface and ground water monitoring system for fertilizer components in the county.
** Pass a county ordinance to ban fall application of chemical nitrogen fertilizer on highly susceptible areas.
** Recommend soil testing every three years on cropland prior to application of fertilizer in order to eliminate over-application.

Stearns County - Selected Implementation Initiatives

** Work with MPCA to characterize NO_3 contamination problems within the urban towns adjacent to St. Cloud.
** Investigate the possibility of establishing a cancer and birth defect registry for Stearns County.
** Seek the delegation of the well code from MDH.
** Amend the Sewer Ordinance and the Subdivision Ordinance to treat differentially those portions of the county that contain sensitive soils.
** Implement a water quality education program in the county.

Redwood County - Selected Implementation Initiatives

** Develop a list of priority abandoned wells in the county and investigate administration of the well code at the county level.
** Bring individual sewage systems into compliance with an amended county septic system ordinance.
** Increase the annual number of private water supplies tested, expand testing capabilities of local water testing laboratory, and in cooperation with state agencies develop a local and regional ground water data base.
** Determine recharge and geologically sensitive areas that are in need of special attention.
** Provide public information on local water quality issues.

Effectiveness of Programs

With less than one year of implementation activity in many counties it is not possible at this point in time to evaluate the actual effectiveness of county plans, programs, or actions on the resource itself.

An alternative measure of effectiveness would be to assess the ability of counties to respond to nitrogen contamination, and adequacy of local controls.

Comprehensive Local Water Planning has been very successful in developing local infrastructures and an informed public, which the BWSR feels will be instrumental to the enhancement and acceleration of local efforts to manage and protect water and related land resources at the county level. Water planning has also strengthened and created new partnerships. These new local-state partnerships will become increasingly important to the state in achieving its goals.

Adequacy of Local Authorities to Respond

Although counties and incorporated areas in Minnesota do have existing authorities to address nitrogen in ground water, through their land use planning and zoning and general health and welfare regulations, it is unclear that they would use these authorities to regulate nitrogen. This is due in part to the complex and very technical nature of defining boundaries or spatial limits, developing remedial action plans, regulating land use, and regulating the sale and use of inorganic nitrogen, and a general unwillingness to adopt regulatory programs at this time.

It should be emphasized that an adequate response to nitrogen in ground water must also include education, information, monitoring, and data collection as well as regulation. To this end counties are beginning to respond to the education, information, monitoring and data collection needs of nitrogen in ground water with implementation of their comprehensive local water management plans.

Counties are also in a position to provide direct technical assistance to landowners in the implementation of nitrogen best management practices through local SWCD and Minnesota Extension Offices.

Recommendations from Local Water Plans are stated at the end of this chapter.

Special Projects

MSEA Project

The U.S. Department of Agriculture (ARS), University of Minnesota, U.S. Geological Survey, Minnesota Pollution Control Agency and U.S. Environmental Protection Agency are working to further develop and refine BMPs, assess ground water impacts from agricultural practices, and understand the mechanisms of ground water contamination in the Anoka Sand Plain area. This project is a Management Systems Evaluation Area (MSEA) project which is part of the President's Initiative for Water Quality. The MSEA is a five year project which began in 1990.

Minnesota River Assessment Project

Beginning in 1989, a four year diagnostic study to assess the water quality and factors affecting the quality of the Minnesota River was started. This multi-agency effort is generating an abundance of information that will aid in developing a plan to implement best management practices throughout the Minnesota River Basin. Since high nitrogen levels are entering the Minnesota River via shallow ground water often drained by tile lines in subwatersheds of the Minnesota River Watershed, nitrogen management BMPs will be a component of the implementation plan. The diagnostic study will be completed in 1993.

Support Programs/Efforts

Ground Water Sensitivity Classification

The 1989 Minnesota Ground Water Protection Act directed the Minnesota Department of Natural Resources in consultation with the Minnesota Geological Survey, Soil and Water Conservation Districts, Local Water Planning Authorities, and other interested parties to develop criteria for identifying sensitive ground water areas and adopt the criteria by rule (Section 3, 103H.101).

A multi-agency workgroup developed statewide criteria and guidelines for assessing geologic sensitivity of ground water during the 1989-91 biennium as part of a Legislative Commission on Minnesota Resources funded project. In the criteria and guidelines report, geologic sensitivity is defined as being proportional to the time required for a contaminant to move vertically from the ground surface to an aquifer. The sensitivity criteria can be applied using one to three methods. A level 1 assessment is a preliminary evaluation of surficial geologic sensitivity using available information. A level 2 assessment is a more detailed evaluation of the geologic sensitivity of surficial materials. Deeper, confined aquifers can be rated using the level 3 method. Actual sensitivity mapping of counties was not performed. Rather these guidelines and criteria will support the development of rules required by the Ground Water Protection Bill.

In accordance with the Ground Water Protection Bill, state agencies must consider the risk identified by sensitivity rules in an area to prevent and minimize ground water degradation in sensitive areas when adopting best management practices, water resource protection plans, and water resource protection requirements. State agencies must consider sensitivity when undertaking activities in order to prevent and minimize ground water degradation.

These sensitivity rankings, once further developed, will affect management of nitrogen pollution sources and perhaps prioritization of state and local efforts. Less sensitive areas may generally have a greater potential to lose nitrogen through denitrification in the soil zone compared to very sensitive areas. Many questions still remain, however, regarding potential long-term nitrogen impacts in "less" geologically sensitive areas. In areas deemed less sensitive, there could be a long-term build-up of nitrogen in the soil zone that could slowly move down to ground water. Without denitrification, much of the soil NO_3 below the rooting zone will eventually reach ground water. Since we do not yet fully understand the amount of denitrification occurring in the vadose zone under various conditions, land management should still be directed at preventing soil nitrogen accumulation even in areas deemed less sensitive.

Monitoring

Many nitrate monitoring efforts in the state were described in Chapter B. Monitoring efforts producing reliable data can provide information useful to state and local government for prioritizing efforts. The degree of utility depends on many factors, including monitoring design.

Research

Numerous research activities related to nitrogen and water quality are underway. Research is an important component of long term water quality protection.

Hydrogeologic Mapping and Modeling

The U.S. Geological Survey, Minnesota Geological Survey and University of Minnesota are currently involved in numerous hydrogeologic mapping, monitoring, and modeling projects throughout the state that will assist state and local water planners in programs designed to control NO_3 contamination. There is a great need for these efforts to be continued.

SUMMARY

Options for **communities** with unacceptable nitrate levels include drilling a new well, blend high and low nitrate water, install a treatment system, or connect to a rural water system. The latter two options are often cost prohibitive and drilling a new well is not always an option. The preferable long term solution is pollution prevention. Implementation of wellhead protection is advised.

Nitrate testing of public and domestic water supplies is necessary to promote public health protection. **Homeowners** with high nitrate may have the following options: drilling a new well, installing treatment systems that remove nitrate, buy bottled water or continue drinking high nitrate water. There are disadvantages with each of these options. The option most recommended by MDH for long term solution is drilling a new well.

In response to a growing national concern about the ecological and health impacts of nitrogenous compounds, the **U.S. Environmental Protection Agency** is developing a nitrogen action plan. The nitrogen action plan workgroup has drafted recommendations that are organized into five categories, including 1) develop state nutrient management programs, 2) improve on-farm nitrogen management to protect water quality, 3) improve public and private drinking water quality, 4) increase control of point sources through current regulatory authority, and 5) research areas of uncertainty. The draft federal nitrogen action plan would be implemented in two phases. Phase I emphasizes using current regulatory authorities, pollution prevention techniques, and research. Activities under Phase II would begin if voluntary efforts and current legal authorities were insufficient.

Minnesota has a number of existing and developing programs that are minimizing or have the potential to minimize ground water nitrogen contamination. The only statewide program that specifically targets **nitrogen** pollution prevention is the Minnesota Nitrogen Fertilizer Management Plan. Several programs exist that each deal with a variety of contaminants from specific sources, such as feedlots, septic systems and municipal and industrial waste. Other programs deal with multiple pollution sources, including the Minnesota Clean Water Partnership Program, Wellhead Protection Program, and Comprehensive Local Water Planning. Nitrogen contamination of ground water is an issue with each of these programs. Several other regional and local efforts are underway. These existing programs show promise for minimizing nitrogen movement to ground water. However, many of these programs are in the developing stages and their effect on ground water nitrogen levels will not be known for many years. Many of the State programs need additional resources to more effectively deal with nitrogen.

RECOMMENDATIONS

General

Due to the many developing programs, increased monitoring, ongoing research and technological advances, it is recommended that the **Nitrogen in Ground Water report and follow-up efforts be re-evaluated in three to five years.**

Numerous recommendations have been made throughout this report, that when implemented, will further prevent nitrogen movement to ground water. Additional actions are also needed to assist existing programs in best directing and focusing their efforts.

Minnesota currently has a goal of nondegradation where practicable. Questions remain about how this goal applies to nitrate in ground water and what the nitrate reduction goals are for areas already experiencing nitrate problems. **Minnesota needs to develop clear-cut and reasonable nitrate reduction and prevention goals.** Goals will serve as a rallying point and yardstick to measure the adequacy of protection efforts. Minnesotans need to have a clearer idea of what the state is aiming for in our protection efforts as it relates to nitrate and to let the state know that high nitrate in ground water is not something that we want to learn to live with.

Some areas of the state currently have severe NO_3 problems in their ground water and focused response to these areas is needed. Intensive best management practice promotion, ground water testing programs, prioritization for entering programs such as Clean Water Partnership and investigation of causes should be focused in geologically sensitive and/or severely impacted areas. Guidelines and criteria have been developed so that a consistent approach can be taken statewide to determine which areas are to be classified as geologically sensitive areas. Criteria should also be established for determining the degree of NO_3 impact for areas of given size (e.g. townships, nitrogen management districts) throughout the state.

Classifying townships or other such areas on the basis of NO_3 levels would:

1) Help county water planners better decide what priority nitrogen management should have and where efforts should be focused;

2) Assist MDA in planning for and designating Special BMP Promotion areas (as discussed in the Nitrogen Fertilizer Management Plan);

3) Help MPCA and MDH to prioritize where Clean Water Partnership projects and Wellhead Protection Projects should be implemented; and

4) Heighten land owner awareness of the urgency for implementing Best Management Practices in impacted areas.

It is therefore recommended that a **multi-agency workgroup be established to determine criteria for classifying the severity of existing NO_3 problems.** Existing data, where sufficiently available, and newly acquired data (see monitoring needs section in Chapter B) could then be used to classify areas according to NO_3 problems in ground water. The criteria will have to consider the number and density of wells, depth or aquifer of analyzed wells, methods of analysis, frequency of sampling, and several other factors.

With the realization that it may take many years before nitrate levels are reduced to safe levels, there needs to be a focused approach for protecting those most vulnerable to problems associated with drinking high nitrate water, primarily infants. **Each family expecting a baby should have knowledge of the nitrate levels in their drinking water supply. The cost and trouble to have domestic well water analyzed should be kept to a minimum.** If high nitrate is found, the family could find an alternative source of water for the first six months of their child's life.

Another issue needing attention is the coordination of existing programs. It is unclear in the nitrogen fertilizer task force report how programs such as Comprehensive Local Water Planning, Clean Water Partnership and Wellhead Protection will fit into the state's Nitrogen Fertilizer Management Plan. Options for clarifying this could include **developing an institutional framework that clearly lays out the roles of various programs for controlling nitrogen** or developing memorandums of understanding between MDH, BOWSR, MDA, MPCA and DNR.

Recommendations from Local Water Plans

One component of local water plans was the opportunity to express to the State of Minnesota through the BWSR, recommended changes to state programs. An analysis of these comments revealed that recommendations from counties generally fell into three broad categories made up of coordination, financing, and technical assistance.

Coordination

- The State needs to do a better job of working with and through local government as it delivers programs and services. Counties indicate that poor communication is the biggest impediment.

- The state needs to do a better job of coordinating the various natural resource programs and activities that it delivers to and through county and local government.

Financing

- Recommendations from counties suggest that the State needs to ensure dependable long-term funding of grant programs such as Clean Water Partnership and the Local Water Resources Protection and Management Program.

Technical Assistance

- The State needs to have available and provide specialized technical assistance to counties through the various state agencies to assist them in project and program implementation.

- The State needs to focus more of its staff resources at the regional level.

- The State needs to expand the capability of the Minnesota Geological Survey to do County Geologic Atlases.

- The State needs to expand its role and efforts in the area of sustainable agriculture.

- In order for counties to adequately respond to nitrogen contamination areas and problems, the State will need to develop practical working models and tools for local government use. These models and tools are needed for the assessment of resource conditions, defining problems or sensitive areas, development of remedial action plans, and ordinance and regulation development.

- In order for SWCD's to provide an adequate level of direct technical assistance to landowners for the development of water quality plans and the implementation of nitrogen best management practices, additional financial resources are necessary.

REFERENCES

Consumer Reports. 1987. Water, water everywhere. 52:1:42-48.

EPA. 1991. Nitrogen Action Plan (DRAFT) March 1991. 161 pp.

Freshwater Foundation. 1989. Home water treatment: what is the use of point-of-use? Health and Environment Digest. 3:6:1989.

Hubert, Cynthia. 1991. D.M.'s water costs rise along with nitrate levels. The Des Moines Federal Register, Tuesday April 12, 1991.

Minnesota Department of Health. 1991. Wellhead protection in Minnesota - a report of recommendations by the ad hoc wellhead protection technical workgroup. April 1991. 48 pp.

Appendix A

Locations and Characteristics of Principal Water Supply Aquifers in Minnesota

APPENDIX A

(Excerpted from Adolphson, et al., 1981)

PRINCIPAL WATER-SUPPLY AQUIFERS IN MINNESOTA

Fourteen aquifers, ranging from Quaternary to Precambrian in age, are the major source of water to wells in Minnesota. These aquifers supply half the municipal population and nearly all the rural population with water. The aquifers occur in two broad geologic categories: (1) glacial deposits, and (2) bedrock (table 1).

Most glacial aquifers consist of sand and gravel deposits called outwash, which is the material washed out of glaciers by melt waters. Outwash occurs both as surficial deposits and, because of repeated glaciations in the State, as buried deposits. Other surficial aquifers also occur as alluvial, valley-fill, ice-contact, and beach-ridge deposits (fig. 1). The buried deposits, which underlie one or more layers of till, are just as important as the surficial aquifers because in many areas they are the only source of water.

The bedrock aquifers are sedimentary formations and crystalline rocks. The sedimentary formations consist of sandstone, dolomite, and limestone that were laid down in seas that covered Minnesota before the glacial period (figs. 2 and 3). The crystalline bedrock makes up the basement complex in the State (fig. 4). Although only small yields of water are available to wells completed in the crystalline rocks, they are important locally where no other source of water is available. Most of the water in the State's aquifers is fresh (defined as having dissolved-solids concentrations of less than 1,000 mg/l). Dissolved solids generally increase from east to west in the State.

Ground water is commonly classified by chemical type on the basis of relative concentrations in milliequivalents of principal cations and anions. This classification provides the basis for grouping waters of similar types and for evaluating chemical mechanisms that affect water quality. Different water-quality types may exist in proximity, varying both vertically and laterally among different aquifers or even within the same aquifer. Six principal water-quality types are present in the aquifers of the State. Ground water in Minnesota is predominately of the calcium magnesium bicarbonate type. This water type generally occurs in recharge areas and, most often, in the upper part of the ground-water system.

The aquifer descriptions given below are assigned numbers from 1 to 14 in the order from the youngest to oldest.

1. **Surficial sand and gravel aquifers:** These aquifers cover about one-third of the State and are comprised of alluvial outwash, beach-ridge, valley-train, and ice-contact deposits (fig. 1). Extensive outwash deposits are a significant source of ground water in central Minnesota. The aquifers are unconfined, and well yields range from 10 to 3,000 gal/min on a short-term basis. Sustained yields are as much as 1,000 gal/min in places. The thickness of the deposits, which consist of fine to coarse sand and gravel, is generally less than 100 feet, but may reach several hundred feet in places. Although water supplies have been only slightly to moderately developed from surficial aquifers in most of the State, there is

a possibility of overdevelopment in heavily irrigated areas. Dissolved-solids concentrations are generally less than 500 mg/l; maximum concentrations are about 1,000 mg/l. Hardness ranges from 200 to 400 mg/l, and, locally, nitrates are as much as 30 mg/l. Calcium magnesium bicarbonate is the dominant water type.

2. **Buried sand and gravel aquifers:** These aquifers can occur in nearly all areas of the State except where the drift is thin or absent much as in the northeast and southeast. The aquifers consist of discontinuous lenses of fine to coarse sand and gravel that are isolated from one another by till. Most lenses are less than 10 feet thick, but they may be as much as 150 feet thick locally. Where present, the lenses occur at depths ranging from a few feet below land surface to the base of the drift. These aquifers are generally confined, and well yields range from about 10 to 1,000 gal/min. Buried aquifers are the major source of water for municipal and farm wells in the central and southwest parts of the state, but are only slightly developed in other areas. The aquifers may have good potential for development in areas where the sand and gravel fill valleys in the bedrock surface. Dissolved-solids concentrations are generally less than 1,000 mg/l; maximum concentrations are about 2,000 mg/l. The hardness of the water ranges from 300 to 1,200 mg/l. Iron and manganese concentrations are commonly troublesome. The dominant water type is calcium magnesium bicarbonate, but in the southwest and northwest where the buried aquifers are underlain by Cretaceous rocks, calcium magnesium bicarbonate sulfate and calcium magnesium chloride water types are present.

3. **Cretaceous aquifer:** These rocks generally consist of gray, soft, argillaceous shale that contains sand beds. The deposits are nearly continuous in the western half of the state, thin or discontinuous in the central and southeast, and absent in the northeast (fig. 2). The aquifer is not widely used except where drift aquifers are absent or where well yields are poor. The Cretaceous aquifer is a major source of water locally southwest of the Minnesota River. Most water use if for farm supplies and pumping rates usually do not exceed 10 gal/min. Ground water in the aquifer is confined and wells yield as much as 25 gal/min where the sedimentary rocks are relatively thick and the sand is more than 10 feet thick. However, the potential for development of large municipal and industrial water supplies is poor.

   Five water types occur in the Cretaceous aquifer. Sodium bicarbonate type water occurs at depths in the northwest. Calcium infiltrates into the Cretaceous rocks from overlying drift aquifers and is removed by cation exchange for sodium. Dissolved-solids concentrations are generally between 500 and 1,500 mg/l and harness ranges from 25 to 200 mg/l. Sodium chloride type water is common in the extreme west. Cretaceous and Paleozoic aquifers in the Williston structural basin of North and South Dakota are the major sources of sodium chloride type water in Minnesota. Dissolved-solids concentrations range from 2,000 to 4,000 mg/l. Sodium chloride type water is also common in the Cretaceous sedimentary rocks southwest of the Minnesota River. Locally, chloride concentrations are as much as 2,000 mg/l. The water is soft, ranging from less than 20 to 120 mg/l. Sodium sulfate type water also occurs southwest of the Minnesota River. Sulfate type water in the drift may result from mixing with the sodium chloride type water or from calcium magnesium bicarbonate sulfate type water in the Cretaceous aquifer undergoing cation exchange in clayey deposits. Dissolved-solids concentrations are as much as 6,000 mg/l. Calcium magnesium bicarbonate type water occurs northeast of the Minnesota River.

4. **Cedar Valley-Maquoketa-Dubuque-Galena aquifer:** the aquifer (hereafter called upper carbonate aquifer) is composed mainly of limestone, dolomite, and dolomitic limestone and is the youngest of a series of sedimentary Paleozoic formations deposited in the Hollandale embayment of southeastern Minnesota (fig. 3). This aquifer, which extends about 80 miles northward into Minnesota from the Iowa border, and the underlying aquifers make this area the most favorable part of the state for developing large water supplies. Wells in the upper carbonate aquifer are completed in solution channels, joints, and fissures. Yields range from 200 to 500 gal/min and are highly variable. The highest yields are obtained where wells penetrate the entire carbonate section. Water supplies have been slightly to moderately developed and there is excellent potential for additional development. However, the aquifer is extensively contaminated from agricultural wastes and other nonpoint sources. Dissolved-solids concentrations range from 200 to 650 mg/l and hardness from 200 to 400 mg/l. Locally, concentrations of iron are greater than 1 mg/l. The water is a calcium magnesium bicarbonate type.

5. **Red River-Winnipeg aquifer:** This aquifer, which underlies several hundred feet of till and lake sediments of Glacial Lake Agassiz in the northwest corner of the state, is composed mainly of sandstone, limestone, and shale of Paleozoic age (fig. 3). The rocks extend westward into the Williston structural basin. Water is under confined conditions throughout most of the aquifer. Flows of 60 gal/min from artesian wells have been recorded, and yields to pumping wells have been recorded, and yields to pumping wells may exceed several hundred gallons per minute from wells that penetrate the entire section. The aquifer, which has a great potential for large supplies, is seldom used because the water generally is not suitable for drinking. Water from the aquifer is highly mineralized. Dissolved-solids concentrations range from 5,000 to 60,000 mg/l. The water is a sodium chloride type.

6. **St. Peter aquifer:** The aquifer, a white, fine- to medium-grained sandstone, is part of the Paleozoic system in the Hollandale embayment of southeastern Minnesota (fig. 3), and extends as far north as the Twin Cities basin. Water occurs under both confined and unconfined conditions. Because of greater water supplies in underlying aquifers and the discontinuous area extent of the aquifer in the Twin Cities area, the St. Peter is generally not used for public supplies. Yields generally range from 10 to 100 gal/min and yields of 1,000 gal/min have been reported locally. Dissolved-solids concentrations range from 100 to 600 mg/l and hardness from 200 to 400 mg/l. Calcium magnesium bicarbonate type water generally occurs in the aquifer.

7. **Prairie du Chien-Jordan aquifer:** The aquifer is composed mainly of dolomite and sandstone and is the major aquifer in southeastern Minnesota (fig. 3). Karstic conditions at the surface are common in the extreme

southeast where the drift is thin. Water supplies from the aquifer have been slightly to moderately developed in the southeast and highly developed in the Twin Cities where is provides about 75 percent of the annual ground-water supply. Wells yield as much as 2,700 gal/min in the Twin Cities basin. In the southeast, well yields generally range from 300 to 600 gal/min. Dissolved-solids concentrations of 200 to 600 mg/l, and hardness of 200 to 400 mg/l, are similar to other aquifers in the Hollandale embayment. The water type is calcium magnesium bicarbonate (fig. 9). Locally, water from the aquifer has nitrate concentrations as much as 20 mg/l and iron and manganese concentrations greater than 1 mg/l.

8. **Franconia-Ironton-Galesville aquifer:** This aquifer, which consists of very fine to coarse sandstone interbedded with shale, dolomitic sandstone, and dolomitic siltstone, is the fourth in the series of bedrock aquifers in the Hollandale embayment of the southeast. The Franconia is not a significant source of water regionally, but the Ironton-Galesville (sandstone) may be an important source outside the boundary of the Prairie du Chien-Jordan aquifer (fig. 3). Yields range from 40 to 400 gal/min. Dissolved-solids concentrations (200 to 650 mg/l) and water type (calcium magnesium bicarbonate) are similar to other aquifers in the Hollandale embayment (fig. 9). Locally, iron and manganese concentrations are greater than 1 mg/l.

9. **Mount Simon-Hinckley-Fond du Lac aquifer:** The aquifer comprises a thick sequence of sandstone, siltstone, and shale that underlies all the southeast part of Minnesota as far north as Duluth (fig. 3). It is an important aquifer in the Hollandale embayment and in the Twin Cities metropolitan area where it supplies about 15 percent of the ground water used. Withdrawals increase significantly north of the Twin Cities. Yields to wells are generally about 500 gal/min but, locally, yields may be as much as 2,000 gal/min. A long-term cone of depression has developed in the Twin Cities metropolitan area. North of the Twin Cities, the aquifer will support additional moderate development. Dissolved-solids concentrations are slightly lower than in water from other aquifers in the Hollandale embayment, ranging from about 100 mg/l in the north to as much as 2,400 mg/l in the south. Dominant water type is calcium magnesium bicarbonate sodium chloride type water occurs at depth in the southeast. The aquifer locally has concentrations of iron and manganese greater than 1 mg/l.

10. **North Shore Volcanic aquifer:** This aquifer, the major bedrock aquifer along the north shore of Lake Superior, is a series of basaltic lava flows and interbedded sedimentary rocks (fig. 4). Water is generally obtained from the upper 300 to 400 feet where feet where fractures and weathering are extensive. Yields to wells are generally less than 25 gal/min, but locally are as much as 100 gal/min. Many wells near the Lake Superior shore flow. The aquifer is moderately developed for rural and public supply. Quality of the water is highly variable ranging from good to highly mineralized. Dissolved solids range from 100 to 50,000 mg/l but most are less than 1,300 mg/l. Hardness is as much as 28,000 mg/l but most hardness is less than 400 mg/l. The water is calcium magnesium bicarbonate sulfate type, but sodium chloride type water occurs locally.

11. **Sioux Quartzite aquifer:** This aquifer underlies most of southwest Minnesota (fig. 4). Locally, it is an important aquifer, furnishing water to seven municipal and to numerous domestic and stock wells. It is a fairly reliable source of water when the wells are completed in fractured and weathered zones near land surface and/or buried zones of porous and poorly cemented sandstone that are interbedded within the well-cemented quartzite. Yields range from 1 to 450 gal/min and, for the municipal wells, average about 100 gal/min. The best quality water is where the aquifer underlies thin drift. Dissolved-solids concentrations are generally less than 900 mg/l and total hardness is less than 400 mg/l. The water is calcium magnesium bicarbonate sulfate type.

12. **Proterozoic metasedimentary aquifer:** This aquifer consists of thinly bedded gray to black argillite that underlies drift and rocks of Cretaceous age in much of the north-central part of the state (fig. 4). Yields to wells are generally less than 20 gal/min, but yields of 30 gal/min can be obtained locally from wells completed in the fractured zones near the upper surface of the aquifer. The water is utilized for numerous domestic and some municipal supplies. The water is of the calcium magnesium bicarbonate type and contains much less iron, manganese, dissolved solids, and hardness than most water from Biwabik Iron-formation and drift aquifers in the area.

13. **Biwabik Iron-formation aquifer:** The aquifer, which is composed of ferruginous chart, underlies drift and crops out in north-central Minnesota (fig. 4). It yields little water to wells where the rocks have not been altered by faulting or leaching. In the altered zones associated with joints, faults, and solution channels, individual yields range from 250 to 750 gal/min. Yields to wells are as much as 1,000 gal/min in highly fractured zones. The aquifer yields water to many municipal and industrial wells and is the most productive source of ground water in the Mesabi Iron Range. The water meets drinking-water standards for all chemical constituents, although hardness ranges from moderate to very hard and the water locally contains much iron, manganese, and silica. The dissolved-solids concentrations range from about 100 to 300 mg/l. The water type is calcium magnesium bicarbonate.

14. **Precambrian aquifer:** The aquifer, consisting of igneous and metamorphic rocks such as granite, greenstone, and slate, underlies the state (fig. 4). Although these rocks are a source of small water supplies in the southwest, central, and northeast parts of Minnesota, they are not usually considered an aquifer in the rest of the state. Yields are dependent on the occurrence of fractures, faults, and weathered zones, and generally increase where the bedrock is overlain by thick drift. Some wells are drilled several hundred feet into the rocks, so that the drilled hole serves as a reservoir. Yields generally range from 1 to 25 gal/min; locally, yields are as much as 150 gal/min. Water is often similar in quality to that of overlying drift. Dissolved-solids concentrations are generally less than 300 mg/l. Calcium magnesium bicarbonate type water is the most common in the aquifer.

Other geologic formations in the state provide an adequate supply of water to localized areas. The Decorah, Platteville and Glenwood Formations are interbedded shales and limestones that lie stratigraphically between the

St. Peter and the Galena formations in southeastern Minnesota. The St. Lawrence Formation also consists of interbedded shales and limestones. This formation, which is considered a confining layer between the Jordan and Franconia aquifers, supplies water for some residences in southeastern Minnesota.

REFERENCE

Adolphson, D.G., J.F. Ruhl and R.J. Wolf. 1981. Designation of principal water-supply aquifers in Minnesota. U.S. Geological Survey Water Resources Investigations 81-51. 19 pp.

Table 1 Generalized stratigraphic and hydrogeologic units of principal aquifers in Minnesota. From Adolphson et al., 1981.

Erathem and Eonothem	System	Stratigraphic unit	Hydrogeologic unit
Cenozoic	Quaternary	Drift	Surficial sand and gravel
			Buried sand and gravel
Mesozoic	Cretaceous	Cretaceous rocks, undifferentiated	Cretaceous
Paleozoic	Devonian	Cedar Valley Limestone	Upper carbonate
	Ordovician	Maquoketa Shale Dubuque Formation Galena Dolomite	
		Red River Formation Winnipeg Formation	Red River-Winnipeg
		St. Peter Sandstone	St. Peter
		Prairie du Chien Group	Prairie du Chien-Jordan
	Cambrian	Jordan Sandstone	
		Franconia Formation Ironton Sandstone Galesville Sandstone	Franconia-Ironton-Galesville
		Mount Simon Sandstone	Mount Simon-Hinckley-Fond du Lac
Precambrian: Proterozoic		Hinckley Sandstone Fond du Lac Formation	
		North Shore Volcanic Group	North Shore Volcanic
		Sioux Quartzite	Sioux Quartzite
		Proterozoic metasedimentary rocks	Proterozoic metasedimentary
		Biwabik Iron-formation	Biwabik Iron-formation
Precambrian: Proterozoic and older		Precambrian rocks, undifferentiated	Precambrian rocks, undifferentiated

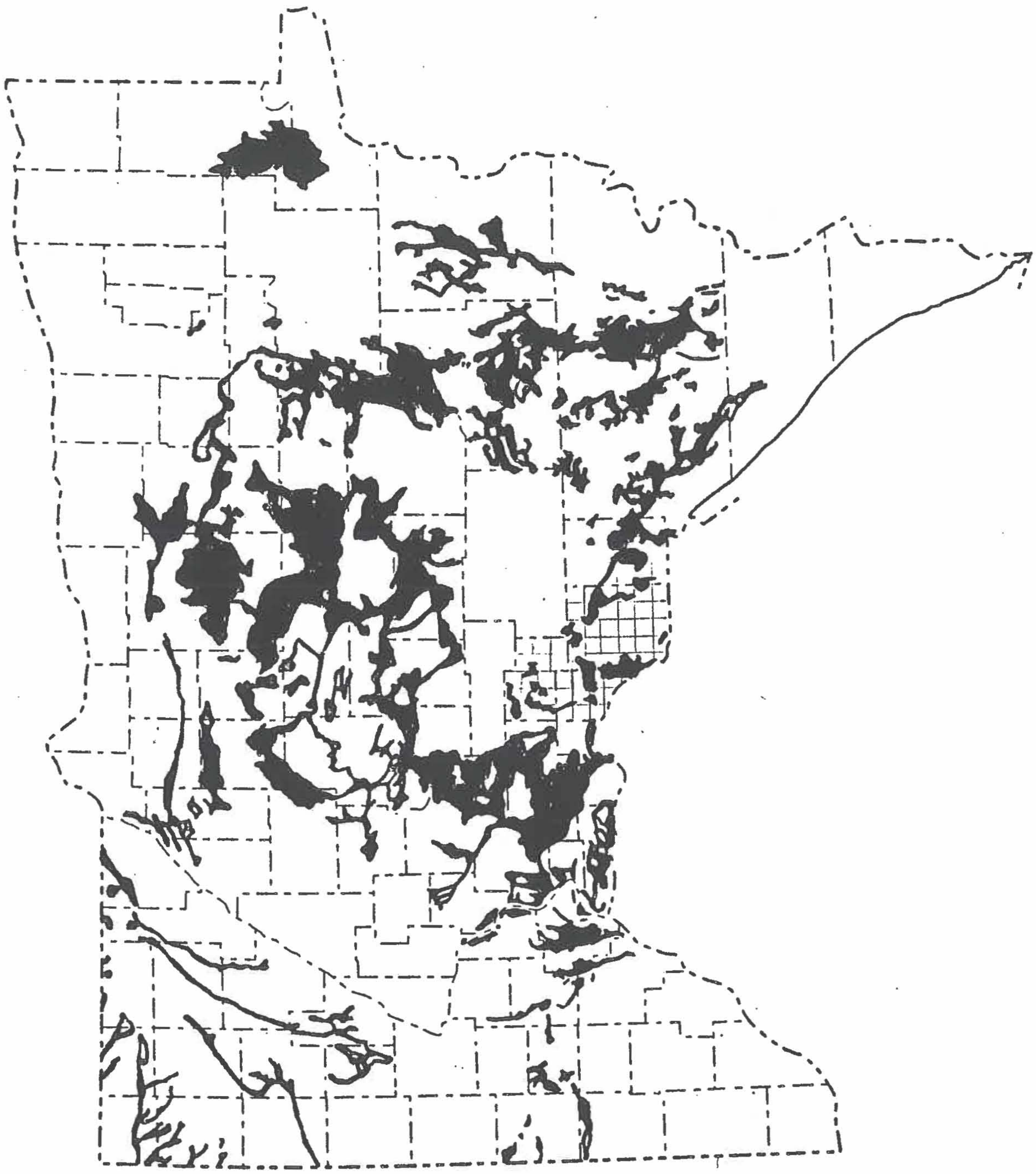

Fig. 1 Location of most major surficial sand aquifers throughout Minnesota (geologic interpretations by H.W. Anderson of the U.S. Geological Survey).

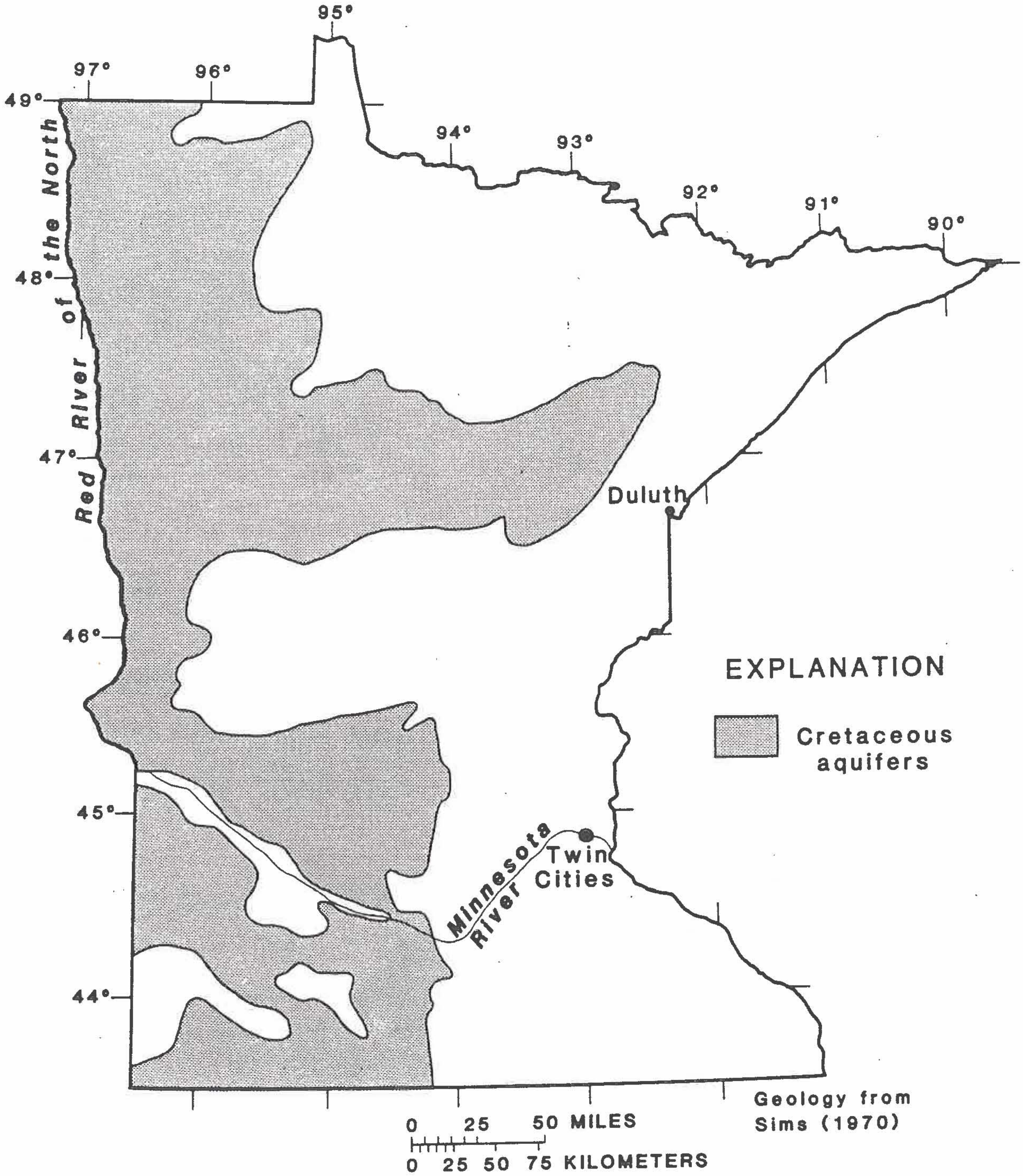

Figure 2.--Extent of Cretaceous aquifers

From Adolphson et al. 1981.

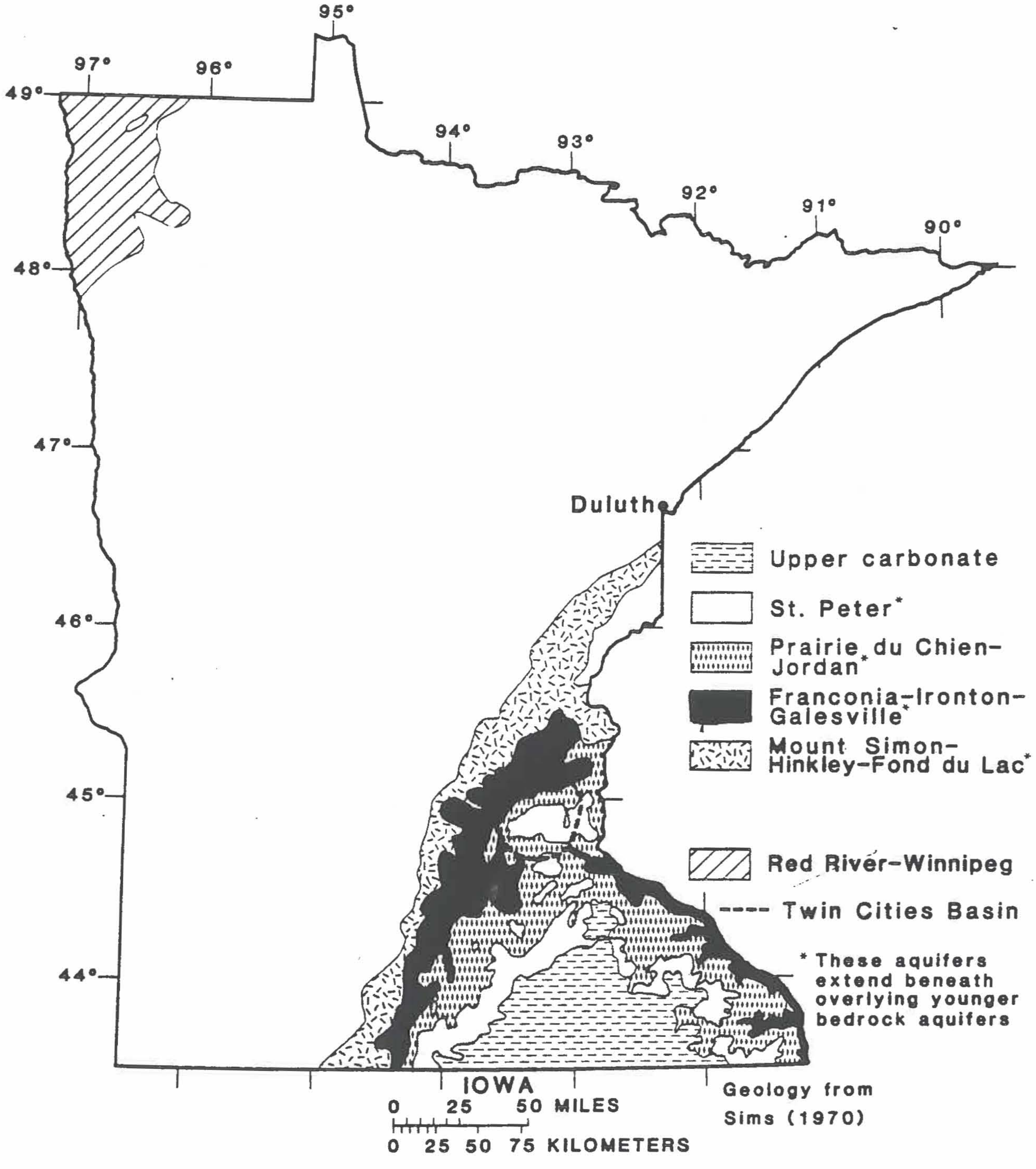

Figure 3 Extent of Paleozoic and late Precambrian aquifers in Minnesota. From Adolphson et al., 1981.

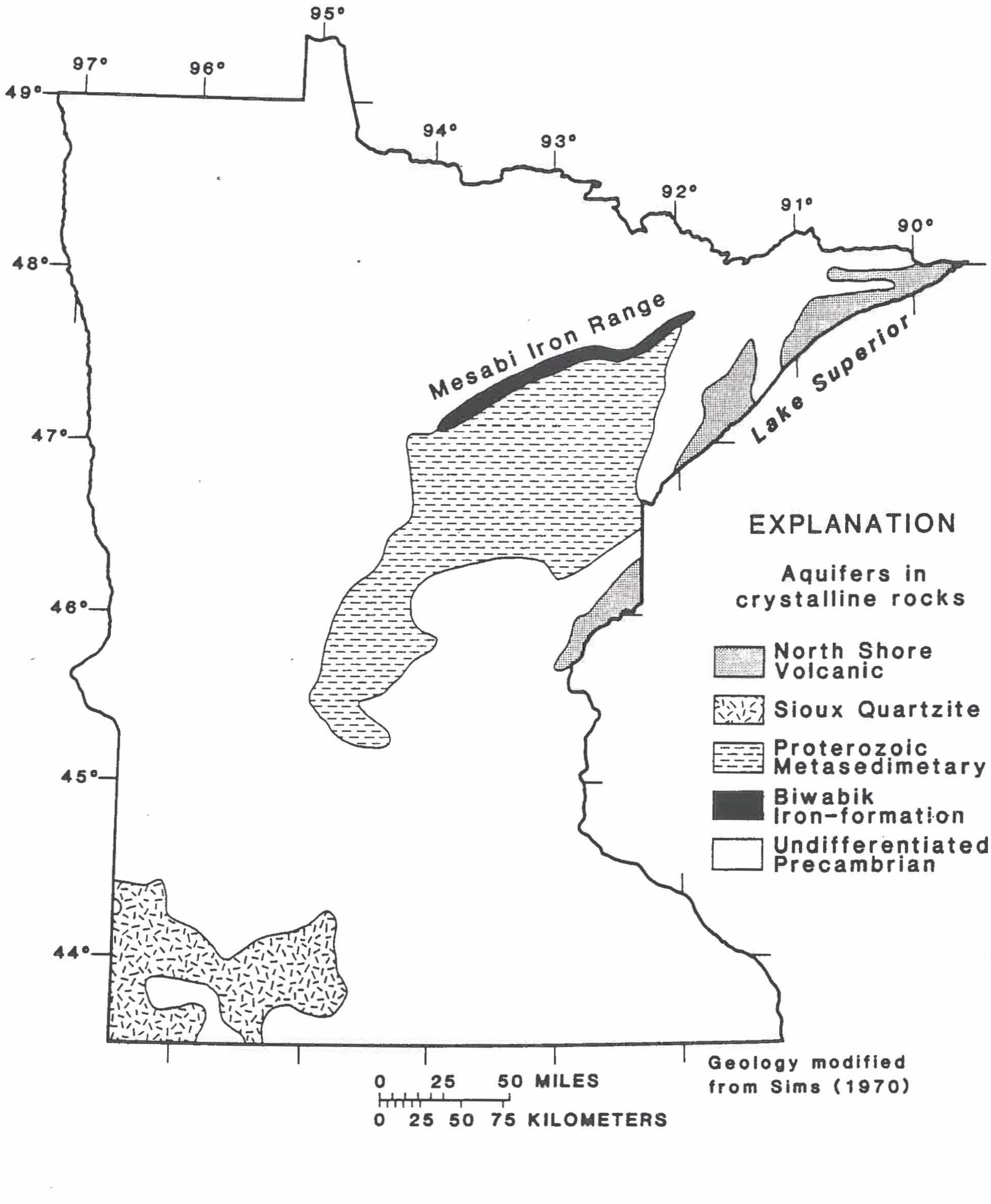

Figure 4 Extent of early Precambrian aquifers in Minnesota. From Adolphson et al., 1981.

Appendix B

Recommendations of the

Nitrogen Fertilizer Task Force on

THE NITROGEN FERTILIZER

MANAGEMENT PLAN

to the Minnesota Commissioner of Agriculture

To receive a copy of Appendix B "Recommendations of the Nitrogen Fertilizer Task Force on the Nitrogen Fertilizer Management Plan," please list your name and address below and send this form to:

Minnesota Department of Agriculture
Agronomy Services Division
90 West Plato Boulevard
St. Paul, Minnesota 55107

Name: ______________________________

Address: ______________________________

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