Effect of summer fallowing, perennial crop cover and conservation reserve practices on soil nitrate distribution
by Lynn S Pannebakker

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Soils
Montana State University

Abstract:

Dryland agriculture is an economically feasible method of producing crops in some parts of the semi-arid regions of the mid-west and western United States. Summer fallowing, which is commonly practiced in these regions, serves to replenish soil moisture and plant-available nitrate-nitrogen (N03--N), while also stabilizing production and more uniformly distributing the work load.

In areas where dryland agriculture is concentrated, summer fallowing may lead to N03--N contamination of ground-water due to lack of plant uptake of excess water and N. Under certain conditions of precipitation, soil percolation, slope, and cropping intensity, summer fallowing has been shown to cause elevated N03--N concentrations in shallow groundwater wells. These elevated N03--N levels have been detected in several areas of the U.S. Two areas where high N03--N concentrations have been repeatedly found in groundwater samples are in northeastern and central Montana.

Judith Basin and Fergus Counties of Montana were selected for soil sampling to assess any differences in soil NO3--N concentrations under three different land use systems: 1) crop fallow rotation, 2) acreage enrolled in the USDA Agricultural Stabilization and Conservation Service (ASCS) conservation reserve program (CRP), 3) and rangeland. Soil samples were collected to a depth of three m (10 ft) at four different sites in each county. Soil samples from all sampling depths at each of the eight sites were analyzed for gravel percent (>2 mm diameter) , N03--N load, and N03--N concentration. Samples from 0 m to 0.6 m were also analyzed for total-N concentration.

Trends in soil N03“-N give evidence that summer fallowing may be the cause of N03--N in shallow groundwater in some areas Of Montana where dryland cropping is practiced. Overall, average soil N03--N concentration throughout the sampled soil profile was 4.2, 2.0, and 1.3 mgkg-1 for the crop fallow, CRP, and rangeland land use practices, respectively. Average N03--N concentration in Fergus County ranged from 2.5 mgkg-1 to 20.4 mgkg-1 under crop fallow while it ranged from only 0.9 to 6.2 mgkg-1 and from 0.9 to 4.2 mgkg-1 for the CRP and rangeland uses, respectively. Average N03--N concentration in Judith Basin County ranged from 1.4 mgkg-1 to 6.9 mgkg-1, 0.6 mgkg-1 to 2.0 mgkg-1, and from 0.8 mgkg-1 to 1.4 mgkg-1 under crop fallow, CRP, and rangeland land use systems, respectively. Assuming that extensive use of crop fallow causes increased N03--N concentrations under some conditions, alternative land use management practices may be effective in reducing N03--N levels in such areas.

EFFECT OF SUMMER FALLOWING, PERENNIAL CROP COVER AND CONSERVATION RESERVE PRACTICES ON SOIL NITRATE DISTRIBUTION

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Lynn S. Pannebakker

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Lynn S. Pannebakker

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4/4/94
Date

James W. Bauder
Chairperson, Graduate Committee

Approved for the Major Department

4/6/94
Date

Head, Major Department

Approved for the College of Graduate Studies

4/11/94
Date

R L Brown
Graduate Dean

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ABSTRACT

Dryland agriculture is an economically feasible method of producing crops in some parts of the semi-arid regions of the mid-west and western United States. Summer fallowing, which is commonly practiced in these regions, serves to replenish soil moisture and plant-available nitrate-nitrogen (NO_3^--N), while also stabilizing production and more uniformly distributing the work load.

In areas where dryland agriculture is concentrated, summer fallowing may lead to NO_3^--N contamination of groundwater due to lack of plant uptake of excess water and N. Under certain conditions of precipitation, soil percolation, slope, and cropping intensity, summer fallowing has been shown to cause elevated NO_3^--N concentrations in shallow groundwater wells. These elevated NO_3^--N levels have been detected in several areas of the U.S. Two areas where high NO_3^--N concentrations have been repeatedly found in groundwater samples are in northeastern and central Montana.

Judith Basin and Fergus Counties of Montana were selected for soil sampling to assess any differences in soil NO_3^--N concentrations under three different land use systems: 1) crop fallow rotation, 2) acreage enrolled in the USDA Agricultural Stabilization and Conservation Service (ASCS) conservation reserve program (CRP), 3) and rangeland. Soil samples were collected to a depth of three m (10 ft) at four different sites in each county. Soil samples from all sampling depths at each of the eight sites were analyzed for gravel percent (>2 mm diameter), NO_3^--N load, and NO_3^--N concentration. Samples from 0 m to 0.6 m were also analyzed for total-N concentration.

Trends in soil NO_3^--N give evidence that summer fallowing may be the cause of NO_3^--N in shallow groundwater in some areas of Montana where dryland cropping is practiced. Overall, average soil NO_3^--N concentration throughout the sampled soil profile was 4.2, 2.0, and 1.3 $mgkg^{-1}$ for the crop fallow, CRP, and rangeland land use practices, respectively. Average NO_3^--N concentration in Fergus County ranged from 2.5 $mgkg^{-1}$ to 20.4 $mgkg^{-1}$ under crop fallow while it ranged from only 0.9 to 6.2 $mgkg^{-1}$ and from 0.9 to 4.2 $mgkg^{-1}$ for the CRP and rangeland uses, respectively. Average NO_3^--N concentration in Judith Basin County ranged from 1.4 $mgkg^{-1}$ to 6.9 $mgkg^{-1}$, 0.6 $mgkg^{-1}$ to 2.0 $mgkg^{-1}$, and from 0.8 $mgkg^{-1}$ to 1.4 $mgkg^{-1}$ under crop fallow, CRP, and rangeland land use systems, respectively. Assuming that extensive use of crop fallow causes increased NO_3^--N concentrations under some conditions, alternative land use management practices may be effective in reducing NO_3^--N levels in such areas.

INTRODUCTION

Voluntary tests of well water samples from throughout Montana during 1989 and 1990 raised concerns about elevated NO_3^--N levels in rural well water samples in central and northeastern Montana (Bauder et al., 1993). Land in these areas is commonly used for dryland cereal grain production. The average NO_3^--N concentration of 66 well water samples from Judith Basin County was 7.3 mgL^{-1}, with 28% of the well water samples from the county having NO_3^--N concentrations exceeding the U.S. Environmental Protection Agency (EPA) standard of 10 mgL^{-1}. One hundred forty-five well water samples from Fergus County had an average NO_3^--N concentration of 4.6 mgL^{-1}, with 11% of well water samples from the county exceeding the EPA standard for NO_3^--N. Nitrate-N concentrations as high as 95.9 mgL^{-1} were measured in samples from the two counties. In contrast, the average NO_3^--N concentration of 3332 samples from throughout Montana was 2.54 mgL^{-1}. Previous studies have shown significant correlations between NO_3^--N leaching below the crop root zone and the practice of summer fallow, particularly in gravelly soils. Furthermore, Bauder et al. (1993) proposed the hypothesis that elevated NO_3^--N levels in wells located in several areas in Montana may be attributed to the summer fallow practice.

OBJECTIVES

Private well water test results and interpretations obtained in Montana between 1989 and 1991 were used as a basis of the hypothesis that NO_3^--N contamination of groundwater in some areas of Montana can be attributed to summer fallow practices under certain physiographic and climatic conditions. The objective of the study reported herein was to determine if differences in soil NO_3^--N concentrations exist among crop fallow, permanent vegetative cover (rangeland), and land recently converted from crop fallow to CRP within Judith Basin County, Montana and Fergus County, Montana. The relationship between NO_3^--N trends under the three land use practices and differences in soil and NO_3^--N contamination of groundwater was then assessed. A second objective was to attempt to define more precisely physiographic and land use characteristics which could be correlated with elevated NO_3^--N concentrations in shallow ground water in Judith Basin and Fergus Counties of Montana.

LITERATURE REVIEW

Plants are dependent upon water and nutrients for production. Although all known elements are not required for plant growth, a number of elements are essential to plant nutrition and, therefore, must be maintained at adequate levels in order to insure healthy plant development. In particular, green plants are highly dependent on available nitrogen (N), a nutrient which is continuously recycled through the biosphere in the N cycle. All aspects of the N cycle have some effect on the quantity of N found in soil. The various processes which occur throughout this cycle influence the amount of available N for plant use. The cycle is composed of fixation, nitrification, ammonification (or mineralization), and denitrification phases (Foth, 1984). When plant needs for N exceed the amount of N available through natural cycling, alternative measures such as fertilization and/or crop fallow rotations can be implemented to supplement or accumulate (respectively) the amount of N needed for healthy plant growth.

Fallowing allows soil to rebuild its NO_3^--N supply through organic matter breakdown (mineralization) in combination with minimal (if any) plant uptake. Crop fallow rotations also offer the opportunity to build up soil moisture reserves for subsequent plant uptake.

Precipitation is the only source of moisture input in dryland farming in much of the northern Great Plains. Fallowing, therefore, is usually practiced wherever growing season evapotranspiration exceeds annual moisture input. Consequently, one purpose of crop rotations in arid and semi-arid agricultural regions is to simultaneously minimize soil moisture withdrawal by plants while rebuilding the available nutrient supply.

Although N is essential for plant growth and productivity, excessive amounts of N can become problematic when N, in its highly mobile NO_3^--N form, is leached into groundwater. Nitrate leaching has been linked to fallowing under conditions of N accumulation in the soil in excess of crop requirements in combination with precipitation amounts during the fallow period which exceed the soil's water holding capacity (Custer, 1977).

Although the atmosphere is composed of 79% inorganic nitrogen (N_2), this atmospheric N is unavailable to most plants. Atmospheric N makes up 99% of the total amount of N in the N cycle (Foth, 1984). Almost all (99%) of the N found within the soil is contained in organic matter. More directly related and available to plants are three forms of N found in mineral soils. These include organic N from humus, inorganic ammonium (NH_4^+-N), which is fixed by clay minerals, and soluble NH_4^+-N ammonium and NO_3^--N. Approximately one-half of the organic N found in soil is in the form of amino compounds. The form of the rest is uncertain. Only 1 to 2% of the soil's total N supply is in the form of soluble NH_4^+-N and NO_3^--N (Brady, 1984). Nitrogen, as a free NH_4^+-N ion, does not persist in the soil and, thus, most of the inorganic N in aerated soils and water is in the form of the NO_3^--N anion (Delwiche, 1981). Nitrate's high mobility explains the frequency of its association with groundwater contamination.

The Nitrogen Cycle

The N cycle can be subdivided into three subcycles, those being: the elemental, heterotrophic, and autotrophic subcycles. The elemental subcycle is the link between biological life and the atmospheric N pathways, including biological N_2 fixation and denitrification. The heterotrophic subcycle is made up of processes carried out by heterotrophic microorganisms and includes mineralization, energy derived from organic matter decomposition, and the products NH_3-N and NH_4^+-N. The autotrophic subcycle includes activities of green plants, including photosynthesis and buildup of primary organic N compounds. All three subcycles are dependent upon mineralized N (Stevenson, 1982).

Although all phases of the N cycle influence the amount of N present in the soil, mineralization, nitrification, and denitrification are of particular importance when considering the process of NO_3^--N leaching to groundwater. These two processes, nitrification and denitrification, occur within the soil and directly determine whether a N shortage or excess is present in relation to plant requirements. The processes also directly influence the amount of NO_3^--N produced and/or consumed. The other processes in the N cycle directly influence the quantity of N taken up or released into the atmosphere.

Mineralization

Organic N compounds (found in organic matter) are converted to inorganic forms such as NH_3-N and NO_3^--N by the process of mineralization (Delwiche, 1981). Organic N is transformed into N forms that are used by plants to carry out photosynthesis (Jansson and Persson, 1982). Mineralization is frequently referred to as ammonification in as much as the final product of mineralization is NH_3-N. Approximately three to four percent of the N contained in soil organic matter may be mineralized during a single year (Foth, 1984).

Mineralization is carried out by soil organisms. Soil microorganisms break down and hydrolyze organic N compounds by a process of enzymatic digestion. The process proceeds best in well drained, aerated soils in the presence of basic cations. The reaction proceeds as follows [Eq. 1,2] (Brady, 1984):

$$R\text{-}NH_2 + HOH \text{---}> R\text{-}OH^- + NH_3 + \text{energy} \qquad [1]$$

$$2NH_3 + H_2CO_3 \text{--->} (NH_4)_2CO_3 \text{--->} 2NH_4^+ + CO_3^{-2} \qquad [2]$$

Most NH_3 formed in soils is rapidly converted to NH_4^+-N because of readily available hydrogen ions. Since NH_4^+-N can be utilized by plants and has a positive charge (allowing it to be strongly adsorbed onto the soil exchange complex), leaching of NH_4^+-N is minimal (Foth, 1984). The amount and extent to which mineralization occurs determine the concentration of N found in the soil.

Immobilization

Immobilization, the process whereby N is complexed into organic matter, is closely related to mineralization. Mineral nutrients are assimilated by microbes or biomass as part of the immobilization process, thereby converting mineral nutrients into organic forms. The net difference between the degree to which immobilization and mineralization occur determines the N supply available for the autotrophic and elemental subcycles of the N cycle (Jansson and Persson, 1982). Through immobilization, mineral N is converted into organic forms which are tightly held in the soil. Immobilization does not occur below the zone of biological activity in the soil (Foth, 1984).

Coarse-textured soils generally have low organic matter content, thereby decreasing the potential for mineralization. Consequently, coarse-textured, sandy soils have limited capacity to supply significant amounts of available N for crop uptake. In contrast, well-drained, organic soils have the potential to provide greater amounts of N than water-logged soils. Aerobic decomposers are responsible for breakdown of organic N in such well-drained soils. Insufficient oxygen inhibits the mineralization process in water-logged soils (Foth, 1984).

Nitrification

Mineralization is followed in the N cycle by nitrification. Nitrification is the process whereby NH_4^+-N is oxidized to NO_3^--N or nitrite (NO_2^-) (Delwiche, 1981). Nitrification is a two-step process which proceeds when NH_4^+-N from mineralization is not absorbed by plants or microorganisms, or adsorbed on the soil exchange complex (Foth, 1984). The first step is production of NO_2^--N ions by Nitrosomas bacteria (primarily Nitrosomas, Nitrosolobus, and Nitrosospira); the second phase is oxidation of the NO_2^--N ions to NO_3^--N by Nitrobacteria. Nitrite is toxic to higher plants (Brady, 1984). All four of the microorganisms responsible for oxidation are autotrophic (Brady, 1984). The second step of the nitrification process proceeds soon after the first. The total process is as follows [Eq. 3,4]:

1) $NH_4^+\text{-N} + 1\ 1/2\ O_2 \text{--->} NO_2^-\text{-N} + 2H^+ + H_2O + \text{energy}$ [3]

2) $NO_2^-\text{-N} + 1/2\ O_2 \text{--->} NO_3^-\text{-N} + \text{energy}$ [4]

Because the reaction releases hydrogen ions, soil acidity is often increased (Foth, 1984).

Nitrogen is released into the soil as NH_4^+-N ions when nitrifying organisms die. The NH_4^+-N is converted to NO_2^--N when combined with oxygen, yielding energy (Delwiche,1981). Since the process is carried out by nitrifying bacteria, factors which affect these organisms and their populations strongly affect the rate of nitrification. These factors include the NH_4^+-N ion concentration, aeration status, soil temperature, soil moisture, soil pH, presence and concentration of exchangeable bases, fertilizer additions, C/N ratio, and pesticide applications (Brady, 1984).

A supply of NH_3 in the soil is necessary for nitrification to occur. However, high NH_3 concentrations in the soil are toxic to Nitrobacter and will constrain nitrification. Restriction of nitrification causes a buildup of NO_2^--N, which can be toxic to higher plants. Alternatively, a high C/N ratio in the soil prevents microbial release of NH_3 which also inhibits the nitrification process (Brady, 1984).

The rate of nitrification is very temperature-dependent; the minimum soil temperature at which nitrification begins is approximately $4^{o}C$ ($40^{o}F$). Nitrification ceases at temperatures in excess of $50^{o}C$. The optimum soil temperature range for nitrification is between $27^{o}C$ and $32^{o}C$ ($80-90^{o}F$) (Brady, 1984).

The amount of oxygen available for oxidation plays a significant role in the rate of nitrification. Increasing soil aeration (to a certain point) increases the rate of the nitrification. Due to this phenomena, plowed and cultivated soils tend to have higher nitrification rates than undisturbed soils or soils under minimum tillage (Brady, 1984).

The nitrification process is hindered by either very high or very low soil moisture content. It is generally assumed that the optimum soil moisture levels for higher plants also apply to nitrification. However, nitrification will proceed significantly at, and sometimes below, the permanent wilting coefficient (Brady, 1984).

Abundant base cations (Ca^{+2}, Mg^{+2}) in the soil solution and on the soil exchange complex are needed for nitrification to occur. Soil acidity has little effect on nitrification if enough base cations are available. Nitrification is generally weak in mineral soils with low pH (Brady, 1984).

Fertilizer additions have the potential to facilitate or restrict nitrification. A reasonable balance of N, P, and K in the soil promotes nitrification. Since high concentrations of NH_3 are toxic to <u>Nitrobacter</u> but not to <u>Nitrosomonas</u>, NO_2^--N tends to accumulate in basic soils under conditions of high NH_3 concentrations (Brady, 1984).

Pesticides, when applied at recommended rates, have minimal effect upon nitrification. At excessively high rates of application, some pesticides inhibit and some pesticides facilitate nitrification (Brady, 1984).

Certain chemicals significantly inhibit nitrification. Nitrapyrin inhibits oxidation of NH_3 to NO_3^--N, particularly in clay-rich soils. Effects of nitrapyrin are least in sandy soils and/or when fertilizers are applied alongside growing plants. Coating urea with sulfur slows the rate of urea conversion to NH_3 and the rate of conversion from NH_3 to NO_3^--N. Thus, less NO_3^--N is readily available for denitrification. Other nitrification inhibitors include dicyandiamide (Dd, DNDN, and Dicyan), Thiourea (Tu), Guanylthiourea (ASU), Sulfathiazole (ST), 2-amino-4-methyl-6-trichloromethyl-1,3,5 triazine (MAST) and 4-amino-1,2,4-trizole-HCl (ATC) (Brady, 1984).

Denitrification

Bacterial denitrification follows nitrification in the N cycle. Denitrification is most common and rapid in soils with poor drainage and low aeration. Denitrification results in reduction of the NO_3^--N content. Nitrate is converted to N_2, which escapes from the soil surface as a gas, either N_2 or N_2O; neither of these denitrification products is available to plants (Brady, 1984). Few microorganisms within the biological cycle are capable of fixing atmospheric N (Delwiche, 1981). Denitrification, therefore, keeps atmospheric N and soil N in a quasi-equilibrium (Foth, 1984). The reaction proceeds as follows in the soil under anaerobic conditions [Eq. 5]:

$$2NO_3^- \text{-N} \text{---}> 2NO_2 \text{---}> 2NO \text{---}> N_2O \text{---}> N_2 \qquad [5]$$

Nitrate acts as an electron acceptor for the organism oxidizing the organic (sometimes inorganic) compound while obtaining energy from the organic compound (Delwiche, 1981). Each stage of denitrification involves a particular reductase, including nitrate reductase, nitrite reductase, nitric oxide reductase, and nitrous oxide reductase, respectively. Furthermore, denitrification can cease at any stage, resulting in release of gaseous products (Brady, 1984).

Denitrification is primarily carried out by microbes, although some chemical reduction does contribute to denitrification. Biochemical reduction is carried out by facultative anaerobes which prefer elemental oxygen rather than oxygen obtained from NO_3-N. Facultative anaerobes will function, however, under inadequate aeration by using the combined oxygen in NO_3^--N and some of their reduced products (Brady, 1984).

The amount of N_2 and N_2O produced during denitrification varies from minuscule to as much as 20% of the NO_3^--N available for denitrification under field conditions. The proportion of N_2O to N_2 produced increases more as the soil pH is increased. Denitrification tends to be more complete under alkaline conditions than under neutral or acidic soil conditions. Acidic soils with low pH tend to produce more nitric oxide (N_2O) and have low rates of denitrification. Denitrification proceeds most effectively at pH values around neutrality (Delwiche, 1981). The production of N_2 and N_2O is also dependent upon the concentration of NO_3^--N in the soil; N_2O is the predominant gas produced under conditions of high NO_3^--N concentrations, while N_2 is the primary product of denitrification when the concentration of NO_3^--N is low (Rolston, 1981).

The rate of denitrification is dependent on the amount of carbon available for microbial use, availability of carbon in the profile and the position of carbon relative to NO_3^--N in the soil. Oxidized carbon is frequently the most limiting factor in the denitrification process because it supplies the substrate for growth of denitrifying bacteria. The amount of carbon in most soils is greatest at the surface and decreases with depth. Organic matter on the soil surface and from shallow roots is the source of most carbon. As precipitation and irrigation water leach NO_3^--N below the rooting depth, the potential for denitrification decreases. Under completely anoxic or anaerobic conditions, denitrification is dominantly governed by the amount of organic matter in the soil (Rolston, 1981). Temperature also influences the rate of denitrification. N_2O is the dominant product of denitrification at low temperatures and N_2 is the dominant product at higher temperatures (Rolston, 1981).

Soil and Land Use Effects on NO_3^--N Production

The amount of precipitation, relative to soil water holding capacity, directly relates to the amount of excess NO_3^--N produced. The amount of soil water available for plant uptake and the nutrient supply are important to the welfare of plant species. Fallowing is practiced with the intent of storing the current year's moisture in the soil for crops of the following year. In this manner, the total precipitation from nearly two years is available for accumulation in the soil. "Thus the way to control drought lies in the transformation of the natural water balance of a farm into a larger moisture accumulation in the soil, in the reduction of evaporation from the soil, and the breeding of drought-resistant plants with productive transpiration." (Kovda, 1980).

The efficiency of summer fallowing is dependent on the interaction of several physical processes. Rainfall infiltrates the soil surface and moves through the soil profile predominantly as unsaturated flow. Volume and rate of water flow are dependent upon soil texture and amount of water infiltrating the soil surface. As the soil surface dries due to evaporation, water moves upward from below, thereby decreasing downward hydraulic conductivity and flow velocity. The soil surface dries further (due to low conductivity) and moisture moves only as vapor flow to the surface layer. The dry soil surface restricts evaporation of underlying water in the soil profile. Recurring precipitation moistens the soil to a greater depth than previously but this water is also subsequently trapped once the surface dries. Repeating this cycle throughout the fallow period progressively increases the depth to which rainfall infiltrates, fills soil pores, and is stored for subsequent plant use (Figure 1).

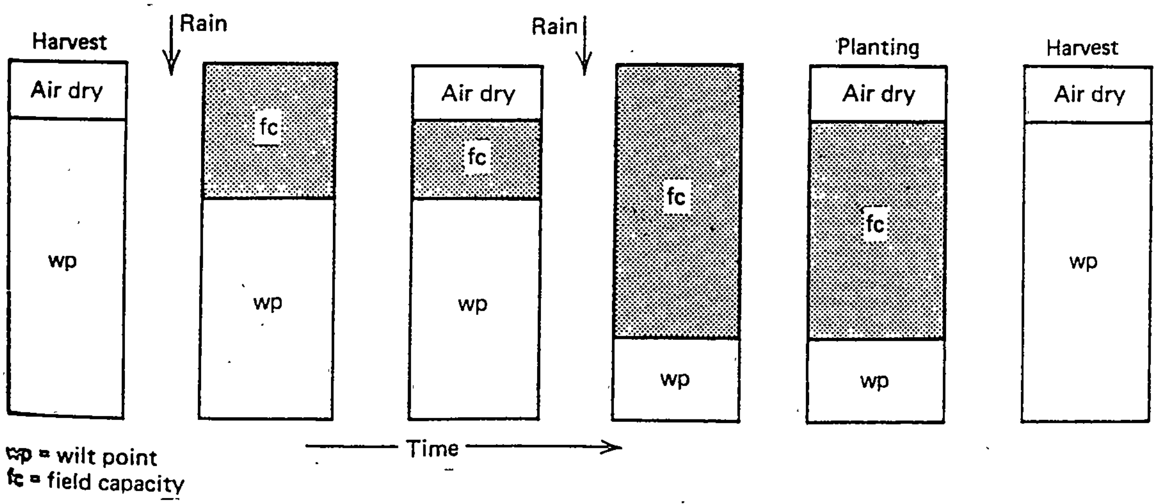

Figure 1: Changes in soil water during a summer fallow period for wheat production (Foth, 1984).

Fallow Efficiency

A common approach to fallowing is to allow stubble or plant residue and debris from the previous crop to remain on the soil surface until the following spring (Brady, 1984). Stubble reduces water and wind erosion and allows nutrients from decomposition of organic matter to penetrate the profile. The soil is then tilled before any significant weed populations are established in the spring. Thereafter, weeds are deterred with cultivation or herbicides (Brady, 1984).

Historically, fallowing was considered a relatively efficient means of conserving available moisture and N during the non-cropped period. Foth (1984) reported that moisture stored during the fallow period is approximately 25% of the total precipitation that falls during the fallow period. Lindwall and Anderson (1981) reported on studies in southern Alberta comparing eight summer fallow methods for spring wheat production. During the fallow period, fallow treatments had an average precipitation storage of about 14%. Overall average moisture conservation during the entire fallow period was approximately 25%. Chemical fallow resulted in the most available moisture in the root zone at the end of the fallow period while tillage fallow methods resulted in the least net gain in stored soil water. Nitrate-nitrogen concentration in the 0 to 60 cm soil depth was least under blade cultivation and highest where herbicides were used for weed control between May and September and blade cultivation was used in October (Table 1).

Crop yields from fallowed soils are usually greater than crop yields from non-fallowed soils, due to greater amounts of available water and N in the soil profile resulting from fallowing (Table 2).

Table 1. Nitrate-nitrogen in the soil and mature plant height, yield, and protein content of wheat, 1968-1976 (Lindwall and Anderson, 1981).

Treatment	NO_3^--N to 60-cm before seeding * (kg/ha)	Mature plant height(cm)	Yield (kg/ha)	Protein content (%)
B	83a	74.3bc	1809c	13.9a
OW	91a	72.6c	1846bc	14.0a
FB/H	85a	74.7bc	1894bc	13.9a
FB/H/FB	84a	74.5bc	1925bc	14.0a
H/FB	92a	76.3ab	2104a	14.0a
SB/H	84a	74.6bc	1845bc	13.9a
SOW/H	86a	74.4bc	1852bc	13.9a
H	84a	75.7ab	1979ab	13.6a

*5-yr mean only
a-c Within each column, means followed by the same letter do not differ (P=0.05).

Key:
B=Blade cultivation (May-Sept)
OW=One way disc cultivation (May-Sept)
FB/H=Blade cult. after harvest/herbicides (May-Sept.)
FB/H/FB=same as above followed by blade cult. in Oct.
H/FB=Herbicides (May-Sept.)/Blade cult. (Oct.)

Table 2. Influence of summer fallow in alternate years on the available soil moisture at wheat seeding time and the yield of wheat fallowing (Brady, 1984).

	Available water at seeding time cm	in.	Wheat yield kg/ha	lb/A
Mandan, ND (av. 20 years)				
Wheat after fallow	17.9	7.08	4618	4120
Wheat after wheat	6.3	2.48	2578	2300
Hays, KS (av. 23 years)				
Wheat after fallow	22.2	7.96	1836	1638
Wheat after wheat	7.4	2.90	1170	1044
Garden City, KS (av. 13 years)				
Wheat after fallow	11.9	4.67	1042	930
Wheat after wheat	2.7	1.08	551	492

Soviet scientists studied crop production efficiency under fallow and reported winter crops on black fallow to produce 1.5 to 2.0 times greater yields than spring crops in a common year (Figure 2).

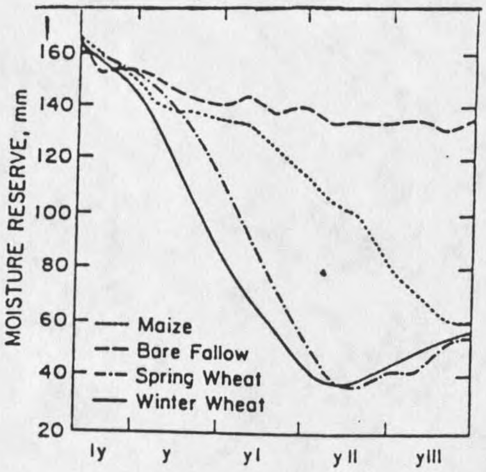

Figure 2: Perennial mean reserve of productive moisture (mm) in a 1-m soil layer under various crops in the zone of complete spring soaking and a deep level of groundwater (Shulgin, 1967). Symbols on x-axis represent the months of April through August.

Several techniques can be incorporated into the fallow process to increase the benefits associated with moisture and nitrogen storage. Straw-covered soils average 3 to 5 times greater total water infiltration and greater infiltration rates than bare soils. Unger (1990) demonstrated that wheat yields and water storage increased with degree of weed control and maintenance of a surface residue (Table 3 and 4).

In summary, the general benefits of summer fallow include increased yield due to increased moisture and N storage, improved weed control, stability in production, and distribution of the work load. However, some additional costs can be associated with fallow rotations. These include loss of production for one year, cost of fallow tillage, and potential for increased soil erosion and saline

Table 3: Progress in fallow systems with respect to water storage and wheat yields in Akron, Colorado (Greb et al., 1967).

Years	Tillage during fallow	Fallow water storage (mm)	Fallow water storage (% of precip)	Wheat yield (Mgha^{-1})
1916-30	Maximum tillage; plow,harrow (dust mulch)	102	19	1.07
1931-45	Conventional tillage shallow disk, rod weeder	118	24	1.16
1946-60	Improved conventional tillage; begin stubble mulch in 1957	137	27	1.73
1961-75	Stubble mulch; begin minimum tillage w/ herbicides in 1969	157	33	2.16
1975-90	Projected estimate;minimum tillage; begin no tillage in 1983	183	40	2.69

Table 4. Straw mulch effects on soil water storage efficiency in Sidney, Montana; Akron, Colorado; and North Platte, Nebraska 1962 to 1965 (Greb et al., 1967).

Mulch rate (Mgha^{-1})	Fallow period precipitation (mm)	Water storage efficiency (%)
0.0	355	16
1.7	355-549	19-26
3.4	355-648	22-30
6.7	355-648	28-33
10.1	648	34

seep (Ali and Johnson, 1981). Recent studies have also reported that NO_3^--N leaching into groundwater may be associated with fallowing under certain conditions (Bauder et al., 1993).

Nitrogen, in the form of NO_3^--N, can become a source of groundwater contamination when it is leached from the soil during deep percolation (Cassel et al., 1971). Leaching of NO_3^--N can occur when the NO_3^--N is not stabilized through denitrification (Jury and Nielsen, 1989) or taken up by plants. It is not unusual for water to infiltrate the soil to below 1.5 m, commonly resulting in significant NO_3^--N movement below the root zone (Cassel et al., 1971). Thickness of underlying unsaturated layers of soil influences the time required for effects of NO_3^--N leaching to become apparent in groundwater. Studies have shown that the significance of NO_3^--N leaching may not be evident for as long as thirty years (Juergans-Gschwind, 1989).

Nitrate Movement and Summer Fallow

Nitrate is extremely mobile in most soils due to the negative charge of the NO_3^--N molecule. Transport of dissolved NO_3^--N through the soil is accomplished by mass flow of the soil solution or through diffusion within the solution. Nitrate is stable only when it is biologically transformed in the soil to N_2 through denitrification. Nitrate will leach downward as long as the amount of water entering the soil exceeds the evapotranspiration rate and soil water storage capacity (Jury and Nielsen, 1989). The amount of NO_3^--N leached is both site and land use specific (Juergans-Gschwind, 1989).

Summer fallowing can be a factor in NO_3^--N leaching, particularly when fallow-period precipitation exceeds soil water-holding capacity. The amount of water required to induce NO_3^--N leaching is dependent upon soil texture, crop root zone depth, and initial soil water content. During the fallow period when soils are relatively wet, mineralization proceeds very rapidly and most of the available N is oxidized to NO_3^--N. This NO_3^--N moves upward in the soil profile as the soil dries and downward as water leaches through the profile. The rooting pattern of subsequent crops determines the fate of NO_3^--N accumulated in the soil profile (Sprent, 1987). More N is often mineralized during the fallow period than is utilized by ground cover or subsequent crops, resulting in an excess of NH_3 and/or NH_4^+-N in the soil. Under these conditions, the possibility of leaching is increased (Lamb et al., 1985; Keeney, 1989).

Cultivation of fallowed and/or virgin soils enhances the likelihood of NO_3^--N leaching. Studies by Custer near Rapelje, MT (pers. comm., 1977) indicate that NO_3^--N problems arise when the following conditions exist. Nitrate must be produced through mineralization yet be unavailable or unused by plants, and a transporting agent must be present (excess water percolating through the soil). The accumulation or leaching of NO_3^--N is dependent upon soil and geologic properties, fertilization rate and timing, crop type, and cultivation method. In these studies, the amount of NO_3^--N found in groundwater below cultivated land was greater than that found under newly cultivated land followed by a period of non-cultivation (Custer, 1977). Cultivation of sod increased soil aeration and soil moisture content, thus increasing the rate of mineralization and nitrification (Custer, 1976). Nitrogen accumulates in the soil under fallow in a manner similar to accumulation under sod. Increased water storage during the fallow period often results in percolation of excess water below the root zone, providing a NO_3^--N transport mechanism upon cultivation (Bahls and Miller, 1973; Power, 1970, 1972).

Lamb et al. (1985) also studied N losses through time (1970-1982) as related to cultivation method on a western Nebraska Duroc loam (fine, silty, mixed, mesic, Pachic Haplustoll). In anticipation of a winter wheat crop, soils were cultivated in a crop fallow rotation. The effects of three tillage systems, i.e., no-till, stubble mulch, and plow (bare fallow), on N movement in the soil were analyzed. Nitrogen losses from 0 to 30 cm were 3% on no-till, 8% on stubble mulch, and 19% under plow after 12 years. Nitrogen losses from the soil under plowing decreased exponentially through time. No-till and stubble mulch systems resulted in constant rates of nitrogen losses through time. Most of the differences in nitrogen losses occurred in the top 10 cm of soil; 40 to 60% of the total N lost was in the form of non-hydrolyzable N. No significant loss of non-exchangeable NH_4^+-N was measured in the cultivated soil and exchangeable

NH_4^+-N did not change under any treatment (Lamb et al., 1985).

Dahnke et al. (1971) conducted a study in North Dakota in 1970 on eleven soils to characterize NO_3^--N behavior under summer fallowing. Fields in the study were fallowed in the summer of 1970 and NH_4NO_3 was applied at 224 kg/ha on one-half of each plot. The second half of each plot did not receive nitrogen. Samples were taken to 0.9 m before N application and two weeks following fertilization. Most of the applied N had been converted to NO_3^--N before the first sampling. Large NO_3^--N fluctuations were observed in the top 30 cm of soil; these fluctuations were associated with incorporation of stubble during cultivation. The amount of fluctuation decreased for two weeks immediately after cultivation and then increased again. The same results were observed in the summer of 1971. Little NO_3^--N moved below 60 to 90 cm in four of five plots. Nitrate-nitrogen accumulations averaged 33.6 kg/ha from November, 1970 through May, 1971. On the fifth plot, a loamy fine sand, up to 112 kg/ha of NO_3^--N leached below 90 cm due to a two week interval in June when 10 cm of rain occurred. Leaching was attributed to the low water holding capacity of the coarser textured soil. Leaching also occurred due to excessive rainfall on several fields which were planted to small grain in 1971. The conclusion of this study was that NO_3^--N fluctuations throughout the profile were related to cultivation and/or excess precipitation (on coarse soils) and that fallowing should not be practiced on coarser soils.

Kanwar et al. (1985) studied NO_3^--N movement in no-till and moldboard plowed field plots under two simulated rainfalls (12.7 cm and 6.35 cm). Simulated rainfall events were applied one day apart. Plowed plots were surface broadcast with approximately 150 kg/ha of NO_3^--N before and after tillage. Surface application of N fertilizer without subsequent tillage was used on the no-till plots. No-till plots had higher NO_3^--N concentrations in the top 30 cm of soil; 40% of the original amount of N applied was present after 12.7 cm of rain (29 kg/ha was leached) and 33% after an additional 6.35 cm of rainfall. The moldboard plow plots had 19% (122 kg/ha was leached) and 9% of the original amounts, respectively, following two simulated rainfall events.

A study on summer fallowed land in Flathead County, MT found high levels of NO_3^--N in the soil in the fall following the fallow period (Graham, pers. comm.). Ten cm of instantaneously applied water were required to move the NO_3^--N into the sand subsoil underlying a calcareous C horizon. The researcher concluded that 15 cm of actual precipitation would be required to achieve the same results. The chance of NO_3^--N leaching was probably greatest under conditions of excessive snowmelt. The same researcher found ponded soils to average 37 kg/ha NO_3^--N at 7.6 to 38.1 cm depths and non-ponded soils to average 31.4 kg/ha NO_3^--N at the same depths. Surface NO_3^--N was also lower on the non-ponded soils due to a greater amount of vegetation. Fertilized and subsequently irrigated sites at two locations had 83 and 112 kg/ha NO_3^--N at 20.3 to 35.6 cm depths.

Another factor influencing movement of NO_3^--N is plant uptake or the lack thereof. Bare soils, lacking plant uptake of N and lower transpiration values, tend to release more NO_3^--N than cropped soils. The depth of NO_3^--N leaching generally ranges from 0.3 to 3 m/yr, depending on climate, soil type, and cropping system. Soils under fallow generally will lose one to two times more NO_3^--N than cropped land and nine times more than grassland (Juergans-Gschwind, 1989).

Legg and Meisinger (1982) found that NO_3^--N accumulated at relatively high concentrations in fallowed soils. Studies conducted in Europe have shown that autumn precipitation moistens the soil, creating favorable conditions for mineralization. Mineralized NO_3^--N is subject to leaching into lower layers during the following winter when the land is in fallow (Juergans-Gschwind, 1989).

Gravel Content and Soil Profile Characteristics

The fine earth fraction of soil includes particles with diameter less than 2.0 mm (Soil Survey Staff, 1975). Coarse fragments are particles with diameter exceeding 2.0 mm. Presence of coarse fragments in the soil profile affects the physical, chemical, and hydrologic properties of the soil.

Coarse fragments can affect crop yield and ease of chemical leaching. Increasing coarse fragments decreases the water holding capacity of the soil and increases hydraulic conductivity of the soil. This results in increasing potential for nutrient, fertilizer, and pesticide leaching with increasing coarse fragments. Furthermore, Grewal et al. (1984) reported yield reductions of up to 50% and associated monetary return reductions as coarse fragment concentrations increased from 18% to 40%. Hanson and Blevins (1979) related amount and size distribution of coarse fragments to plant available water. They concluded that coarse materials <5 mm in diameter tend to be more weathered than larger fragments. Weathering results in greater porosity at fragment surfaces and, therefore, greater water holding capacity and plant available water. Data in the Montana pedon base (Jersey and Nielsen, 1992) suggest that concentration of coarse fragments tends to increase with depth for horizons containing more than 15% coarse materials. Several factors may contribute to this distribution. One possible reason is that a considerable portion of Montana was glaciated. Loess deposition over parent soils is common in glacial soils. Montana soils have also been affected by volcanic ash deposits (Montagne et al., 1982). Secondary deposits of loess and ash are of finer textures and, thus, increase profile depth and depth to coarse fragments associated with the initial parent materials.

Increased profile depth also acts to shield underlying materials from weathering near the soil surface. The coarser fragments at the deeper depths are therefore "preserved". Weathering at the soil surface degrades particles into smaller fractions. Thus, soil-sized particles (<2 mm) at the soil surface have been further degraded due to exposure to varying climatic conditions.

METHODS AND MATERIALS

Locating Sampling Areas

Areas where high concentrations of NO_3^--N are known to occur in Montana were found using Montana private well test data from 1989 and 1990 (Bauder et al., 1993). Two geographic areas were selected for investigation, those areas being in Judith Basin and Fergus Counties (Figure 3). Individual well owners and their places of residence within the study areas were located with assistance from the Judith Basin and Fergus County Extension Service agents and county Agricultural Stabilization and Conservation Service (ASCS) personnel. Individual well water test results and location of the respective wells were noted on county maps. Well water NO_3^--N concentrations were then categorized (high >10 mgL^{-1}, medium = 5-10 mgL^{-1}, low <5 mgL^{-1}) to determine the location where high concentrations of NO_3^--N in groundwater repeatedly occurred. A composite map was then assembled using soil survey sheets from Judith Basin and Fergus Counties. This map was used to identify general or broad geographic areas within each county where high and medium NO_3^--N concentrations in groundwater were repeatedly detected through well water sampling (5 to >10 mgL^{-1}).

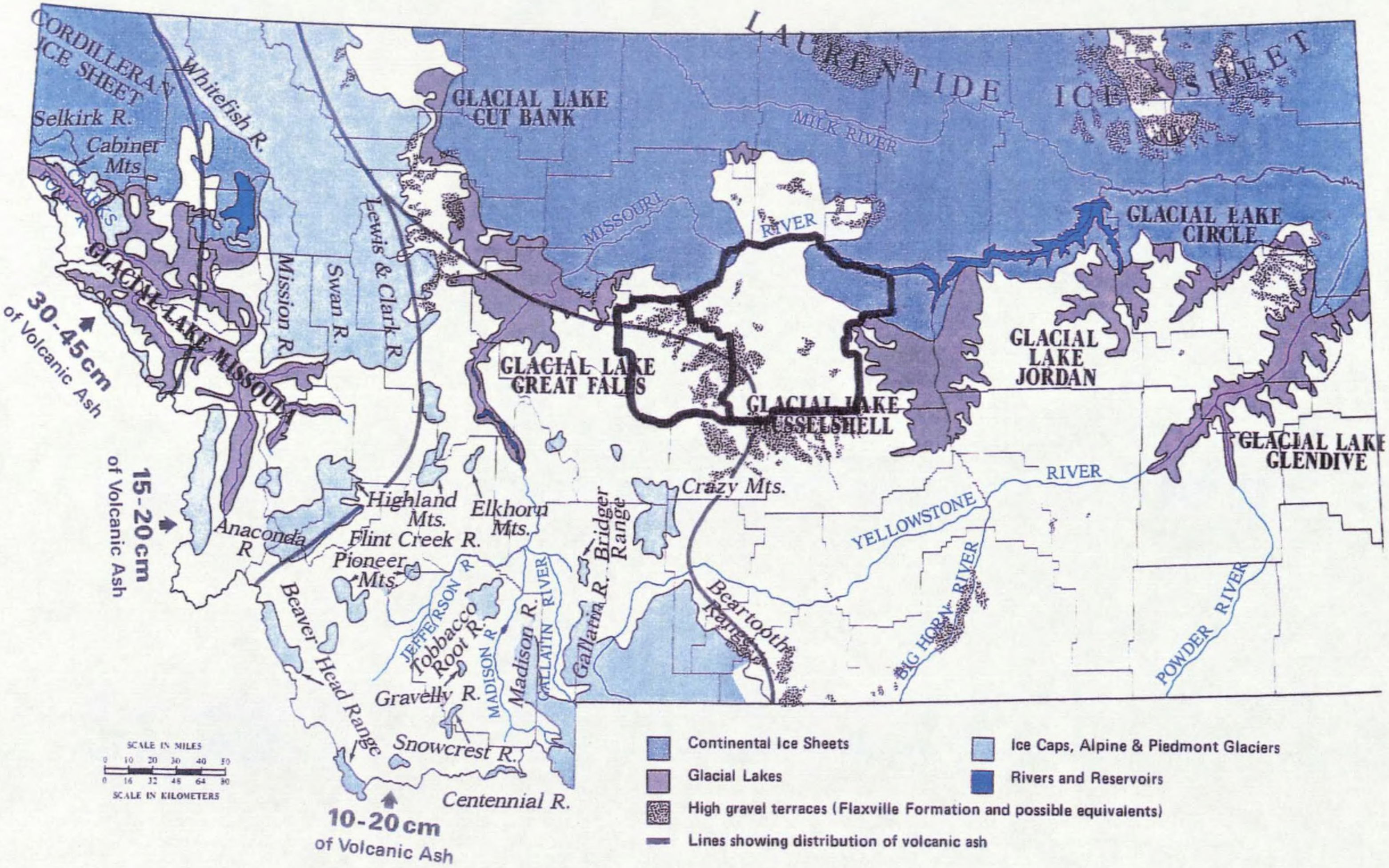

Figure 3. Location of high gravel terraces in Judith Basin and Fergus Counties, Montana (Montagne et al., 1982).

General Area Description

General information regarding the sampling areas is contained in Table 5. Information was obtained using the Montana Agricultural Potentials System (MAPS) (Caprio et al., 1990).

Table 5. Physiographic and climate data for the general sampling areas in Judith Basin County, MT and Fergus County, MT.

Parameter	County Judith Basin	Fergus
Growing deg. days (40 1/2 base) May-Oct.	3300-3500	3500-3600
Geology	Kc Colorado	Kc Colorado
Slope	2-4%	2-4%
Mean annual precipitation	16"	14-16"
Soil water holding capacity	5.2-8"	6.1-7.5"
General soils * (Montagne et al., 1982)	Gb5,Ap7,Nh1	Gb5,Ap7,Nh1,Np
Mean annual soil temperature (MAAT +2)	44.5-45.5 F	45.5 F
Soil depth classes	10-60+"	10-60+"
Average soil pH	7.3-7.6	7.2-7.6
Mean annual air temperature	42.5-43.5 F	44.5 F

*Key:
Gb5 = Argiborolls-Calciborolls
Ap7 = Haploborolls-Argiborolls-Ustifluvents
Nh1 = Haploborolls-Calciborolls-Argiborolls
Np = Mollisols-Entisols

Data in Table 5 represent two arbitrary sites whose legal descriptions fall approximately in the center of the four sampling sites within Judith Basin (Township 17N, Range 11E, Section 24) and Fergus Counties (Township 18N, Range 15E, Section 7). Therefore, values in Table 5 provide an average and may differ slightly from each individual site.

Variables/Treatments

Three land use practices which are predominant to these geographic areas were identified and selected for soil sampling: 1) crop fallow, 2) land enrolled in CRP, and 3) rangeland. These three land uses were chosen, based on the following criteria: each land use practice was located within approximately 30 m (100 ft) of the other two practices to reduce soil variability; rangeland sites were undisturbed (unplowed) for at least 30 years prior to sampling; CRP land was used for dryland agriculture prior to CRP enrollment. CRP land use was selected to serve as "remnant" crop fallow ground but not long-term rangeland. Land enrolled in CRP prior to 1989 (as opposed to more recent enrollment) was chosen for sampling to maximize the potential for observable NO_3^--N movement through the soil profile with time since enrollment in CRP. Crop fallow sites were chosen to show current soil NO_3^--N conditions under lands in fallow during sampling but recently cropped.

Individual Sample Site Selections

Potential cooperators who operated on land which fell within the problem areas outlined on the composite map mentioned above were identified. Possible cooperators were then contacted to determine their willingness to assist and to establish whether all three land uses existed within the

30 m (100 ft) boundary. Sites were then visited to determine sampling suitability in terms of accessibility and location. Where actual rangeland was unavailable, a long-standing fence line was used for sampling this treatment, with the assumption that land along the fence was undisturbed during the lifetime of the fence. Individual site descriptions and locations are defined in Table 6 and Figures 4 through 11.

Sample Collection

Four sites were sampled for each land use practice in both Judith Basin and Fergus Counties. Thus, a total of 8 sites were sampled with a crop fallow, CRP, and rangeland sampling point at each site.

Sites in Judith Basin County were sampled in June, 1992 and sites in Fergus County were sampled in July, 1992. Where possible, each of the land uses at every site was sampled to a depth of 3 m (10 ft). A soil sampling pit was excavated using a backhoe. From within each pit, samples were collected in plastic lined bags at 30 cm (1 ft) depth increments and placed in a cooler for preservation. Samples were then taken to the lab and stored in a cooler prior to analyses.

Table 6. Individual site and soil descriptions.

Site	Cooperator	Legal Description Twp.	Range	Sec.	1/4	Soil Series	Survey Sheet Number
1	E. Koski & E. Hall	16N	12E	8	SE	Danvers-Judith gravelly clay loam, 0-2% slope	49
2	J. Kulish E. Hall	17N	12E	12	NW	Danvers-Judith gravelly clay loam, 0-4% slope	30
3	L. Mikeson & B. Schmidt	17N	11E	1	SE	Danvers gravelly clay loam, 0-4% slope	28
4	E. Pollari	18N 18N	11E 11E	22 23	NE NW	Judith gravelly clay loam 4-8% slope	19,20
5	D. Donaldson	18N	15E	21	SW	Fairfield Danvers Loam, 0-2% slope	133
6.	W. Crabtree	18N	15E	30	SE	Fairfield Danvers Loam, 0-2% slope	147
7.	Hilltop Angus	19N	14E	24	NE	Martinsdale-Judith Loam, 4-8% slope	102
8.	L. Barber	19N	14E	33	NW	Danvers Clay Loam, 0-2% slope	117

Note: Sites 1 through 4 were obtained from the Judith Basin County Soil Survey (1967); site 5 through 8 were obtained from the Fergus County Soil Survey (1988).

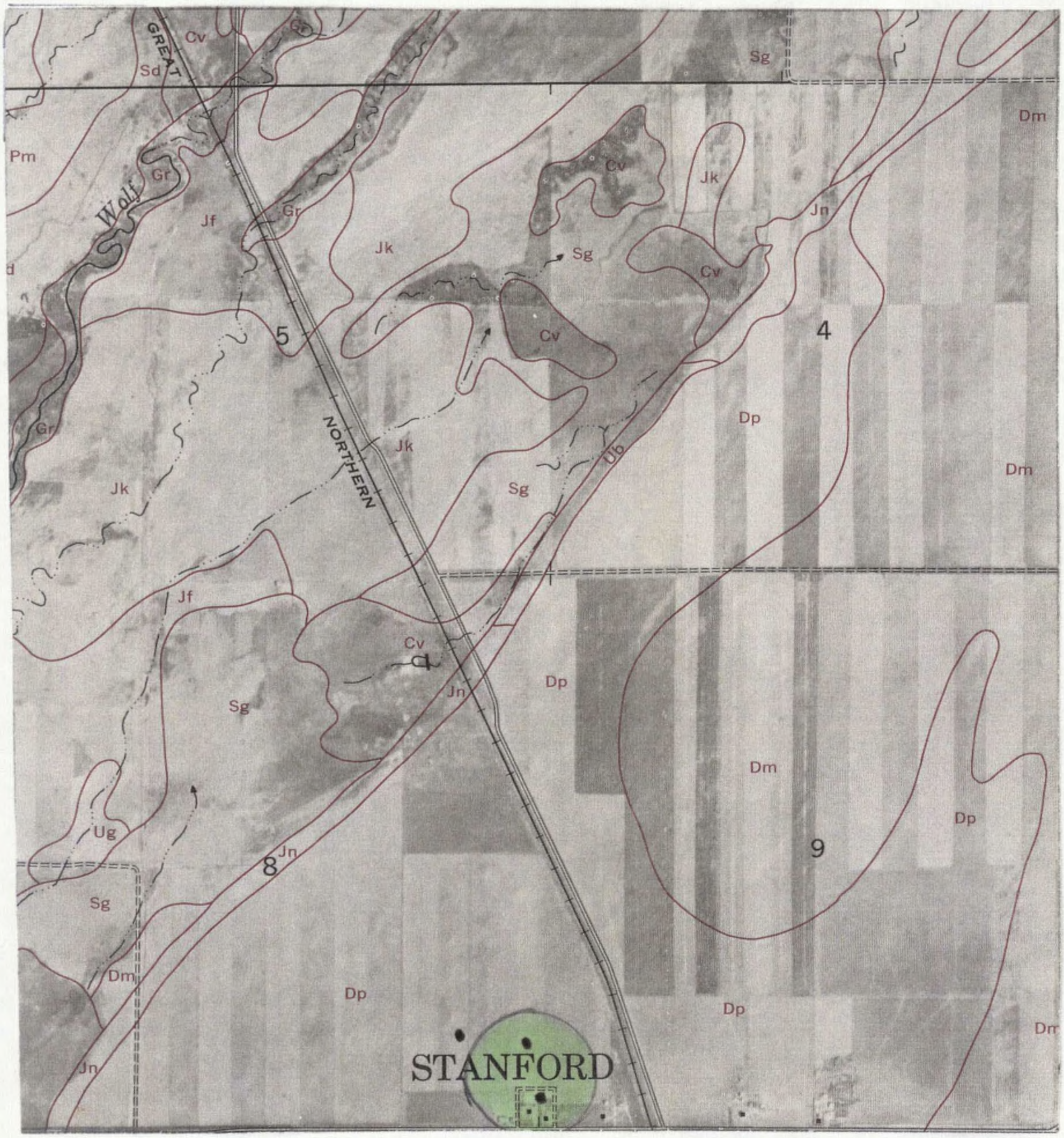

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Figure 4. Soil survey map of sampling site #1, Judith Basin County; the area encircled designates sampling location. (Source: Judith Basin County Soil Survey, 1967, Sheet #49)

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Figure 5. Soil survey map of sampling site #2, Judith Basin County; the area encircled designates sampling location. (Source: Judith Basin County Soil Survey, 1967, Sheet #30)

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Figure 6. Soil survey map of sampling site #3, Judith Basin County; the area encircled designates sampling location. (Source: Judith Basin County Soil Survey, 1967, Sheet #28)

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Figure 7. Soil survey map of sampling site #4, Judith Basin County; the area encircled designates sampling location. (Source: Judith Basin County Soil Survey, 1967, Sheet #19, 20)

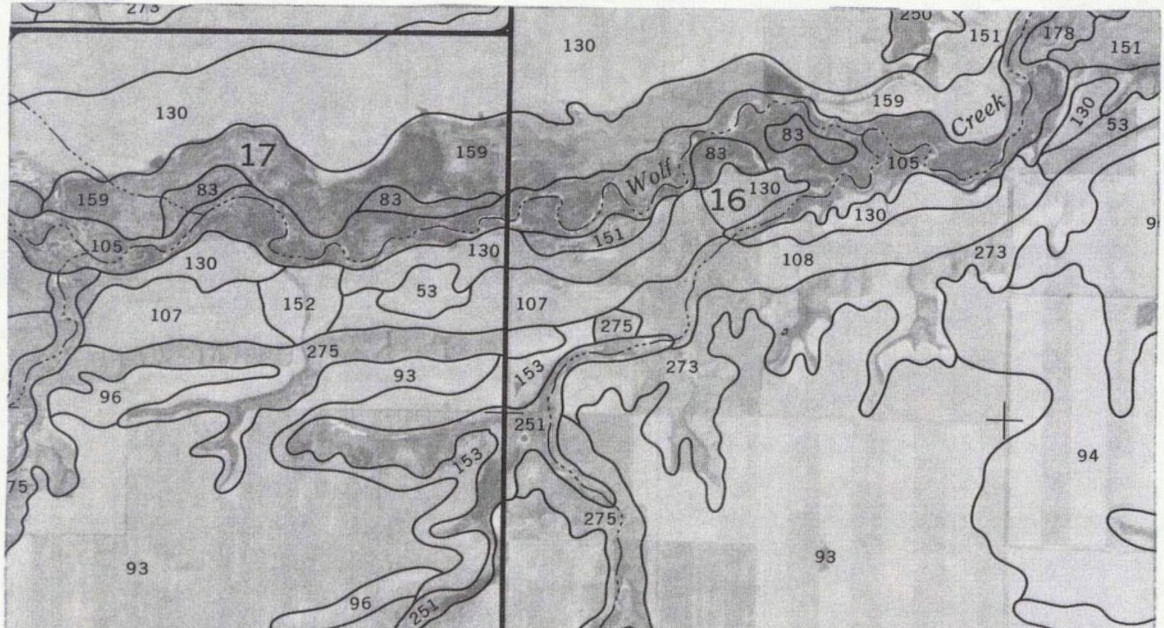

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Figure 8. Soil survey map of sampling site #5, Fergus County; the area encircled designates sampling location. (Source: Fergus County Soil Survey, 1988, Sheet #133)

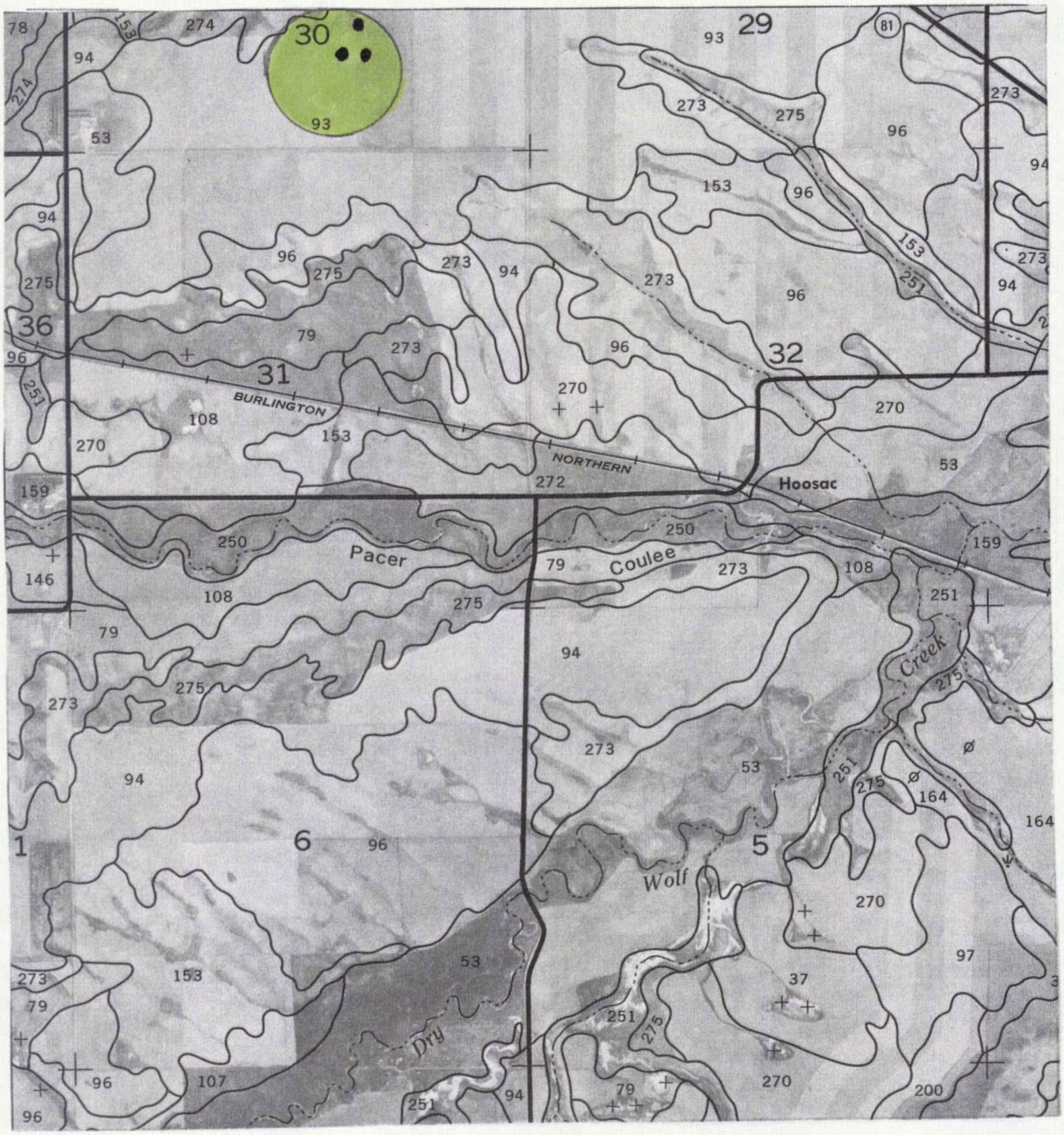

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Figure 9. Soil survey map of sampling site #6, Fergus County; the area encircled designates sampling location. (Source: Fergus County Soil Survey, 1988 Sheet #147)