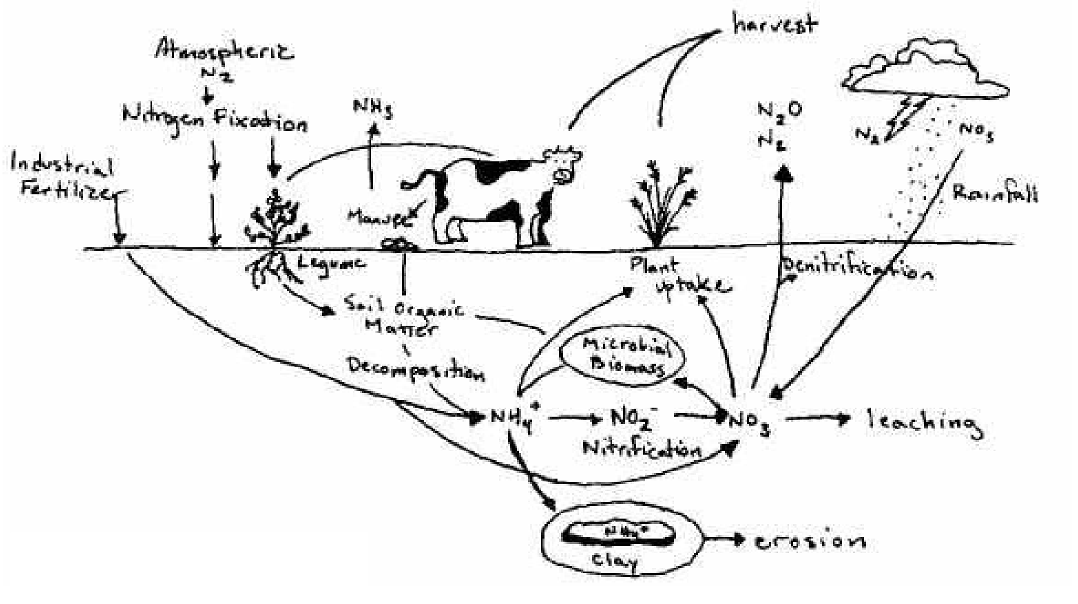
Evaluation of the Achilles' Heel of Mixed Grass Prairie Ecosystems: the Availability of Mineral Nitrogen



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Biogeochemical Processes of Nitrogen in Rangeland Soils

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Native rangelands of the Northern Plains are deficient in available soil mineral nitrogen (Power and Alessi 1971; Wight and Black 1972, 1979; Goetz 1984; Tilman 1990). The widespread deficient quantities of mineral nitrogen in the soils of mixed grass prairie pastures is the major cause for grass herbage production and calf weight gains to be at less than potential levels.

Wight and Black (1972,1979) evaluated herbage yield, plant species composition, and precipitation (water) use efficiency of mixed grass prairie at various quantities of available soil mineral nitrogen over a ten year period that had average annual precipitation at 13% above the long-term mean. They concluded that nitrogen was a major growth limiting factor in the Northern Plains, that increasing herbage biomass production to biological potential levels on rangeland ecosystems would require sustained mineralization rates that supplied 100 to 165 pounds of available mineral nitrogen per acre per year, and that the inhibitory deficiencies of mineral nitrogen on rangelands that had less than 100 lbs/ac of available soil mineral nitrogen caused the weight of herbage production per inch of precipitation received to be reduced by an average of 49.6% below the weight of herbage produced per inch of precipitation on the rangeland ecosystems that had greater than 100 lbs/ac of mineral nitrogen and had no mineral nitrogen deficiencies.

Manske (2012a) documented quantities of available mineral nitrogen that ranged from 31.2 lbs/ac to 76.7 lbs/ac on five traditional management treatments with operational histories of 20 to 75 years. These low quantities of available mineral nitrogen that are well below the threshold level of 100 lbs/ac indicate that long-term traditional grazing management and long-term nongrazing practices result in rangeland ecosystems with considerable inhibitory mineral nitrogen deficiencies (table 1).

The symptoms of low herbage biomass production, deterioration of plant density and species composition, and reduced livestock weight performance caused by deficient quantities of mineral nitrogen in pasture soils are easily observable. Livestock producers typically treat these symptoms with costly practices of feeding creep to the calves and providing supplemental crude protein to the cows. Supplemental amendments of nitrogen through agronomic practices of fertilization and alfalfa interseeding have been typically used to increase soil mineral nitrogen. Extensive research has found that these agronomic practices are not successful solutions and that they actually cause additional long lasting problems (Manske et al. 2005, 2014c). Treating the symptoms has not corrected the problems.

Deficiencies of soil mineral nitrogen in rangeland soils are not the result of low quantities of nitrogen. About 78% of the atmospheric volume consists of dinitrogen gas (N_2). The column of air above an acre of land contains about 34,500 tons of dinitrogen gas (Foth 1978). Dinitrogen gas in the atmosphere is not directly available to higher plants. Atmospheric dinitrogen can be fixed by a few types of bacteria and by lightning and then moved into the soil. Lightning discharges combine dinitrogen (N_2) and oxygen (O_2) to produce nitric acid (NO) and dinitrogen oxide (N_2O) that are deposited on the land in precipitation at a rate around 5 to 6 pounds per acre per year in temperate regions (Brady1974, Gibson 2009). Gaseous nitrogen oxides [nitric acid (NO) and dinitrogen oxide (N_2O)] and ammonia (NH_3) occur in the atmosphere and can be absorbed directly by leaves of higher plants through stomata (Coyne et al. 1995) and are eventually moved into the soil.

Rangeland soils of the Northern Plains have been accumulating nitogen at a few pounds per acre per year for about 5000 years since the current climate started (Bluemle 2000, Manske 2008). Most prairie soils contain 5 to 6 tons of nitrogen per acre. The nitrogen in rangeland soils is primarily in the organic form and not available to plants. A very large portion of the soil organic nitrogen is contained in the soil organic matter detritus at various stages of physical breakdown and decomposition (Legg 1975). Some of the organic nitrogen is immobilized in living tissue of plants, microorganisms, and animals as essential constituents of proteins and nucleotides. A small amount of the soil nitrogen is in the mineral form as ammonium (NH_4) and nitrate (NO_3) with an immeasurably minute amount as nitrite (NO_2). Annually, the amount of nitrogen entering a prairie ecosystem through wet and dry deposition or leaving an ecosystem as livestock weight is exceedingly small compared with the amounts held as organic forms in plants and microbes or in soil organic matter (Coyne et al. 1995).

Rangelands of the Northern Plains are not inherently low producing ecosystems. Typical low grass herbage production and calf weight gains on rangeland pastures that result from deficient quantities of available soil mineral nitrogen is strictly a management caused problem. Development of management solutions requires a working knowledge of the biogeochemical processes of nitrogen in rangeland ecosystems.

Biogeochemical Processes

The nitrogen cycle within rangeland soils functions around two major biogeochemical processes: immobilization and mineralization. These processes take place simultaneously with plant growth, dieback, and microbial decomposition (Legg 1975). Immobilization is the process of assimilation of mineral nitrogen into organic forms of living organisms. Mineralization is the process of converting organic nitrogen into mineral (inorganic) nitrogen. The quantity of mineral nitrogen in a soil is the net difference between the total quantity of organic nitrogen mineralized by soil microorganisms and the quantity of mineral nitrogen immobilized by plants and soil microbes (Brady 1974, Legg 1975).

Immobilization of nitrogen is a biological process by which plants and soil microorganisms can assimilate absorbed inorganic nitrogen into essential organic nitrogen compounds. Ammonium (NH_4) and nitrate (NO_3) are the two most important forms of available mineral nitrogen in soils and both are readily absorbed by grassland plant roots and soil microorganisms. At low temperatures, ammonium uptake is greater than nitrate uptake (Coyne et al. 1995).

Nitrogen uptake of both ammonium and nitrate directly into grassland plant roots requires energy and takes place during the daytime while plants are conducting photosynthesis and a source of energy is readily available (Coyne et al. 1995). In prairie soils where ammonium is the major nitrogen source, mycorrhizal symbiotic fungi enhance nitrogen uptake and reduce the energy cost to host plants (Coyne et al. 1995). Endomycorrhizal fungi can absorb both ammonia (NH_3) and ammonium (NH_4) and pass these nitrogen forms directly into the host plant, or the fungi can nitrify these forms of nitrogen into nitrate (NO_3) and then transport it into the roots of the host plant (Moorman and Reeves 1979, Harley and Smith 1983, Allen and Allen 1990, Box and Hammond 1990, Marschner 1992, Manske 1996).

Absorbed ammonium must have immediate assimilation with carbon upon entering root tissue and then be converted into amino acids (Coyne et al. 1995). Absorbed nitrates can be reduced to ammonia or ammonium and then converted to amino acids in the roots, transported to the shoots or leaves for assimilation into amino acids or stored in cell vacuoles for later use (Coyne et al. 1995). The greatest quantity of nitrate use occurs during the vegetative growth stage until just past the flower stage (Brady 1974).

The recently formed amino acids can be used immediately to build complex nitrogenous compounds, or the amino acids can float around inside the plant for later use (Coyne et al. 1995). Amino acids are building blocks for proteins, nucleotides, and chlorophyll. Proteins are used to form enzymes, hormones, and structural components of cells. Nucleotides build nucleic acids, deoxyribonucleic acid (DNA) and ribonucleic acid (RNA), that are the genetic material that control all cellular functions and heredity (Coyne et al. 1995). About half of the organic nitrogen is in the form of amino compounds (Brady 1974).

The nitrogen assimilated into complex organic compounds is immobilized and thus protected from loss by leaching or volatilization (Legg 1975, Gibson 2009). Organic nitrogen is tied up in living tissue and in soil organic matter for three to four years in northern rangeland soils (Power 1972) and is unavailable for use by higher plants until after it has been mineralized by soil microorganisms.

Mineralization is a complex biogeochemical process conducted by a large number of saprotrophic and heterotrophic soil microorganisms that convert immobilized organic nitrogen from soil organic matter detritus and from living tissue of plant roots or other soil microbes into mineral (inorganic) nitrogen (Power 1972). Ammonium salts are the first inorganic nitrogen compounds produced by microbial digestion. Complex proteins and other organic nitrogen compounds are simplified by enzymatic digestion that hydrolyze the peptide bonds and liberate and degrade the amino acids by deamination to produce ammonia

(NH_3) and carbon dioxide, or other low molecular weight carbon compounds (Power 1972, Brady 1974). Most of the ammonia released into soil water is readily hydrolyzed into stable ammonium (NH_4).

The ammonium ions (NH_4+) are fairly immoble but have several optional biological and chemical pathways; the cations can be absorbed directly by plant roots, absorbed by soil microbes or symbiotic fungi, adsorbed to clay lattice structure, attached to organic matter or mineral soil, or be oxidized during nitrification producing nitrite (NO_2) and then nitrate (NO_3) (Brady 1974, Legg 1975, Coyne et al. 1995).

The nitrate ions (NO_{3-}) are repelled by soil particles and considered to be mobile moving freely in the soil with water (Coyne et al. 1995). The quantity of available nitrate in soil increases when the soil moisture content increases (Brady 1974).

Some forms of mineral nitrogen are fairly active and can be lost from the soil. In dry soils low in hydrogen ions, some of the gaseous ammonia (NH_3) near the soil surface that has not been hydrolyzed are volatilized into the atmosphere. In mesic regions where soil water can move below the rooting depth, some of the nitrate (NO_3) in solution can be lost by leaching (Legg 1975, Gibson 2009). However, none of the mineral nitrogen in the Northern Plains rangelands is lost by hydrologic leaching through the soil profile (Power 1970) because very little water moves below the three foot soil depth and water loss by leaching is low or nonexistent in the arid and semiarid regions of the short grass and mixed grass prairies that are covered with perennial vegetation (Brady 1974, Wight and Black 1979). Leaching losses of mineral nitrogen in the sub humid and humid regions of the mixed grass and tall grass prairies are negligible (Brady 1974, Coyne et al. 1995).

Burning of rangelands by prescribed fire and wild fire causes volatilization of the nitrogen contained in the aboveground herbage and litter. Combustion causes nitrogen losses approaching 90% primarily as ammonia (NH_3), dinitrogen oxide (N_2O), and other nitrogen oxides (Russelle 1992). Little belowground nitrogen is volatilized when soil is moist during the burn, however, when soil is dry, belowground temperatures can increase enough to denature protein, killing portions of the grass crowns and root material and volatilizing some belowground nitrogen.

The quantity of available ammonium in grassland ecosystems is dependant on the rate of mineralization of soil organic nitrogen (Coleman et al. 1983). The mineralization rate is determined by the microorganism biomass, and the microorganism biomass is limited by access to simple carbohydrate energy (Curl and Truelove 1986).

The available energy from soil organic matter is inadequate to increase soil microorganism biomass substantially. Soil organic matter (SOM) contains energy at a rate of about 4-5 kilocalories/gram which would translate into nearly 200 million kilocalories of energy in the top six inches of an acre of soil (Brady 1974). Most of this energy is lost during microbial decomposition and dissipated from the soil as heat. The small amount of energy available to soil organisms in fresh organic material comes from short chain carbohydrates of sugars and starches (1-5%) and from water soluble proteins (very low %) (Brady 1974).

Manipulation of the mineralization rates with grazing management will require increases in available energy from plant sources in order to increase the soil microorganism biomass. Grassland plants exudate substances through the roots into the surrounding soil. Root exudates include sugars, amino acids, proteins, and numerous carbon compounds (Coyne et al. 1995). The quantity of exudates differ with plant species and is variable with phenological growth stage. The quantity of root exudate leakage from ungrazed grassland plants support only a small microorganism biomass that are capable of mineralizing low amounts of mineral nitrogen (Manske 2012b). Rangeland ecosystems require a minimum of 100 lbs/ac of available mineral nitrogen to produce herbage biomass at the biological potential rates (Wight and Black 1972).

Partial defoliation of grass tillers at vegetative phenological growth stages by large grazing graminivores causes greater quantities of exudates containing simple carbohydrate energy to be released from the grass tillers through the roots into the rhizosphere (Hamilton and Frank 2001, Manske 2011a). With the increase in availability of energy from carbon compounds in the rhizosphere, the biomass and activity of the microorganisms increases (Anderson et al. 1981, Curl and Truelove 1986, Whipps 1990). The increase in rhizosphere organism biomass and activity results in greater rates of mineralization of soil organic nitrogen transforming into greater quantities of available mineral nitrogen (Coleman et al. 1983, Klein et al. 1988, Burrows and Pfleger 2002, Rillig et al. 2002, Bird et al. 2002, Driver et al. 2005). Increasing available mineral nitrogen in rangeland soils with grazing management strategies will require the participation of large grazing graminivores, grass plants, and rhizosphere microorganisms and the activation of the defoliation resistance mechanisms.

Defoliation Resistance Mechanisms

Grassland ecosystems developed as a result of the global climate cooling near the end of the Eocene epoch around 34 mya that reduced the forest ecosystems to open savannah ecosystems then to grassland ecosystems (Chintauan-Marquier et al. 2011, Gomez et al. 2012, Anissimov 2013, Anonymous 2013). During the period of 30 to 20 mya, modern native grasses, rhizosphere microorganisms, and large grazing mammalian graminivores, coevolved and developed complex interactive processes that improved mutual survival of the organisms and perpetuation of the grassland ecosystems.

The complex interrelationships amoung grass plants, soil microbes, and graminivores are symbiotic. The grazing graminivores depend on grass plants for nutritious forage. Grass plants depend on rhizosphere organisms for mineralization of essential elements, primarily nitrogen, from the soil organic matter. The main sources of soil organic matter are grazing animal waste, dead plant material, and soil microbe remains. Rhizosphere organisms depend on grass plants for energy in the form of short carbon chains. Grass plants exudate short carbon chain energy through the roots into the rhizosphere following partial defoliation by grazing graminivores of the aboveground leaf material at vegetative phenological growth stages. Grass plants produce double the leaf biomass than is needed by the plant in order to provide nutritious leaf forage for grazing graminivores.

The complex interactions that occur in grassland ecosystems that help grass tillers withstand and recover from partial defoliation by grazing are the defoliation resistance mechanisms (McNaughton 1979, 1983; Briske 1991; Briske and Richards 1994, 1995; Manske 1999). The three primary mechanisms are: compensatory internal physiological processes (McNaughton 1979, 1983; Briske 1991); asexual internal processes of vegetative tiller production (Mueller and Richards 1986, Richards et al. 1988, Murphy and Briske 1992, Briske and Richards 1994, 1995); and external symbiotic rhizosphere organism activity (Coleman et al. 1983, Ingham et al. 1985). These processes have been reviewed and described by Manske (2011b) and are summarized in the following paragraphs.

The compensatory internal physiological processes increase the restoration of biological and physiological processes enabling rapid and complete recovery of plant biomass in partially defoliated grass tillers (Langer 1972, Briske and Richards 1995). The growth rates of replacement leaves and shoots increase producing larger leaves with greater mass (Langer 1972, Briske and Richards 1995). Photosynthetic capacity increases in remaining mature leaves and rejuvenated portions of older leaves not completely senescent (Atkinson 1986, Briske and Richards 1995). Allocation of the carbon recently fixed by photosynthesis in remaining rejuvenated mature leaves and the nitrogen recently mineralized from soil organic nitrogen by active rhizosphere organisms increases and moved to active growing points (Richards and Caldwell 1985, Briske and Richards 1995, Coyne et al. 1995). Water (precipitation) use efficiency with increased herbage biomass production improves (Smika et al. 1965; Wight and Black 1972, 1979; Whitman 1976, 1978).

Asexual internal processes of vegetative tiller development from axillary buds is the dominant form of reproduction in semiarid and mesic grasslands (short grass, mixed grass, and tall grass prairies) (Belsky 1992, Chapman and Peat 1992, Briske and Richards 1995, Chapman 1996, Manske 1999) not sexual reproduction and the development of seedlings. Secondary tiller development from axillary buds increases (Moser 1977, Dahl and Hyder 1977, Dahl 1995). Initiated tiller density increases with the suppression of the inhibiting hormone, auxin (Murphy and Briske 1992, Briske and Richards 1994, 1995).

External symbiotic rhizosphere organism activity is absolutely necessary for the recycling of the essential elements (Coleman et al. 1983, Ingham et al. 1985). Mineralization of essential elements increases (Coleman et al. 1983, Klein et al. 1988); the macronutrients, nitrogen (N), phosphorus (P), and sulfur (S) are in the form of organic compounds in soil organic matter and are mineralized by soil microbes, and potassium (K), calcium (Ca), and magnesium (Mg) are inorganic cations adsorbed to organic particles in soil organic matter. Ecosystem biogeochemical cycling of essential elements, carbon (C), hydrogen (H), oxygen (O), and the micronutrients, renews the nutrient flow activities in ecosystem soils (Coleman et al. 1983, Klein et al. 1988). Belowground resource uptake competitiveness of grass plants improves (Li and

Wilson 1998, Kochy 1999, Kochy and Wilson 2000, Peltzer and Kochy 2001).

Activation of the Defoliation Resistance Mechanisms

Activation of the defoliation resistance mechanisms requires a complex assemblage of biogeochemical processes that involve intricate interactions among grass plants, rhizosphere microorganisms, and large grazing graminivores (Manske 1999)

Healthy grass plants capture and fix carbon from atmospheric carbon dioxide during photosynthesis that combines carbon, hydrogen, and oxygen to produce carbohydrates in quantities greater than the amount needed for tiller growth and development (Coyne et al. 1995). The surplus short chain carbon compounds are available to supply the energy needed by the rhizosphere microorganisms. The only time in which the surplus carbon energy can be moved from the grass tiller through the roots into the rhizosphere is while a tiller is in the vegetative growth stage. During vegetative growth, the aboveground foliage consists primarily of crude protein (nitrogen) and water; most of the carbon is still in the belowground parts. Partial defoliation of the aboveground vegetative leaves removes more nitrogen than carbon from the plant and disrupts the tillers C: N ratio forcing that tiller to release (exudate) some of its carbon into the rhizosphere.

Partial defoliation of tillers before the third new leaf stage, when the plants are low in carbohydrates, results in reduced growth rates of herbage production for the remainder of the growing season (Campbell 1952, Rogler et al. 1962, Manske 2000b) because the amount of photosynthetic product synthesized by the small leaf area of remaining early growth leaves is insufficient to meet the requirements for new leaf growth (Heady 1975, Coyne et al. 1995, Manske 1994, 2000b). After the tiller has produced three and half new leaves, the leaf area is large enough to synthesize photosynthate at sufficient quantities to meet leaf growth requirements (Manske 2011b).

Almost all grass tillers live for two growing seasons, tillers produce vegetative growth during the first growing season and during the second growing season, the lead tillers develop flower stalks (Manske 2014a). As the vegetative leaves mature, complex structural carbon compounds, cellulose and hemicellulose, increase and lignin forms in the cell walls (Manske 2011b). Partial defoliation after mid July no longer disrupts the C: N ratio and carbon energy is not forced out of the grass tillers into the rhizosphere. The period during which the defoliation resistance mechanism can be activated with partial defoliation by large grazing graminivores that causes surplus carbon energy to be moved from the tiller into the rhizosphere is short; for cool and warm season native grasses, the combined time is 45 days from 1 June to 15 July (the 3.5 new leaf stage to the flower stage), and for the major domesticated cool season grasses, the time is 40 days from 1 May to 10 June (also the 3.5 new leaf stage to the flower stage) (Manske 2011b).

Maintaining functionality of the activated mechanisms has a high carbon and nitrogen demand at the increased quantity of active growing points. The source of the carbon is preferentially allocated from the carbon recently fixed by photosynthesis in the remaining rejuvenated older leaves and the source of the nitrogen is preferentially allocated from the nitrogen recently mineralized from soil organic nitrogen by rhizosphere microorganisms (Ryle and Powell 1975, Richards and Caldwell 1985, Briske and Richards 1995). The variable quantities of available mineral nitrogen and carbon that can be used for rapid restoration of lost plant biomass regulates the variable levels of performance of the defoliation resistance mechanisms (Manske 2010b).

Wight and Black (1972, 1979) found that a minimum threshold quantity of 100 lbs/ac of available mineral nitrogen was required to fully activate the water (precipitation) use efficiency processes that enable a sustained herbage biomass production at biological potential levels on mixed grass prairie. Rangelands that have less than 100 lbs/ac mineral nitrogen have nitrogen deficiencies that cause the weight of herbage production per inch of precipitation received to be reduced an average of 49.6%. Mineralization at high rates that supply mineral nitrogen at 100 lbs/ac can not be obtained from traditional grazing practices (Wight and Black 1972, Manske 2012a, table 1).

Manske (2010a, 2010b) found that partial defoliation by large grazing graminivores of grass tillers at phenological growth stages between the three and a half new leaf stage and the flower (anthesis) stage activated the compensatory physiological processes and activated the asexual processes of vegetative tiller production on rangeland pastures that had 100 lbs/ac or greater of available mineral nitrogen. However, this same defoliation treatment did not activate the defoliation resistance mechanisms of grass plants on rangeland ecosystems that had soil mineral nitrogen available at quantities of less than 100 lbs/ac.

The defoliation resistance mechanisms do not function automatically; they require annual activation by partial defoliation of tillers at the vegetative growth stages between the three and a half new leaf stage and the flower stage; and the mechanisms do not function unless 100 lbs/ac of mineral nitrogen is available from the rhizosphere microorganisms (Wight and Black 1972, 1979, Manske 2010a, 2010b, 2014a, 2014b). Activation of the defoliation resistance mechanisms requires that the rhizosphere microbe biomass be increased to the level that is capable of mineralizing nitrogen at a high rate of 100 lbs/ac or greater. Rangelands in typical condition that have had a history of management of strong land stewardship ethics with traditional grazing practices requires two to three growing seasons to increase the soil microbe biomass; rangelands in poorer condition require five to seven years; and rangelands with histories of winter grazing or nongrazing require more than ten years to increase the soil microbe biomass sufficiently (Manske 2011b, 2012b).

Crider (1955) conducted an extensive study that found that grass tillers with 50% or more of the aboveground leaf material removed reduce root growth, root respiration, and root nutrient absorption. These physiological functions of grass plants have recently been described as the process of “belowground resource uptake competitiveness” that were documented by Kochy and Wilson (2000). Crider’s work led to the familiar range axiom “take half and leave half” which is still sound advice for grazing periods after peak herbage biomass has been reached during the last two weeks of July (Manske 2000a).

Removal of 50% of the leaf weight of vegetative grass tillers between the 3.5 new leaf stage and the flower stage did not activate the compensatory physiological processes nor the asexual processes of vegetative production of tillers from axillary buds regardless of whether the available mineral nitrogen was above or below the threshold of 100 lbs/ac. The remaining 50% leaf material had insufficient leaf area to provide adequate quantities of fixed carbon for grass plant restoration. Because of the deficient availability of carbon, the weight of the leaf biomass removed by defoliation was not replaced by compensatory processes and fewer vegetative tillers were produced than that developed on the ungrazed control treatment (Manske 2010a, 2010b, 2014a, 2014b).

Removal of 25% of the leaf material from grass tillers between the three and a half new leaf stage and the flower stage caused an increase in the quantities of exudates containing simple carbon energy to be released from partially defoliated grass tillers through the roots into the rhizosphere to be great enough to significantly increase the rhizosphere volume in a grazing treatment study (Gorder, Manske, Stroh 2004) and to significantly increase the rhizosphere weight by the third year after initiation of the activation treatment in a prairie restoration study (Manske 2013).

Removal of 25% of the leaf material by partial defoliation of grass tillers between the three and a half new leaf stage and the flower stage fully activated the compensatory physiological processes and fully activated the asexual processes of vegetative tiller production when 100 lbs/ac or greater of mineral nitrogen was available. The weight of the leaf biomass removed by partial defoliation was replaced with new leaf growth at a rate of 140% of the lost weight. The quantity of vegetative tiller production increased at a rate of 214% greater tiller density/m^2 (Manske 2010a, 2010b, 2014a, 2014b). The remaining 75% leaf material of rejuvenated older leaves had sufficient leaf area to fix carbon at adequate quantities for compensatory growth and development of the replacement leaves and shoots. Removal of 25% of the leaf weight during vegetative growth stages also removed sufficient quantities of the growth-inhibiting hormone, auxin, permitting synthesis or utilization of the growth hormone, cytokinin, in the axillary buds and activated growth and development of vegetative tillers from a high percentage of the axillary buds (Manske 2011b).

Activation of the defoliation resistance mechanisms; the compensatory physiological processes and coprocesses, and the asexual processes of vegetative tillering and coprocesses; requires grazing management strategies that specifically include annual activation with partial defoliation by large grazing graminivores that removes 25% to 33% of the aboveground leaf and shoot weight from grass tillers in vegetative phenological growth between the three and a half new leaf stage and the flower stage after 100 lbs/ac of available mineral nitrogen has been mineralized annually by the rhizosphere microorganism biomass that has been enhanced over two or three growing seasons by increased exudate release containing short chain carbon energy from vegetative grass tillers by partial defoliation and after fixed carbon is available in large enough quantities from the leaf area of rejuvenated tillers equal to 75% to 67% of the ungrazed vegetative tiller leaf area.

Summary

Many of the problems found on rangeland pastures are the result of deficiencies in the amount of available mineral nitrogen in the soil. Nongrazing and traditional grazing practices are the origins of these management caused problems.

Rangeland soils are actually not deficient in nitrogen; they contain large quantities of organic nitrogen, but it is not available to higher plants. Nitrogen cycles within rangeland soils from the organic form to the mineral form and back to the organic form through biogeochemical processes: immobilization and mineralization. Immobilization of mineral nitrogen to organic nitrogen requires new growth of plants and microorganisms. Mineralization of organic nitrogen into mineral nitrogen requires soil microorganism decomposition.

These biogeochemical processes are integral to the complex mechanisms developed by grass plants that permit grass tillers to withstand and recover from partial defoliation by large graminivores. These mechanisms are the defoliation resistance mechanisms. The grass plants require large quantities of mineral nitrogen and fixed carbon to be available for the rapid production of new leaves and shoots and the development of vegetative tillers that replace the plant biomass lost by grazing.

Starting the functionality of the defoliation resistance mechanisms is not instantaneous. Adequate quantities of mineral nitrogen and fix carbon must be available before the mechanisms will start to function. Mineral nitrogen needs to be available at the threshold quantity of 100 lbs/ac or greater. Available mineral nitrogen at these high rates requires a great rhizosphere microorganism biomass that can be maintained through exudation of large quantities of short chain carbon energy from grass tillers that have had partial defoliation by large grazing graminivores while the tillers were at vegetative growth stages between the three and a half new leaf stage and the flower stage. Adequate quantities of available fixed carbon can be provided by the partially defoliated tillers when the combined leaf area of the remaining and rejuvenated leaves is 75% to 67% of the leaf area of ungrazed tillers at vegetative growth stages. This means that the quantity of leaf removal during the defoliation resistance mechanism activation treatment of partial defoliation by large grazing graminivores needs to be 25% to 33% of the vegetative tillers leaf weight. Rangelands with long-term management by traditional grazing practices with moderate stocking rates typically require two to three growing seasons to increase the rhizosphere microorganism biomass to levels that are capable of mineralizing available mineral nitrogen at rates of 100 lbs/ac. The defoliation resistance mechanisms can then be maintained at full functional levels through grazing management strategies that specifically include annual activation treatments of partial defoliation by large grazing graminivores that remove 25% to 33% of the aboveground leaf weight of grass tillers at phenological growth stages between the three and a half new leaf stage and the flower stage.

Stopping the functionality of the defoliation resistance mechanisms is also not instantaneous. When or if the annual activation treatments are terminated, the functional levels of the defoliation resistance mechanisms fade rapidly to ineffectiveness during the following two or three growing seasons.

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Table 1. Mineral nitrogen (lbs/ac) available on mixed grass prairie ecosystems effected by traditional management treatments.

Traditional Management Treatment	Operational Duration Years	Available Mineral Nitrogen lbs/ac
4.5 m Seasonlong	20 yr	76.7
6.0 m Seasonlong	20 yr	61.6
Deferred Grazing	50 yr	31.2
7.0 m Seasonlong	75 yr	42.4
Nongrazed	75 yr	39.5

Data from Manske 2012a.

Literature Cited

Allen, E.B., and M.F. Allen. 1990. The mediation of competition by mycorrhizae in successional and patchy environments. p. 307-389. *in* J.B. Grace and D. Tilman (eds.). Perspectives on plant competition. Academic Press Inc., San Diego, CA.

Anderson, R.V., D.C. Coleman, C.V. Cole, and E.T. Elliott. 1981. Effect of nematodes *Acrobeloides sp.* and *Mesodiplogaster lheritieri* on substrate utilization and nitrogen and phosphorus mineralization. Ecology 62:549-555.

Anissimov, M. 2013. What is the evolutionary history of insects? http://www.wisegeek.com.

Anonymous. 2013. Grasshopper. Wikimedia Foundation, Inc. http://www.en.wikipedia.org.

Atkinson, C.J. 1986. The effect of clipping on net photosynthesis and dark respiration rates of plants from an upland grassland, with reference to carbon partitioning in *Festuca ovina.* Annals of Botany 58:61-72.

Belsky, A.J. 1992. Effects of grazing competition, disturbance and fire on species composition and diversity in grassland communities. Journal of Vegetation Science 3:187-200.

Bird, S.B., J.E. Herrick, M.M. Wander, and S.F. Wright. 2002. Spatial heterogeneity of aggregate stability and soil carbon in semi-arid rangeland. Environmental Pollution 116:445-455.

Bluemle, J.P. 2000. The face of North Dakota. 3rd edition. North Dakota Geological Survey. Ed. Series 26. 206p. 1pl.

Box, J.E., and L.C. Hammond. 1990. Rhizosphere dynamics. Westview Press, Boulder, CO.

Brady, N.C. 1974. The nature and properties of soils. MacMillan Publishing Co., Inc., New York, NY. 639p.

Briske, D.D. 1991. Developmental morphology and physiology of grasses. p. 85-108. *in* R.K. Heitschmidt and J.W. Stuth (eds.). Grazing management: an ecological perspective. Timber Press, Portland, OR.

Briske, D.D., and J.H. Richards. 1994. Physiological responses of individual plants to grazing: current status and ecological significance. p. 147-176. *in* M. Vavra, W.A. Laycock, and R.D. Pieper (eds.). Ecological implications of livestock herbivory in the west. Society for Range Management, Denver, CO.

Briske, D.D., and J.H. Richards. 1995. Plant response to defoliation: a physiological, morphological, and demographic evaluation. p. 635-710. *in* D.J. Bedunah and R.E. Sosebee (eds.). Wildland plants: physiological ecology and developmental morphology. Society for Range Management, Denver, CO.

Burrows, R.L., and F.L. Pfleger. 2002. Arbuscular mycorrhizal fungi respond to increasing plant diversity. Canadian Journal of Botany 80:120-130.

Campbell, J.B. 1952. Farming range pastures. Journal of Range Management 5:252-258.

Chapman, G.P., and W.E. Peat. 1992. An introduction to the grasses. C.A.B. International, Wallingford, UK. 111p.

Chapman, G.P. 1996. The biology of grasses. C.A.B. International, Wallingford, UK. 273p.

Chintauan-Marquier, I.C., S. Jordan, P. Berthier, and C. Amedegnata. 2011. Evolutionary history and taxonomy of a shorthorned grasshopper subfamily: The Melanoplinae (Orthoptera: Acrididae). Molecular Phylogenetics and Evolution 58:22-32.

Coleman, D.C., C.P.P. Reid, and C.V. Cole. 1983. Biological strategies of nutrient cycling in soil ecosystems. Advances in Ecological Research 13:1-55.

Coyne, P.I., M.J. Trlica, and C.E. Owensby. 1995. Carbon and nitrogen dynamics in range plants. p. 59-167. *in* D.J. Bedunah and R.E. Sosebee (eds.). Wildland plants: physiological ecology and developmental morphology. Society for Range Management, Denver, CO.

Crider, F.J. 1955. Root-growth stoppage resulting from defoliation of grass. USDA Technical Bulletin 1102.

Curl, E.A., and B. Truelove. 1986. The rhizosphere. Springer-Verlag, New York, NY.

Dahl, B.E., and D.N. Hyder. 1977. Developmental morphology and management implications. p. 257-290. *in* R.E. Sosebee (ed.). Rangeland plant physiology. Range Science Series No. 4. Society for Range Management, Denver, CO.

Dahl, B.E. 1995. Developmental morphology of plants. p. 22-58. *in* D.J. Bedunah and R.E. Sosebee (eds.). Wildland plants: physiological ecology and developmental morphology. Society for Range Management, Denver, CO.

Driver, J.D., W.E. Holben, and M.C. Rillig. 2005. Characterization of glomalin as a hyphal wall component of arbuscular mycorrhizal fungi. Soil Biology and Biochemistry 37:101-106.

Foth, H.D. 1978. Fundamentals of soil science. John Wiley and Sons, New York, NY. 436p.

Gibson, D.J. 2009. Grasses and grassland ecology. Oxford University Press Inc., New York, NY. 305p.

Goetz, H. 1984. A synopsis of rangeland fertilization in western North Dakota. p. 17-27. *in* Proceedings of North Dakota Chapter of the Society for Range Management, 1983. Dickinson, ND.

Gomez. R.A., D.C. Lightfoot, and K.B. Miller. 2012. A phylogenetic review of the North American bandwinged grasshopper genus, *Encoptolophus* Scudder with description of *Nebulatettix* gen. n. (Orthoptera: Acrididae: Oedipodinae). Insect Systematics and Evolution 43:117-145.

Gorder, M.M., L.L. Manske, and T.L. Stroh. 2004. Grazing treatment effects on vegetative tillering and soil rhizospheres of western wheatgrass. NDSU Dickinson Research Extension Center. Range Research Report DREC 04-1056. Dickinson, ND. 13p.

Hamilton, E.W., and D.A. Frank. 2001. Can plants stimulate soil microbes and their own nutrient supply? Evidence from a grazing tolerant grass. Ecology 82:2397-2402.

Harley, J.L., and S.E. Smith. 1983. Mycorrhizal symbiosis. Academic Press, New York, NY.

Heady, H.F. 1975. Rangeland management. McGraw-Hill Book Company, New York, NY.

Ingham, R.E., J.A. Trofymow, E.R. Ingham, and D.C. Coleman. 1985. Interactions of bacteria, fungi, and the nemotode grazers: effects of nutrient cycling and plant growth. Ecological Monographs 55:119-140.

Klein, D.A., B.A. Frederick, M. Biondini, and M.J. Trlica. 1988. Rhizosphere microorganism effects on soluble amino acids, sugars, and organic acids in the root zone of *Agropyron cristatum, A. smithii,* and *Bouteloua gracilis*. Plant and Soil 110:19-25.

Kochy, M. 1999. Grass-tree interactions in western Canada. Ph.D. Dissertation. University of Regina. Regina, Saskatchewan, Canada.

Kochy, M., and S.D. Wilson. 2000. Competitive effects of shrubs and grasses in prairie. Oikos 91:385-395.

Langer, R.H.M. 1972. How grasses grow. Edward Arnold, London, Great Britain.

Legg, J.O. 1975. Influence of plants on nitrogen transformation in soils. pg. 221-227. *in* M.K. Wali (ed.). Prairie: A multiple view. University of North Dakota Press. Grand Forks, ND.

Li, X., and S.D. Wilson. 1998. Facilitation among woody plants establishing in an old field. Ecology 79:2694-2705.

Manske, L.L. 1994. Problems to consider when implementing grazing management practices in the Northern Great Plains. NDSU Dickinson Research Extension Center. Range Management Report DREC 94-1005. Dickinson, ND. 11p.

Manske, L.L. 1996. Adaptive tolerance mechanisms in grass plants. p. 97-99. *in Z.* Abouguendia (ed.). Total ranch management in the Northern Great Plains. Grazing and Pasture Technology Program, Saskatchewan Agriculture and Food. Regina, Saskatchewan, Canada.

Manske, L.L. 1999. Can native prairie be sustained under livestock grazing? Provincial Museum of Alberta. Natural History Occasional Paper No. 24. Edmonton, Alberta. p.99-108.

Manske, L.L. 2000a. Grass growth in height. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 00-3020. Dickinson, ND. 4p.

Manske, L.L. 2000b. Grazing before grass is ready. NDSU Dickinson Research Extension Center. Range Management Report DREC 00-1032. Dickinson, ND. 6p.

Manske, L.L. 2005. Evaluation of alfalfa interseeding techniques. NDSU Dickinson Research Extension Center. Rangeland Research Extension Program 4008. 140p.

Manske, L.L. 2008. Prehistorical conditions of rangelands in the Northern Plains. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 08-3015b. Dickinson, ND. 5p.

Manske, L.L. 2010a. Leaf stage development of western wheatgrass tillers. NDSU Dickinson Research Extension Center. Range Research Report DREC 10-1075. Dickinson, ND. 48p.

Manske, L.L. 2010b. Evaluation of the defoliation resistance mechanisms influence on vegetative tiller initiation and tiller density. NDSU Dickinson Research Extension Center. Range Research Report DREC 10-1076. Dickinson, ND. 13p.

Manske, L.L. 2011a. Soil mineral nitrogen increased above the threshold quantity of 100 pounds per acre in rangeland ecosystems. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 11-3056. Dickinson, ND. 8p.

Manske, L.L. 2011b. Biology of defoliation by grazing. NDSU Dickinson Research Extension Center. Range Management Report DREC 11-1067b. Dickinson, ND. 25p.

Manske, L.L. 2012a. Degradation and biological restoration of mixed grass prairie ecosystems. NDSU Dickinson Research Extension Center. Summary Range Research Report DREC 12-3058. Dickinson, ND. 16p.

Manske, L.L., and S.A. Schneider. 2012b. Evaluation of biological restoration management of degraded native mixed grass prairie. NDSU Dickinson Research Extension Center. Rangeland Research Outreach Program 4017. 83p. and Appendix.

Manske, L.L. 2013. Initial restoration changes of degraded rangeland with the twice-over rotation grazing strategy. NDSU Dickinson Research Extension Center. Range Research Report DREC 13-1080. Dickinson, ND. 24p.

Manske, L.L. 2014a. Grass vegetative tillering responses to partial defoliation. NDSU Dickinson Research Extension Center. Range Research Report DREC 14-1086. Dickinson, ND. 35p.

Manske, L.L. 2014b. Vegetative forage tiller development in response to partial defoliation. NDSU Dickinson Research Extension Center. Range Research Report DREC 14-1087. Dickinson, ND. 26p.

Manske, L.L., and S.A. Schneider. 2014c. Evaluation of nitrogen fertilization on native rangeland. 2nd Edition. NDSU Dickinson Research Extension Center. Rangeland Research Outreach Program DREC 14-4013b. Dickinson, ND. 193p.

Marschner, H. 1992. Nutrient dynamics at the soil-root interface (Rhizosphere). p. 3-12. *in* D.J. Read, D.H. Lewis, A.H. Fitter, and I.J. Alexander (eds.). Mycorrhizas in ecosystems. C.A.B. International, Wallingford, U.K.

McNaughton, S.J. 1979. Grazing as an optimization process: grass-ungulate relationships in the Serengeti. American Naturalist 113:691-703.

McNaughton, S.J. 1983. Compensatory plant growth as a response to herbivory. Oikos 40:329-336.

Moorman, T., and F.B. Reeves. 1979. The role of endomycorrhizae in revegetation practices in the semi-arid west. II. A bioassay to determine the effect of land disturbance on endomycorrhizal populations. American Journal of Botany 66:14-18.

Moser, L.E. 1977. Carbohydrate translocation in range plants. p. 47-71 *in* R.E. Sosebee (ed.). Rangeland plant physiology. Range Science Series No. 4. Society for Range Management, Denver, CO.

Mueller, R.J., and J.H. Richards. 1986. Morphological analysis of tillering in *Agropyron spicatum* and *Agropyron desertorum.* Annals of Botany 58:911-921.

Murphy, J.S., and D.D. Briske. 1992. Regulation of tillering by apical dominance: chronology, interpretive value, and current perspectives. Journal of Range Management 45:419-429.

Peltzer, D.A., and M. Kochy. 2001. Competitive effects of grasses and woody plants in mixed grass prairie. Journal of Ecology 89:519-527.

Power, J.F. 1970. Nitrogen management of semiarid grasslands in North America. Proceedings of the XI International Grassland Congress. 1970:468-471.

Power, J.F., and J. Alessi. 1971. Nitrogen fertilization of semiarid grasslands: plant growth and soil mineral N levels. Agronomy Journal 63:277-280.

Power, J.F. 1972. Fate of fertilizer nitrogen applied to a Northern Great Plains rangeland ecosystem. Journal of Range Management 25:367-371.

Richards, J.H., and M.M. Caldwell. 1985. Soluble carbohydrates, concurrent photosynthesis and efficiency in regrowth following defoliation: a field study with *Agropyron* species. Journal of Applied Ecology 22:907-920.

Richards, J.H., R.J. Mueller, and J.J. Mott. 1988. Tillering in tussock grasses in relation to defoliation and apical bud removal. Annals of Botany 62:173-179.

Rillig, M.C., S.F. Wright, and V.T. Eviner. 2002. The role of arbuscular mycorrhizal fungi and glomalin in soil aggregation: comparing effects of five plant species. Plant and Soil 238:325-333.

Rogler, G.A., R.J. Lorenz, and H.M. Schaaf. 1962. Progress with grass. North Dakota Agricultural Experiment Station. Bulletin 439. 15p.

Russelle, M.P. 1992. Nitrogen cycling in pastures and range. Journal of Production Agriculture 5:13-23.

Ryle, G.J., and C.E. Powell. 1975. Defoliation and regrowth in the graminaceous plant: the role of current assimilate. Annals of Botany 39:297-310.

Smika, D.E., H.J. Haas, and J.F. Power. 1965. Effects of moisture and nitrogen fertilizer on growth and water use by native grass. Agronomy Journal 57:483-486.

Tilman, D. 1990. Constraints and tradeoffs: toward a predictive theory of competition and succession. Oikos 58:3-15.

Whipps, J.M. 1990. Carbon economy. p. 59-97. *in* J.M. Lynch (ed.). The rhizosphere. John Wiley and Sons, New York, NY.

Whitman, W.C. 1976. Native range fertilization and interseeding studies. Annual Report. Dickinson Experiment Station. Dickinson, ND. p. 11-17.

Whitman, W.C. 1978. Fertilization of native mixed prairie in western North Dakota. Annual Report. Dickinson Experiment Station. Dickinson, ND. p. 20-22.

Wight, J.R., and A.L. Black. 1972. Energy fixation and precipitation use efficiency in a fertilized rangeland ecosystem of the Northern Great Plains. Journal of Range Management 25:376-380.

Wight, J.R., and A.L. Black. 1979. Range fertilization: plant response and water use. Journal of Range Management 32:345-349.

Evaluation of the Importance of Mineral Nitrogen in Restoration of Severely Degraded Mixed Grass Prairie Ecosystems

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Grass plants produce stems, leaves, and roots from vital organic compounds of carbohydrates, proteins, and nucleotides which are comprised of major essential elements, carbon, hydrogen, nitrogen, and oxygen and minor essential elements, macronutrients and micronutrients plus energy from sunlight. Procurement and cycling of the major and minor essential elements occurs through the numerous complex processes of the defoliation resistance mechanisms and biogeochemical processes that requires extensive interactions among the grass plants, soil microorganisms, and grazing graminivores.

Any factor that causes the ecosystem processes to not function properly causes the quantity of essential elements to decrease resulting in reduced ecosystem productivity. Low mineral nitrogen available at quantities less than the threshold of 100 lbs/ac is the major growth limiting factor of native grassland ecosystems and causes ecosystem degradation down to the level of available mineral nitrogen (Manske 1999, 2011, 2014). Removal of grazing graminivores results in severe degradation of native grassland ecosystems.

The objectives of this study are to show the importance of mineral nitrogen to be available at quantities of 100 lbs/ac or greater in restoration of degraded native grassland ecosystems.

Study Area

The native rangeland study sites were on the Schnell Recreation Area (SRA) managed by the USDI Bureau of Land Management (BLM) since 1993, and were located in eastern Stark County approximately 2 miles (3.22 kilometers) east of Richardton, North Dakota, USA.

The western North Dakota region near Richardton has cold winters and hot summers typical of continental climates. Long-term mean annual temperature was 43.0° F (6.1° C). January was the coldest month, with a mean of 13.5° F (-10.3° C). July and August were the warmest months, with mean temperatures of 70.0° F (21.1° C) and 68.9° F (20.5° C), respectively. Long-term (1971-2000) mean annual precipitation was 17.8 inches (451.6 mm). The amount of precipitation received during the perennial plant growing season (April to October) was 14.8 inches (375.7 mm) and was 83.2% of annual precipitation. The precipitation received in the three month period of May, June, and July was 8.2 inches (207.0 mm) and was 45.8% of the annual precipitation.

The native rangeland vegetation was the Wheatgrass-Needlegrass Type (Barker and Whitman 1988, Shiflet 1994) of the mixed grass prairie. The dominant native range grasses were western wheatgrass (Agropyron smithii) (Pascopyrum smithii), needle and thread (Stipa comata) (Hesperostipa comata), blue grama (Bouteloua gracilis), and threadleaf sedge (Carex filifolia).

Management Treatments

The study area was degraded native mixed grass prairie. The area was a working cattle ranch prior to 1993. The entire area was managed exclusively for recreation during the 13 year period of 1993 to 2005 with no cattle grazing permitted. The rangeland ecosystems became severely degenerated by the nondefoliation with complete rest management that was antagonistic to ecosystem biogeochemical processes reducing native plant density, opening plant communities to subsequent invasion by undesirable introduced cool season domesticated grasses, primarily smooth bromegrass, crested wheatgrass, and Kentucky bluegrass.

A study was conducted to evaluate the restoration of severely degraded native mixed grass prairie managed with the twice-over rotation grazing management strategy. Three grassland pastures were grazed from early June until mid October, with each pasture grazed for two periods. Each of the three pastures in the rotation were grazed for 14 to 16 days during the first period, the 45 day interval from 1 June to 15 July. During the second period, the 90 day interval from mid July to mid October, each pasture was grazed for double the number of days that it was grazed during the first period. A fourth pasture was not grazed and was used as a control. Nongrazed 4 was dominated by Kentucky bluegrass; Grazed pasture 3 was dominated by smooth bromegrass; and Grazed pastures 1 and 2 were dominated by Kentucky bluegrass; thus forming the 3 treatments of the study.

Procedure

Temperature and precipitation data were taken from climatological data collected at the Richardton Abbey, Stark County, latitude 46.88° N, longitude 102.31° W at 2467 feet (752 mm) above sea level, 2006-2011.

Grazing pressure determined by animal unit equivalent, herd weight, and stocking rate were assessed at the start of the study using the then current ecological site maps, and were assessed a second time using updated ecological site maps.

Restoration of degraded mixed grass prairie was evaluated on silty ecological sites with permanent sample plots organized in a paired-plot design. Two adjacent plots were at every site, each 16' X 32' (4.88 m X 9.75 m) in size with one grazed and the other ungrazed inside a stock panel exclosure.

Aboveground herbage biomass was collected by the standard clipping method (Cook and Stubbendieck 1986) at each pasture rotation date (seven periods per year). The herbage biomass was partially defoliated by the selected twice-over rotation grazing treatment on pasture 3 and pastures 1 & 2. The nongrazed 4 area had no defoliation treatments. The reported herbage biomass values represent the residuum vegetation and the regrowth vegetation resulting from the respective treatments. Clipped herbage material was collected from five 0.25 m^2 quadrats (frames) at silty ecological sample sites for each of the study treatments during the first study section, 2006 to 2011, and during the second study section, 2013 to 2014. The herbage material in each frame was hand clipped to ground level and sorted in the field by biotype categories: domesticated grass, cool season grass, warm season grass, sedges, forbs, standing dead, and litter. The herbage of each biotype category from each frame was placed in labeled paper bags of known weight, oven dried at 140° F (60° C), and weighed. Herbage biomass in pounds per acre for each category were determined from the clipping data. Domesticated grass and native grass (cool and warm season grass) herbage biomass weights were reported for this study.

Plant species basal cover for individual species were determined by the ten-pin point frame method (Cook and Stubbendieck 1986), with 2000 points collected along permanent transect lines at silty ecological sites for each of the study treatments annually during peak growth between mid July and mid August during the first study section, 2006 to 2011, and during the second study section, 2013 to 2014. Basal cover plant species data were sorted into biotype categories: domesticated grass, cool season grass, warm season grass, upland sedges, forbs, and litter. Domesticated grass and native grass (cool season and warm season grass) percent basal cover were reported for this study.

Rhizosphere biomass was collected on silty ecological sites for each of the study treatments during the first study section, 2006 to 2011. Sample areas had been grazed by the twice-over rotation treatment on pasture 3 and on pastures 1 & 2. The nongrazed 4 area had no defoliation treatments. Three replicated soil cores 3 inches (7.6 cm) in diameter and 4 inches (10.2 cm) in depth were collected at each study site during 3 grazing season periods: pregrazing (May), first rotation (July), and second rotation (October) using a humane soil beastie catcher (Manske and Urban 2012). The fresh rhizosphere material, which included the rhizosphere organisms, the active plant roots, and the adhered soil particles, was separated from matrix soil by meticulous excavation with fine hand tools. Both wet and dry rhizosphere weights were collected. Rhizosphere biomass per volume of soil was determined from the soil core rhizosphere weight data and reported as kilograms per cubic meter. Reference samples of rhizosphere weights on silty ecological sites managed long-term with a twice-over rotation grazing strategy were collected by the same methods during 2006 resulting in mean reference rhizosphere weights at 406.44 kg/m^3.

Soil mineral nitrogen, nitrate (NO_3) and ammonium NH_4), was determined from two replicated soil core samples collected at silty ecological sites outside exclosures exposed to the selected twice-over rotation grazing treatment on pasture 3 and on pastures 1 & 2. The nongrazed 4 area had no defoliation treatments. Soil cores were collected with the 1 inch Veihmeyer soil tube at incremental depths of 0-6, 6-12, and 12-24 inches on monthly periods during May to October of the second study section, 2013 to 2014. Analysis of soil core samples for available mineral nitrogen (NO_3 and NH_4) was conducted by the North Dakota State University Soil Testing Laboratory. Mean available mineral nitrogen was reported as pounds per acre.

Transformation (immobilization) of nitrate (NO_3) and of ammonium (NH_4) was determined by the net mineralization measurement of the nitrogen balance equation of a soil-plant system (Bloem et al. 2006). The quantity of mineral nitrogen in a soil is the net difference between the total quantity of organic nitrogen mineralized by soil microorganisms and the quantity of mineral nitrogen immobilized by plants and soil microbes (Brady 1974, Legg 1975). The general nitrogen balance equation is simply: the quantity of nitrogen at time 2 minus the quantity of nitrogen at time 1, the difference is the quantity of the transformed nitrogen. Nitrogen quantity at time 1 is the May values. Nitrogen quantity at time 2 is the values at each successive month. Transformed nitrogen is the quantity of uptake by plants and soil microbes and converted into organic nitrogen plus the quantity of nitrogen loss by leaching or volatilization. Loss by leaching on Northern Plains prairies is negligible (Power 1970, Brady 1974, Wight and Black 1979, Coyne et al. 1995). Loss by volatilization during 2013 and 2014 with high soil water content would also be negligible. Transformation of ammonium (NH_4) could include some conversion to nitrate (NO_3). Most of the transformed nitrogen would be the quantity converted into organic nitrogen by plants and soil microbes.

A standard t-test was used to analyze differences among means (Mosteller and Rourke 1973).

Results

Precipitation during the 2006 to 2011 grazing seasons at Richardton, ND was slightly below normal during 2006 and 2007, and was normal during 2008 to 2011. A dry period occurred during May through August 2006. High rainfall occurred during April, May, and July 2011. The mean six year growing season precipitation was 12.3 inches (83.2% of LTM) (table 1). Mean April through July precipitation was 82.8% of LTM and mean August through October precipitation was 84.1% of LTM. Generally, precipitation was normal but a little below average.

The stocking rate used to graze the three pastures of the Schnell Recreation Area was originally intended to be at 75%, 85%, and 95% of the assessed stocking rate during the first three years and then remain at less than 100% for the duration of the study. Cow-calf pairs grazed during the first three years at 72.3%, 82.6%, and 107.5%, with a mean of 87.4%, of the assessed stocking rate. As the cow herd stocking rates increased during the first three years, the relative composition of native grass herbage biomass and basal cover also increased on the grazed pastures. Heifers grazed during 2009 and 2010 at 79.8% and 87.4%, respectively, with a mean of 83.6% of the assessed stocking rate. The reduction in stocking rate of the heifers resulted in a decrease in the relative composition of native grass herbage biomass and basal cover. Steers grazed during 2011 at 37.8% of the assessed stocking rate (table 2). This low stocking rate resulted in a large decrease in the relative composition of native grass herbage biomass and basal cover. The light to moderate stocking rates were not beneficial for improvement of native grass herbage biomass and basal cover.

Degradation of the native grass plant community not only decreased grass plant basal cover but also decreased plant stature. The invading Kentucky bluegrass and smooth bromegrass increased in basal cover and plant stature causing greatly reduced sunlight intensities reaching understory native grass leaves. Getting sunlight to native grass leaves requires annual removal of large quantities of herbage biomass from the dominating introduced domesticated grasses. Relatively high stocking rates are required. Stocking rates greater than 100% can remove great quantities of domesticated grass herbage in a short time period but this is more harmful than helpful. The native grasses can not respond any faster than the rate of increase of the rhizosphere microorganism biomass. However, weedy forbs have mechanisms that can initiate extreme increases to greater sunlight and larger bare spaces and remain problems for many years. Stocking rates less than 85% do not remove enough domesticated grass herbage to effectively reduce the shading problem. Shaded native grasses do not improve but continue to decline. Stocking rates between 85% and 100% remove enough domesticated grass herbage biomass to permit an increased intensity of sunlight to reach the leaves of native grasses that increases the photosynthetic rates and fixes carbon at greater quantites that more closely matches the rate of increase in available mineral nitrogen mineralized by the increasing rhizosphere microorganism biomass.

The mixed grass prairie study area of nongrazed control 4 was a degraded silty ecological site dominated by Kentucky bluegrass. At the start of the study, the aboveground vegetation biomass consisted of 72.3% standing dead and litter and 27.7% live herbage. The live herbage biomass was 95.2% domesticated grasses and 2.1% native grasses. After 6 growing seasons, the aboveground vegetation biomass consisted of 61.7% standing dead and litter and 38.3% live herbage. The live herbage was 85.1% domesticated grasses and 7.7% native grasses.

Domesticated grass herbage production on nongrazed 4 started early and generally continued to increase through the growing season with peak or near peak production during mid October. Annual herbage biomass production of domesticated grass on nongrazed 4 was not significantly different during 2006, 2007, 2009, 2010, and 2011 growing seasons. Domesticated grass biomass during 2008 was significantly less than that produced during the 2006, 2007, 2009, and 2011 growing seasons (tables 3 and 9). The reduction in domesticated grass production during 2008 was caused by water deficiency conditions during October 2007, and April and May 2008 (table 1). Domesticated grass herbage production on nongrazed 4 was significantly greater than that on pasture 3 during the 2007, 2008, and 2009 growing seasons and significantly greater than that on pastures 1 & 2 during all six growing seasons (tables 9 and 11). Mean basal cover of domesticated grass was 12.97% on nongrazed 4 that was not significantly different than that on pasture 3 and significantly greater than that on pastures 1 & 2 (tables 9 and 11).

Native grass herbage production on nongrazed 4 was low with monthly biomass at less than 500 lbs/ac. Annual herbage biomass production of native grass on nongrazed 4 was not significantly different during the 2006, 2007, 2009, 2010, and 2011 growing seasons (tables 4 and 10). During the 2008 growing season, native grass herbage biomass production was reduced as a result of water deficiency conditions (table 1). Native grass biomass during 2008 was significantly less than that produced during 2009. Native grass biomass on nongrazed 4 was significantly greater that that on pasture 3 during the 2007, 2009, and 2010 growing seasons and significantly lower than native grass biomass on pastures 1 & 2 during the 2011 growing season (table 10). Mean basal cover of native grass on nongrazed 4 was 1.46%, that was significantly greater than that on pasture 3 and significantly less than that on pastures 1 & 2 (tables 10 and 11).

The mixed grass prairie study area of grazed pasture 3 was a degraded silty ecological site dominated by smooth bromegrass and Kentucky bluegrass. At the start of the study, the aboveground vegetation biomass consisted of 68.0% standing dead and litter and 32.0% live herbage. The live herbage biomass was 93.9% domesticated grasses and 2.1% native grasses. After 6 grazing seasons, the aboveground vegetation biomass consisted of 35.6% standing dead and litter and 64.4% live herbage. The live herbage was 89.6% domesticated grasses and 3.5% native grasses.

Domesticated grass herbage production on pasture 3 was abundant usually peaking early during June. Annual herbage biomass production of domesticated grass on pasture 3 was not significantly different during the 2006, 2007, 2009, and 2010 growing seasons. Domesticated grass biomass during 2008 was significantly less than that produced during the 2006, 2009, and 2011 grazing seasons (tables 5 and 9). Some of the reduction in domesticated grass herbage production during 2008 was caused by water deficiency conditions during October 2007, and April and May 2008 (table 1). However, a large portion of the reduction in domesticated grass herbage biomass resulted from the affects of three grazing seasons of cow and calf grazing at 87.4% of the assesses stocking rate (table 2). The herbage production of domesticated grasses during 2011 was significantly greater than that produced during the other grazing seasons (table 9). Some of the increased domesticated grass herbage production was caused by precipitation at 169% of long-term mean during April, May, and July 2011 (table 1) and some of the increase resulted from low grazing pressure from steers stocked at 37.8% of the assessed stocking rate (table 2). Domesticated grass herbage production on pasture 3 was significantly less than that produced on nongrazed 4 during the 2007, 2008, and 2009 growing seasons and was significantly greater than that produced on pastures 1 & 2 during 2006, 2009, 2010, and 2011 growing seasons (tables 9 and 11). Mean basal cover of domesticated grass was 13.83% on pasture 3 that was not significantly different than that on nongrazed 4 and was significantly greater than that on pastures 1 & 2 (tables 9 and 11).

Native grass herbage production on pasture 3 was low with monthly biomass at less than 200 lbs/ac after the first year, 2006. Annual herbage biomass production of native grass on pasture 3 was not significantly different during all six growing seasons (tables 6 and 10). Native grass biomass production on pasture 3 was significantly less than that produced on nongrazed 4 during the 2007, 2009, and 2010 grazing seasons and was significantly less than that on pastures 1 & 2 during the 2007, 2008, 2009, 2010, and 2011 grazing seasons (tables 10 and 11). Mean basal cover of native grass was 0.18% on pasture 3 that was significantly less than those on nongrazed 4 and grazed pastures 1 & 2 (tables 10 and 11).

The mixed grass prairie study areas of grazed pastures 1 & 2 were degraded silty ecological sites dominated by Kentucky bluegrass. At the start of the study, the aboveground vegetation biomass consisted of 63.6% standing dead and litter and 36.4% live herbage. The live herbage biomass was 64.8% domesticated grasses and 3.4% native grasses. After 6 grazing seasons, the aboveground vegetation biomass consisted of 39.5% standing dead and litter and 60.5% live herbage. The live herbage biomass was 56.0% domesticated grasses and 27.0% native grasses.

Domesticated grass herbage biomass on pastures 1 & 2 progressively decreased during the first three grazing seasons as a result of the grazing pressure from cow-calf pairs. The low monthly domesticated grass biomass was less than 500 lbs/ac during the 2008 grazing season. Domesticated grass herbage production during the 2008, 2009, and 2010 grazing seasons was significantly less than that produced during the 2007 and 2011 grazing seasons (tables 7 and 9). Some of the increase in domesticated grass herbage production during 2011 was caused by high rainfall during April, May, and July 2011 (table 1) and some of the increase in domesticated grass resulted from the low stocking rates of grazing steers (table 2). Domesticated grass herbage biomass on pastures 1 & 2 was significantly less than that on nongrazed 4 during the six grazing seasons and was significantly less than that on pasture 3 during the 2006, 2009, 2010, and 2011 grazing seasons (tables 9 and 11). Mean basal cover of domesticated grass was 5.7% on pastures 1 & 2 that was significantly less than those on nongrazed 4 and grazed pasture 3 (tables 9 and 11).

Native grass herbage production on pastures 1 & 2 increased gradually during the six grazing seasons. Annual herbage biomass production of native grass on pastures 1 & 2 during the 2009 and 2011 grazing seasons was significantly greater than that produced during the 2006 and 2008 grazing seasons (tables 8 and 10). Native grass herbage production on pastures 1 & 2 was significantly greater than that produced on nongrazed 4 during the 2011 grazing season and was significantly greater than that produced on pasture 3 during the 2007, 2008, 2009, 2010, and 2011 grazing seasons (table 10). Mean basal cover of native grass was 6.62% on pastures 1 & 2 that was significantly greater than those on nongrazed 4 and pasture 3 (tables 10 and 11).

Rhizosphere weight (tables 12, 13, and 14) changed very little during the first two growing seasons and were not significantly different on nongrazed 4, pasture 3, and pastures 1 & 2. From the second growing season to the sixth growing season, the rhizosphere weights increased at different rates on the three management treatments resulting in significantly different mean annual rhizosphere weights on nongrazed 4 (table 12), pasture 3 (table 13), and pastures 1 & 2 (table 14).

Rhizosphere weight on nongrazed 4 increased 115.8% from a mean pregrazing rhizosphere weight of 60.49 kg/m^3 (14.9% of reference weight) at a mean rate of 13.2 kg/m^3/yr and after six growing seasons, the mean rhizosphere weight reached 130.56 kg/m^3 (table 12), which was 32.1% of the reference rhizosphere weight of 406.44 kg/m^3.

Rhizosphere weight on pasture 3 increased 176.2% from a mean pregrazing rhizosphere weight of 60.49 kg/m^3 at a mean rate of 23.8 kg/m^3/yr and after six grazing seasons, the mean rhizosphere weight reached 167.05 kg/m^3 (table 13), which was 41.1% of the reference rhizosphere weight of 406.44 kg/m^3.

Rhizosphere weight on pastures 1 & 2 increased 254.3% from a mean pregrazing rhizosphere weight of 60.49 kg/m^3 at a mean rate of 30.5 kg/m^3/yr and after six grazing seasons, the mean rhizosphere weight reached 214.34 kg/m^3 (table 14), which was 52.7% of the reference rhizosphere weight of 406.44 kg/m^3.

Mean annual rhizosphere weights on nongrazed 4 responded differently than those on grazed pasture 3 and on grazed pastures 1 & 2. The severely degraded silty ecological site on nongrazed 4 was dominated by Kentucky bluegrass and had no defoliation treatments. The changes in annual rhizosphere weights were related only to changes in growing season precipitation. The first growing season had low precipitation at 63.7% of the long-term mean and four growing season months (May, June, July, and August) had water deficiency conditions (table 1). The growing season precipitation during the second to the fourth growing seasons had mean precipitation at 76.2% of the long-term mean. Rhizosphere weight on nongrazed 4 changed little during the first four growing seasons. The precipitation increased during the fifth and sixth growing seasons at 103.4% of the long-term mean causing the rhizosphere weight to increase during the sixth growing season (table 12). The rhizosphere weight on nongrazed 4 increased 115.8% from a mean pregrazing rhizosphere weight of 60.49 kg/m^3

to 130.56 kg/m^3 in six years as a result of changes in annual growing season precipitation.

Mean annual rhizosphere weights on pasture 3 responded differently than that on nongrazed 4. The severely degraded silty ecological site on pasture 3 was dominated by smooth bromegrass and Kentucky bluegrass at a pretreatment composition of 93.9% and were partially defoliated annually by livestock grazing managed with the twice-over rotation system during six grazing seasons. Cow-calf pairs grazed during the first three years at a mean of 87.4% of the assessed stocking rate. Utilization of the smooth bromegrass herbage by the cows was around 36.4%. In three years, domesticated grass herbage biomass decreased 52.7% to 723.31 lbs/ac, standing dead biomass decreased 74.2% to 351.40 lbs/ac, litter biomass decreased 10.5% to 1883.70 lbs/ac, and native grass herbage biomass increased 64.7% to 57.60 lbs/ac. Heifers grazed during the next two years at a mean of 83.6% of the assessed stocking rate. Utilization of the smooth bromegrass herbage by the heifers was around 19.4%. In two years, standing dead biomass decreased 0.4% to 349.94 lbs/ac, litter biomass decreased 57.8% to 794.03 lbs/ac, domesticated grass herbage biomass increased 71.2% to 1238.61 lbs/ac, and native grass herbage biomass decreased 3.0% to 55.86 lbs/ac. Steers lightly grazed during the sixth year at 37.8% of the assessed stocking rate. Utilization of the smooth bromegrass herbage by the steers was around 6.1%. Domesticated grass herbage biomass increased 77.0% to 2191.98 lbs/ac, standing dead biomass decreased 26.8% to 256.28 lbs/ac, litter biomass increased 37.7% to 1093.64 lbs/ac, and native grass herbage biomass increased 52.4% to 85.12 lbs/ac.

The twice-over rotation grazing treatment on pasture 3 caused the rhizosphere weight to increase significantly greater than that on nongrazed 4. The increase of rhizosphere weight was not connected to changes in native grass. This increase was associated with the increase in Kentucky bluegrass basal cover. Native grass did not increase in herbage biomass and basal cover. The defoliation resistance mechanisms did not activate in native grass on pasture 3 most likely from deficiencies in the quantity of available fixed carbon. The grazing pressure did not remove sufficient quantities of standing smooth bromegrass that shaded the shorter native grasses and prevented the required rates of photosynthesis to occur causing the shortage of available carbon.

Smooth bromegrass is considered to be nonmycorrhizal and does not readily develop symbiotic relationships with rhizosphere organisms and cannot assist with increasing rhizosphere weight in an ecosystem. During another study, Manske (2007) recorded 32.3% of the root segments of smooth bromegrass from the control (no defoliation) treatment to be infected with endomycorrhizal fungi assessed by a present or absent grid-intersect method. However, nearly all of the fungal infections observed in the biologically active root segment samples were restricted to the root hairs. Almost none of the root segment samples had fungal colonization within the root tissue.

During six years of twice-over rotation grazing on pasture 3, basal cover of smooth bromegrass increased 63.3% and basal cover of Kentucky bluegrass increased 511.1%. The relative composition during the first year was 52.2% smooth bromegrass and 31.3% Kentucky bluegrass. As a result of the effects of grazing, the Kentucky bluegrass increased. The relative composition during the sixth year was 64.7% Kentucky bluegrass and 28.8% smooth bromegrass (table 15). Six years of livestock grazing on pasture 3 caused the relative composition of Kentucky bluegrass to increase 106.7% and the associated 176.2% increase in rhizosphere weight (table 15). The rhizosphere weight on pasture 3 was significantly greater during the third through the sixth years than the rhizosphere weight on nongrazed 4 (table 15).

Mean annual rhizosphere weights on pastures 1 & 2 responded differently that that on nongrazed 4. The severely degraded silty ecological sites on pastures 1 & 2 were dominated by Kentucky bluegrass and were partially defoliated annually by livestock grazing managed with the twice-over rotation system during six grazing seasons. Cow-calf pairs grazed during the first three years at a mean of 87.4% of the assessed stocking rate. In three years, domesticated grass herbage biomass decreased 70.9% to 310.77 lbs/ac, standing dead biomass decreased 65.9% to 420.37 lbs/ac, litter biomass decreased 32.9% to 1114.80 lbs/ac, and native grass herbage biomass increased 295.5% to 211.74 lbs/ac. Domesticated grass basal cover increased 18.3% to 4.08% and native grass basal cover increased 198.7% to 6.81%. Heifers grazed during the next two years at a mean of 83.6% of the assessed stocking rate. In two years, domesticated grass herbage biomass increased 120.7% to 685.72 lbs/ac, standing dead biomass decreased 13.5% to 363.68 lbs/ac, litter biomass decreased 57.5% to 473.94 lbs/ac, and native grass herbage biomass increased 63.8% to 346.76 lbs/ac. Domesticated grass basal cover increased 68.6% to 6.88% and native grass basal cover increased 12.5% to 7.66%. Steers lightly grazed during the sixth year at 37.8% of the assessed stocking rate. Domesticated grass herbage biomass increased 83.1% to 1261.24 lbs/ac, standing dead biomass increased 40.2% to 509.77 lbs/ac, litter biomass increased 89.5% to 898.17 lbs/ac, and native grass herbage biomass increased 63.5% to 567.07 lbs/ac. Domesticated grass basal cover remained the same at 6.88% and native grass basal cover decreased 15.7% to 6.46%.

The twice-over rotation grazing treatment for six years on pastures 1 & 2 caused the rhizosphere weight to increase significantly greater than that on nongrazed 4 (table 16 and figure 1). The increase of rhizosphere weight on pastures 1 & 2 was associated with the increase in native grass. The rhizosphere weights on pastures 1 & 2 were not significantly different than those on nongrazed 4 during the first two growing seasons. From the second growing season to the sixth growing season, the rate of rhizosphere weight increase was greater on pastures 1 & 2 than on nongrazed 4. The rhizosphere weights on pastures 1 & 2 were significantly greater than those on nongrazed 4 during each year from year three to year six (table 16 and figure 1). The rhizosphere weight on pastures 1 & 2 increased 254.3% from a mean pregrazing rhizosphere weight of 60.49 kg/m^2 to 214.34 kg/m^3 in six years in association with 183.3% increase in basal cover and 959.4% increase in herbage biomass of native grasses.

Domesticated grass biomass production was not significantly different on nongrazed 4 and pasture 3 during the 2013 and 2014 growing seasons. Domesticated grass biomass production on pastures 1 & 2 was significantly less than that produced on nongrazed 4 and pasture 3 during the 2013 and 2014 growing seasons (tables 17 and 18).

Domesticated grass biomass production on nongrazed 4 during the 2013 and 2014 growing seasons was not significantly different than that produced during 2006, 2007, 2009, 2010, and 2011 growing seasons and was significantly greater than that produced during the 2008 growing season (tables 3, 9, 11, 17, and 18). Domesticated grass biomass production on pasture 3 during the 2013 and 2014 grazing seasons was not significantly different than that produced during 2011 grazing season and was significantly greater than that produced during the 2006 to 2010 grazing seasons (tables 5, 9, 11, 17, and 18). Domesticated grass biomass production on pastures 1 & 2 during the 2013 and 2014 grazing seasons was significantly greater than that produced during the 2008 and 2009 grazing seasons but not significantly different than that produced during the 2006, 2007, 2010 and 2011 grazing seasons (tables 7, 9, 11, 17, and 18). Domesticated grass basal cover during 2013 and 2014 on nongrazed 4 and pasture 3 was significantly greater than that on the respective treatments during 2006 to 2011 (tables 9, 11, and 18). Domesticated grass basal cover during 2013 and 2014 on pastures 1 & 2 was not significantly different than that during 2006 and 2011 (tables 9, 11, and 18).

Native grass biomass production on pastures 1 & 2 was significantly greater than that on pasture 3 during the 2013 and 2014 grazing seasons. Native grass biomass production was not significantly different on nongrazed 4 and pasture 3 during the 2013 and 2014 growing seasons (tables 17 and 18).

Native grass biomass production on nongrazed 4 during the 2013 and 2014 growing seasons was not significantly different than that produced during the 2006 to 2011 growing seasons (tables 4, 10, 11, 17, and 18). Native grass biomass production on pasture 3 during the 2013 and 2014 grazing seasons was not significantly different than that produced during the 2006 to 2011 grazing seasons (tables 6, 10, 11, 17, and 18). Native grass biomass production on pastures 1 & 2 during the 2013 and 2014 grazing seasons was not significantly different than that produced during the 2006 to 2011 grazing seasons (tables 8, 10, 11, 17, and 18). Native grass basal cover during 2013 and 2014 on nongrazed 4, pasture 3, and pastures 1 & 2 was not significantly different than that on the respective treatments during 2006 to 2011 (tables 10, 11, and 18).

The quantity of mineral nitrogen available in a soil is the net difference between the total quantity of organic nitrogen mineralized by soil microorganisms and the quantity of mineral nitrogen immobilized by plants and soil microbes (Brady 1974, Legg 1975). The quantity of available mineral nitrogen varies cyclically with changes in soil temperature, soil microorganism biomass, and plant phenological growth and development during the growing season (Whitman 1975). The relationships between soil microorganism activity and phenology of plant growth activity results in a dynamic cycle of available mineral nitrogen (Goetz 1975). When mineralization activity by soil microbes is greater than plant growth activity, the quantity of available mineral nitrogen increases. When transformation (immobilization) of mineral nitrogen by plant and soil microbe growth activity is greater than mineralization activity, the quantity of available mineral nitrogen decreases.

The available mineral nitrogen cycle model for a typical growing season would have three peaks and three valleys (Whitman 1975). The first peak of mineral nitrogen would occur in mid May. As plant growth rates increased in June, transformation would increase with available mineral nitrogen at a low value during late June-early July. Mineral nitrogen would increase and reach a second peak during late July or early August. Fall tillers and fall tiller buds start development in mid August and would cause a decrease in available mineral nitrogen until mid October. A third peak would occur shortly after mid October. And when liquid water becomes unavailable with winter soil freeze up, available mineral nitrogen would decline for a third low period.

Nitrate (NO_3) cycle on nongrazed 4 had minor peaks during May at the 0-6 and 6-12 inch soil depths (table 19). Generally, available mineral nitrate was low and transformation was low during the growing season at the 0-6 and 6-12 inch soil depths. At the 12-24 inch soil depth, mineral nitrate accumulated during June, July, and August. There was a slight decrease in available mineral nitrate and a slight increase in transformation during August and September at the 0-6 inch soil depth. There was a slight increase in available mineral nitrate and a slight decrease in transformation during July and again during October at the 0-6 inch soil depth (table 19).

Ammonium (NH_4) cycle on nongrazed 4 had a peak during May at the 0-6 inch soil depth and a minor peak at the 6-12 inch soil depth (table 19). Generally, available mineral ammonium was moderate during the growing season at all soil depths and transformation was moderate at the 0-6 inch soil depths, low at the 6-12 inch soil depth, and ammonium accumulated at the 12-24 inch soil depth. There was a slight decrease in available mineral ammonium and a slight increase in transformation during August at the 0-6 inch soil depth. There was a slight increase in available mineral ammonium and a slight decrease in transformation during July at the 0-6 inch soil depth (table 19).

Nitrate (NO_3) cycle on pasture 3 had a peak during May at the 0-6 inch soil depth (table 20). Generally, available mineral nitrate was low and transformation was low during the growing season at all soil depths. Except transformation was at a moderate level during July to October at the 0-6 inch soil depth. There was a slight decrease in available mineral nitrate and a slight increase in transformation during August and again during October at the 0-6, 6-12, and 12-24 inch soil depths. There was a slight increase in available mineral nitrate and a slight decrease in transformation during July and again during September at the 0-24 inch soil depth (table 20).

Ammonium (NH_4) cycle on pasture 3 was delayed. The available mineral ammonium was greater during June than during May at the 0-6, 6-12, and 12-24 inch soil depths (table 20). There was a slight decrease in available mineral ammonium and a slight increase in transformation during July and August and again during October at the 0-6, 6-12, and 12-24 inch soil depths. There was a slight increase in available mineral ammonium and a great enough decrease in transformation to cause ammonium accumulation during September at the 0-24 inch soil depth (table 20).

Nitrate (NO_3) cycle on pastures 1 & 2 had a peak during May at the 0-6, 6-12, and 12-24 inch soil depths (table 21). Generally, available mineral nitrate was high and transformation was high during June through October at the 0-6, 6-12, and 12-24 inch soil depths. There was a slight increase in available mineral nitrate and a slight decrease in transformation during August at the 0-6 and 12-24 inch soil depths. There was a slight decrease in available mineral nitrate and a slight increase in transformation during July and again during September and October at the 0-6 and 12-24 inch soil depths (table 21).

Ammonium (NH_4) cycle on pastures 1 & 2 had a peak during May at the 0-6 inch soil depth and had minor increases at the 6-12 and 12-24 inch soil depths (table 21). Generally, available mineral ammonium was high and transformation was high during the growing season at all soil depths. There was a trend in the 0-24 inch soil column for a slight increase in available mineral ammonium and a slight decrease in transformation during July, August, and September. There was a slight decrease in available mineral ammonium and a slight increase in transformation during June and again in October at the 0-24 inch soil depth (table 21).

Available mineral nitrogen ($NO_3 + NH_4$) on nongrazed 4 was 15.8% greater than that on pasture 3. However, transformed mineral nitrogen on pasture 3 was 16.5% greater than that used on nongrazed 4. Mineral nitrogen use on nongrazed 4 was primarily from the 0 to 12 inch soil depths. Both nitrate (NO_3) and ammonium (NH_4) tended to accumulate at the 12 to 24 inch soil depths on nongrazed 4 (table 22).

Available mineral nitrogen ($NO_3 + NH_4$) on pastures 1 & 2 was 8.9% greater than that on nongrazed 4 and was 26.1% greater than that on pasture 3. Transformed mineral nitrogen on pastures

1 & 2 was 234.3% greater than that used on nongrazed 4 and was 187.0% greater than that used on pasture 3 (table 22).

Available mineral nitrogen ($NO_3 + NH_4$) during May was lowest on pasture 3. Pasture 3 was dominated by smooth bromegrass and nongrazed 4 and pastures 1 & 2 were dominated by Kentucky bluegrass. Available mineral nitrogen on nongrazed 4 during May was 10.5% greater than that on pasture 3. Nitrate (NO_3) was 8.4% greater and ammonium (NH_4) was 12.2% greater on nongrazed 4 during May than that on pasture 3 (table 23).

Available mineral nitrogen ($NO_3 + NH_4$) during May was greatest on pastures 1 & 2. Both nitrate and ammonium was available during May at greater quantities at each soil depth on pastures 1 & 2 than those on nongrazed 4 and pasture 3. Available mineral nitrogen on pastures 1 & 2 during May was 39.5% greater than that on nongrazed 4 and was 54.2% greater than that on pasture 3. Pastures 1 & 2 were the only treatment that had mineral nitrogen available at quantities near 100 lbs/ac (table 23).

Summary of Results

The first four growing seasons of the study received precipitation that was 73.1% of the long-term mean. During the fifth and sixth years, growing season precipitation was 103.4% of the long-term mean. Nongrazed 4 had no defoliation treatment during the study. Pasture 3 and pastures 1 & 2 had partial defoliation by grazing treatment that was controlled by the twice-over rotation strategy. Cow-calf pairs grazed during the first three years at a mean of 87.4% of the assessed stocking rate, heifers grazed during the fourth and fifth years at a mean of 83.6% of the assessed stocking rate, and steers grazed during the sixth year at 37.8% of the assessed stocking rate.

The nongrazed 4 plant communities on the degraded silty ecological site was dominated by Kentucky bluegrass. Mean annual domesticated grass herbage biomass was 1847.69 lbs/ac that changed little. There was a 24.8% decrease and a 17.0% decrease during the third and fifth growing seasons, respectively, and a 39.1% increase during the sixth growing season. Mean annual domesticated grass basal cover was 12.97% that had small annual changes. These changes to the domesticated grasses were related to changes in the precipitation pattern. Mean annual native grass herbage biomass was 240.42 lbs/ac and mean annual basal cover was 1.46%. There was a small quantity of native grasses remaining on nongrazed 4 and most of the native grass was a small remnant colony of prairie sandreed that was able to persist because it was taller than the Kentucky bluegrass.

The pasture 3 plant community on the degraded silty ecological site was dominated by smooth bromegrass and Kentucky bluegrass. Mean annual domesticated grass herbage biomass was 1434.37 lbs/ac. There was a 49.1% decrease during the first three grazing seasons. Increased herbage was produced during the fourth and fifth grazing seasons with an 85.2% increase during the sixth grazing season. Mean annual domesticated grass basal cover was 13.83%. Basal cover increased during the second to the fifth grazing season at a mean of 25.4% per year. During the six year period, smooth bromegrass basal cover increased 63.3% and composition decreased from 52.2% to 28.8%; Kentucky bluegrass basal cover increased 511.1% and composition increased from 31.3% to 64.7%. These changes to the domesticated grasses were related to changes in the precipitation pattern and to changes in the grazing pressure. Mean annual native grass herbage biomass was 80.92 lbs/ac and mean annual basal cover was 0.18%. There was an extremely small quantity of native grasses remaining on pasture 3. Livestock utilization of the smooth bromegrass vegetation was too low to help the native grasses. Cow-calf pairs grazed three seasons at a mean of 87.4% of assessed stocking rate and utilized 36.4% of the smooth bromegrass; heifers grazed two seasons at a mean of 83.6% of the assessed stocking rate and utilized 19.4% of the smooth bromegrass; and steers grazed one season at 37.8% of the assessed stocking rate and utilized 6.1% of the smooth bromegrass.

The pastures 1 & 2 plant communities on the degraded silty ecological sites were dominated by Kentucky bluegrass. Mean annual domesticated grass herbage biomass was 847.25 lbs/ac. The three grazing seasons of cow-calf grazing at a mean of 87.4% of the assessed stocking rate reduced the domesticated herbage biomass 65.6%. The two grazing seasons of heifer grazing at a mean of 83.6% of the assessed stocking rate maintained the domesticated herbage at relatively low weight levels with only slight increases. The one grazing season of steers grazing at 37.8% of the assessed stocking rate plus above average precipitation during April, May, July, and August of the sixth grazing season permitted domesticated herbage to increase 85.5% during one growing season. Mean annual domesticated grass basal cover was 5.70%. The basal cover increased 2 percentage points during six grazing seasons. The reductions in the domesticated grass vegetation were related to the grazing strategy. Mean annual native grass herbage biomass was 377.91 lbs/ac. The native grass herbage biomass generally increased with a six year increase of 166.3%. Mean annual native grass basal cover was 6.62%. Native grass basal cover had a six year increase of 152.9%. The increases in the native grass vegetation were related to the grazing strategy.

Rhizosphere weight on nongrazed 4 increased 115.9% at a mean rate of 13.2 kg/m^3/yr, with 50.3% of that increase occurring during the sixth growing season, from 14.9% to 32.1% of the reference rhizosphere weight. The small increase in rhizosphere weight on nongrazed 4 were related to the increase in precipitation during the fifth and sixth growing seasons.

Rhizosphere weight on pasture 3 increased 176.2% at a mean rate of 23.8 kg/m^3/yr from 14.9% to 41.1% of the reference rhizosphere weight. The increase in rhizosphere weight on pasture 3 was related to the increase in Kentucky bluegrass composition from 31.3% to 64.7% and the decrease in smooth bromegrass composition from 52.2% to 28.8% that was caused by the grazing management. Smooth bromegrass does not develop symbiotic relationships with rhizosphere microorganism and Kentucky bluegrass can live with or without the symbiotic soil microbes. On pasture 3, the rhizosphere microbes developed in association with Kentucky bluegrass roots.

Rhizosphere weight on pastures 1 & 2 increased 254.3% at a mean rate of 30.5 kg/m^3/yr from 14.9% to 52.7% of the reference rhizosphere weight. The increase in rhizosphere weight on pastures 1 & 2 was related to the increase in native grasses, the decrease in domesticated grasses, and the increase in ecosystem biogeochemical processes caused by the twice-over rotation strategy.

Available mineral nitrate during May and during the growing season was greatest on pastures 1 & 2. Available mineral nitrate on pastures 1 & 2 during May was 78.3% greater than that on pasture 3 and was 64.6% greater than that on nongrazed 4. During the growing season, available mineral nitrate on pastures 1 & 2 was 33.1% greater than that on pasture 3 and was 5.2% greater than that on nongrazed 4. During May, available mineral nitrate on nongrazed 4 was 8.4% greater than that on pasture 3, and during the growing season, available mineral nitrate on nongrazed 4 was 26.5% greater than that on pasture 3. Transformation of mineral nitrate was greatest on pastures 1 & 2. Transformation of mineral nitrate on pastures 1 & 2 was 173.7% greater than that used on pasture 3 and was 291.2% greater than that used on nongrazed 4. Transformation of mineral nitrate on pasture 3 was 42.9% greater than that used on nonrgazed 4.

Available mineral ammonium during May and during the growing season was greatest on pastures 1 & 2. Available mineral ammonium on pastures 1 & 2 during May was 35.2% greater than that on pasture 3 and was 20.5% greater than that on nongrazed 4. During the growing season, available mineral ammonium on pastures 1 & 2 was 22.1% greater than that on pasture 3 and was 11.3% greater than that on nongrazed 4. During May, available mineral ammonium on nongrazed 4 was 12.2% greater than that on pasture 3 and during the growing season, available mineral ammonium on nongrazed 4 was 9.7% greater than that on pasture 3. Transformation of mineral ammonium was greatest on pastures 1 & 2. Transformation of mineral ammonium on pastures 1 & 2 was 241.7% greater than that used on pasture 3 and was 124.1% greater than that used on nongrazed 4. Transformation of mineral ammonium on nongrazed 4 was 52.5% greater than that used on pasture 3.

Discussion

Native grassland ecosystems are complex and consist of numerous interrelated essential components. The major biotic components are the grassland plants, the grazing graminivores, and the rhizosphere microorganisms. The abiotic components are sunlight energy, the major essential elements of carbon, hydrogen, nitrogen, and oxygen, the minor essential elements of macronutrients and micronutrients, and the environmental factors.

Traditional management of grasslands is based on the use of the aboveground grass plants as forage for livestock. The belowground rhizosphere microorganisms that cycle the essential elements and are the renewable portion of natural resources are not considered in traditional management. Sometimes the use priority changes from forage for livestock to recreation and/or wildlife habitat. The graminivores are than considered to be a competing use and are removed. The symbiotic relationships among the three main biotic components and the cycling of essential elements are destroyed without the presence of graminivores.

Grass plants produce double the leaf biomass than that needed for growth and development (Crider 1955, Coyne et al. 1995). The extra plant biomass provides nutrious forage to graminivores and acts as an enticement for partial defoliation when the lead tillers are at vegetative growth stages that forces the exudation of large quantities of short chain carbon energy into the rhizosphere. This exudated energy causes the biomass of the microbes to increase that results in greater mineralization of organic nitrogen. Removal of the graminivores effectually turns the previously beneficial extra leaf biomass into detrimental shade producing vestiges and prevents the exudation of carbon energy into the rhizosphere. With the reduction of carbon energy exudates, the rhizosphere microbes decrease in biomass resulting in a reduction in organic nitrogen mineralization. The decrease in available mineral nitrogen causes a great reduction of native grass leaf and root growth and a reduction in vegetative tiller development to less than one secondary tiller per lead tiller.

Prior to the decision to eliminate the competitive usage of resources by grazing graminivores, the area was a working cattle ranch that was managed with traditional concepts supported by good land stewardship ethics and the land most likely was in a typical low good condition with mineral nitrogen available at 50 to 60 lbs/ac and with the biomass of the rhizosphere microorganisms at 50% to 60% of the biomass needed to mineralize 100 lbs/ac of mineral nitrogen. The native grassland ecosystems deteriorated rapidly after grazing graminivores were eliminated. The rhizosphere microorganism biomass would decrease greatly during the first two growing seasons then continue to decrease at a slower rate for another year or two. Rhizosphere biomass can be sustained by the leakage of carbon energy from a grass plant at a low weight at about 14.9% of the rhizosphere biomass required to mineralize 100 lbs/ac of mineral nitrogen.

Analytically, the primary initial cause for the native grassland ecosystem degradation after the graminivores were removed would be the huge reduction in the quantity of available mineral nitrogen. The reduction in mineral nitrogen causes great decreases in native grass herbage production and decreases in basal cover which creates open bare ground spots in the plant community.

Opportunistic plants like smooth bromegrass and Kentucky bluegrass invade and rapidly fill the open spaces. Kentucky bluegrass expansion is limited in the area that the taller smooth bromegrass has invaded. Both smooth bromegrass and Kentucky bluegrass have labile roots that rapidly breakdown after death providing the essential elements back to the plants to produce new growth of leaves, stems, and roots. Within a few years, the greater aboveground growth of the invading domesticated cool season grasses dominate the plant communities and shade access of sunlight energy to the remaining stunted native grasses that have impeded growth as a result of great deficiencies in the essential elements of nitrogen, carbon, hydrogen, and oxygen.

After 13 years without grazing graminivores, the aboveground vegetative biomass on the degraded plant communities consisted of 40.0% litter, 27.4% standing dead, and 32.6% live herbage. The live herbage biomass was 79.9% domesticated grasses, 2.7% native grasses, 11.9% upland sedge, and 5.5% forbs. The mean rhizosphere biomass was reduced to 60.49 kg/m^3 which was only 14.9% of the reference rhizosphere weight.

The severely degraded silty ecological sites on pastures 1 & 2 were partially restored by grazing management with the twice-over rotation strategy during a nine year period 2006 to 2014. Composition of the invader domesticated grass, Kentucky bluegrass, decreased 50.9% on pastures 1 & 2 as a result of the grazing treatment. Kentucky bluegrass composition was 64.8% in 2006, decreased to 56.0% in 2011, and decreased to 31.8% during 2013 to 2014. Kentucky bluegrass herbage biomass decreased 51.8% to a mean low of 537.14 lbs/ac during the three year period, 2008 to 2010. Composition of native grass (cool and warm season grasses) increased 641.2% on pastures 1 & 2 as a result of the grazing treatment. Native grass composition was 3.4% in 2006, increased to 27.0% in 2011, and was at 25.2% during 2013 to 2014. Native grass herbage biomass increased 166.3% to a mean high at 616.43 lbs/ac during 2011. The rhizosphere weight increased 254.3% on pastures 1 & 2 at a mean rate of 30.5 kg/m^3/yr to a mean high during 2011 at 214.34 kg/m^3, which was 52.7% of the reference rhizosphere weight. Available mineral nitrogen on pastures 1 & 2 was the greatest during 2013 to 2014 at 99.35 lbs/ac; almost at the threshold quantity of 100 lbs/ac. Available mineral nitrogen on pastures 1 & 2 was 39.5% greater than that on nongrazed 4 and 54.2% greater than that on pasture 3. The grassland ecosystems on pastures 1 & 2 can be further restored with a rhizosphere biomass near the reference weight of 406.44 kg/m^3, and mineral nitrogen available at quantities greater than 100 lbs/ac and the plant communities can be restored with a dominance of native grasses and negligible amounts of Kentucky bluegrass with continuance of stocking rates between 85% and 100% of the assessed rate and management of grazing with the twice-over rotation strategy.

The native grassland ecosystems on pasture 3 were not restored with the twice-over treatment because the inhibitive shading effects from the live and standing dead smooth bromegrass was not reduced sufficiently with defoliation by grazing at stocking rates of less than 85% of the assessed rate with yearling heifer and steer stocker livestock. Shading of sunlight energy impeded native grass growth. Native grass monthly herbage biomass remained low at less than 200 lbs/ac during the second to the sixth growing seasons and the mean basal cover of native grass was only 0.18% on pasture 3.

Light energy from the sun is necessary for photosynthesis. Light penetration through a tall grass leaf canopy can be about 20% of the light levels above the canopy (Peltzer and Kochy 2001). Native grasses have high light saturation points and require near full sunlight. Warm season grasses have higher light saturation points than cool season grasses (Kochy 1999, Kochy and Wilson 2000). Shading reduces native warm season grasses more than native cool season grasses. Introduced cool season domesticated grasses have lower light saturation points than native grasses, permitting domesticated grasses to live and survive in low light conditions.

Shaded native grasses with low amounts of sunlight reaching the leaves have reduced rates of photosynthesis, which reduces the quantity of atmospheric carbon dioxide fixed and reduces the quantity of simple carbohydrates produced (Coyne et al. 1995). Low quantities of carbohydrates causes decreases in growth of roots, leaves, and stems, and reduces the development of secondary tillers. Low quantities of carbohydrates also cause increases in the rates of leaf senescence and increases tiller mortality that results in reductions of grass plant density (basal cover) and reductions of herbage biomass production (Langer 1972, Grant et al. 1983, Briske and Richards 1995).

Severely degraded native grassland ecosystems can not be restored rapidly. The rhizosphere microorganism biomass must be restored before the native grass species composition can be recovered. The restoration of the ecosystem can progress at the rate of rhizosphere restoration. Rhizosphere restoration requires increasing the quantity of short chain carbon energy exudation from partially defoliated lead tillers of host grass species that form symbiotic relations with rhizosphere organisms and that retain 67% to 75% of their live leaf biomass for active carbon fixation during vegetative growth stages. Mowing at hay cutting height or burning removes too much leaf biomass for the host grass species to increase rhizosphere microbe biomass with surplus short chain carbon energy.

Smooth bromegrass was domesticated several centuries ago and does not form effective symbiotic relations with rhizosphere microbes. Because smooth bromegrass is not a host species, fifty percent of its leaf biomass can be removed by grazing during vegetative growth stages to reduce the detrimental shading effects on the growth of small understory native grass plants. However, if too much of the shading material is removed early in the restoration process, the increased sunlight reaching the ground will activate seeds of weedy plants to grow, turning restoration backwards and lengthening the recovery time.

The rhizosphere weight increased 176.2% on pasture 3 at a mean rate 23.8 kg/m^3/yr to a mean high during 2011 at 167.05 kg/m^3, which was 41.1% of the reference rhizosphere weight. Available mineral nitrogen on pasture 3 was the lowest during 2013 to 2014 at 64.45 lbs/ac. Available mineral nitrogen on pasture 3 was 9.5% lower than that on nongrazed 4 and 35.1% lower than that on pastures 1 & 2. Domesticated grass herbage biomass and basal cover on pasture 3 was not different than that on nongrazed 4 during 2010 to 2011 and 2013 to 2014. The composition of the domesticated grass shifted from 52.2% smooth bromegrass and 31.3% Kentucky bluegrass to 64.7% Kentucky bluegrass and 28.8% smooth bromegrass in six grazing seasons.

Native grass herbage biomass was barely present at 77.37 lbs/ac during 2010 to 2011 and at 56.50 lbs/ac during 2013 to 2014. These data indicate that native grasses on pasture 3 were impeded by deficiencies in available nitrogen and carbon and that rhizosphere microorganisms with biomass at less the 50% of the reference rhizosphere biomass are not great enough to restore native grassland ecosystems. Even though the ecosystems on pasture 3 are a long way from being restored, some progress was made. The rhizosphere biomass had increased and was 28.0% greater than that on nongrazed 4. The composition of smooth bromegrass was reduced 44.8%. The shading problem caused by live and standing dead smooth bromegrass must be reduced in order for the native grass leaves to fix sufficient quantities of carbon. After the rhizosphere biomass increases above 50% of the reference rhizosphere weight, the available mineral nitrogen will increase and then the native grass species will increase and eventually outcompete the Kentucky bluegrass as was done on pastures 1 & 2.

Degradation of native grassland ecosystems occurs when the available mineral nitrogen (NO_3 nitrate and NH_4 ammonium) drops below 100 lbs/ac. However, native grassland soils are not deficient of nitrogen. Most of the nitrogen is immobilized in the soils as organic nitrogen. Grassland soils in the Northern Plains contains organic nitrogen at a range of 3 to 8 tons per acre. Organic nitrogen is not usable by plants. Soil organic nitrogen must be converted into mineral nitrogen through mineralization by soil microorganisms. In grasslands, almost all of the soil microorganisms live in the rhizosphere around living active perennial grass roots. The quantity of the rhizosphere microorganism biomass determines the quantity of organic nitrogen mineralized into mineral nitrogen. Mineral nitrogen available at quantities of 100 lbs/ac or greater are needed to produce herbage biomass and calf weight gain per acre at ecosystems biological potential levels (Wight and Black 1972). Native grassland ecosystems are deficient in mineral nitrogen when it is not available at quantities of 100 lbs/ac and those grassland ecosystems degrade down to the level of available mineral nitrogen. Most traditional grazing practices are based on the use of the aboveground grass herbage biomass as forage for livestock and typically have available mineral nitrogen at 50 to 70 lbs/ac because these practices are supported by good land stewardship ethics. When livestock are removed from the grassland ecosystem, the degradation is severe and the rhizosphere biomass decreases to less than 15% of the reference rhizosphere biomass.

Restoration of degraded native grassland ecosystems requires the elevation of the rhizosphere microorganism biomass. The primary producer trophic level in the rhizosphere are achlorophyllous saprophytes and they can not fix carbon energy. Growth and development of all rhizosphere microorganisms is limited by access of short chain carbon energy (Manske 2011, 2014).

Healthy grass plants fix greater quantities of atmospheric carbon during photosynthesis than the plant needs for growth and development (Coyne et al. 1995). Some of this surplus short chain carbon energy can be moved from the grass tiller by exudation through the roots into the rhizosphere with partial defoliation by large grazing graminivores that remove 25% to 33% of the aboveground leaf weight while the lead tiller is at vegetative phenological growth stages between the three and a half new leaf stage and the flower stage. This growth stage occurs for 45 days each year from 1 June to 15 July for cool and warm season native grasses (Manske 1999, 2011, 2014).

The quantity of available mineral nitrogen below 100 lbs/ac determines the degree of grassland ecosystem degradation. The biomass of the rhizosphere microorganisms determines the quantity of soil organic nitrogen converted into mineral nitrogen by mineralization. Grassland ecosystems that are deficient in mineral nitrogen below 100 lbs/ac are low in rhizosphere microorganism biomass and the rhizosphere microorganisms are deficient in short chain carbon energy. The solution is to implement a twice-over rotation grazing strategy that activates the existing physiological mechanisms and biogeochemical processes of the native grassland ecosystems. The cattle partially defoliate grass lead tillers at vegetative growth stages that moves surplus carbon energy from the grass plants to the rhizosphere microorganisms that increase in biomass. The greater biomass of rhizosphere microorganisms convert greater quantities of organic nitrogen into mineral nitrogen. Mineral nitrogen available at quantities of 100 lbs/ac or greater produce greater herbage biomass at high quality. The cows milk at their genetic potential for most of the grazing season and their calves gain weight at their genetic potential producing greater calf weight per acre.

Acknowledgment

I am grateful to Sheri Schneider for assistance in production of this manuscript and for development of the tables and figure.

Table 1. Precipitation in inches for growing season months for 2006-2011, Richardton, North Dakota.. . . .

	Apr	May	Jun	Jul	Aug	Sep	Oct	Growing Season
Long-term mean (1971-2000)	1.75	2.49	3.39	2.27	1.88	1.60	1.41	14.79
2006	2.53	0.60	0.37	0.79	1.40	2.33	1.40	9.42
% of LTM	144.57	24.10	10.91	34.80	74.47	145.63	99.29	63.69
2007	1.04	3.57	2.22	0.44	1.57	1.29	0.62	10.75
% of LTM	59.43	143.37	65.49	19.38	83.51	80.63	43.97	72.68
2008	0.45	1.32	3.93	2.04	0.56	1.70	1.45	11.45
% of LTM	25.71	53.01	115.93	89.87	29.79	106.25	102.84	77.42
2009	0.59	0.85	3.09	2.82	0.53	1.67	2.08	11.63
% of LTM	33.71	34.14	91.15	124.23	28.19	104.38	147.52	78.63
2010	0.71	3.29	4.35	1.42	0.90	2.30	0.46	13.43
% of LTM	40.57	132.13	128.32	62.56	47.87	143.75	32.62	90.80
2011	2.01	4.94	1.76	4.06	2.07	0.96	1.35	17.15
% of LTM	114.86	198.39	51.92	178.85	110.11	60.00	95.74	115.96
2006-2011	1.22	2.43	2.62	1.93	1.17	1.71	1.23	12.31
% of LTM	69.71	97.59	77.28	85.02	62.23	106.88	87.23	83.23

Table 2. Animal Unit Equivalent, Herd Weight, and Stocking Rate used 2006 to 2011 compared to 2011 assessed values on Schnell Recreation Area.

	Animal Units		Herd Weight		Stocking Rate	
	Used	2011 Assessed	Used	2011 Assessed	Used	2011 Assessed
	AUE	175.53		175533		1.92
Year	#	%	lbs	%	ac/AUM	%
2006	113.82	64.84	126114	71.85	2.66	72.30
2007	132.56	75.52	144462	82.30	2.33	82.55
2008	171.51	97.71	188095	107.16	1.79	107.46
2009	145.74	83.03	139332	79.38	2.41	79.80
2010	154.90	88.25	152561	86.91	2.20	87.42
2011	73.45	41.84	66005	37.60	5.09	37.80

Table 3. Mean aboveground herbage production for domesticated grass in lbs/ac during the growing season on the silty ecological sites of nongrazed 4 on the Schnell Recreation Area, 2006-2011.

Domesticated Grass	30 May	15 Jun	30 Jun	15 Jul	15 Aug	15 Sep	15 Oct
2006	1684.81	1951.70	2071.58	1782.57	1676.96	1639.85	2029.48
2007	1489.28	1741.90	1772.40	1760.81	1742.97	1955.26	2076.58
2008	901.99	1296.61	1211.69	1394.37	1490.00	1476.44	1470.02
2009	1111.79	1868.92	1554.93	1711.21	1871.77	2201.46	2137.23
2010	1076.82	1106.79	1462.17	1532.81	1370.83	1609.17	2122.25
2011	911.98	1615.59	1894.61	2096.56	3109.87	3120.57	3588.69
Mean	1196.11	1596.92	1661.23	1713.06	1877.07	2000.46	2237.38

Table 4. Mean aboveground herbage production for native grass in lbs/ac during the growing season on the silty ecological sites of nongrazed 4 on the Schnell Recreation Area, 2006-2011.

Native Grass	30 May	15 Jun	30 Jun	15 Jul	15 Aug	15 Sep	15 Oct
2006	36.39	104.90	42.82	217.65	156.28	294.00	74.22
2007	2.14	313.98	434.41	386.24	303.28	126.66	90.98
2008	94.91	86.70	194.81	254.05	168.40	165.55	32.83
2009	111.32	291.15	401.75	427.44	439.58	234.06	229.06
2010	175.55	260.47	497.38	287.58	226.21	361.80	176.97
2011	104.89	219.07	249.76	344.67	490.24	10.00	59.94
Mean	87.53	212.71	303.49	319.61	297.33	198.68	110.67

Table 5. Mean aboveground herbage production for domesticated grass in lbs/ac during the growing season on the silty ecological sites of pasture 3 on the Schnell Recreation Area, 2006-2011.

Domesticated Grass	30 May	15 Jun	30 Jun	15 Jul	15 Aug	15 Sep	15 Oct
2006	1529.96	1634.14	1460.74	1241.66	1158.17	1620.59	1771.87
2007	1089.67	1767.94	1596.32	1424.35	1341.57	1111.08	759.98
2008	536.63	1299.61	923.40	785.67	598.00	439.58	480.25
2009	1014.74	1183.15	1587.05	1437.19	1460.74	1502.13	1382.96
2010	1009.74	1345.14	1569.21	1538.52	1126.06	1081.10	1000.47
2011	1155.32	1972.39	2312.06	2156.50	2931.47	2425.53	2390.56
Mean	1056.01	1533.73	1574.80	1430.65	1436.00	1363.34	1297.68

Table 6. Mean aboveground herbage production for native grass in lbs/ac during the growing season on the silty ecological sites of pasture 3 on the Schnell Recreation Area, 2006-2011.

Native Grass	30 May	15 Jun	30 Jun	15 Jul	15 Aug	15 Sep	15 Oct
2006	34.97	72.07	88.49	227.64	89.91	282.58	57.08
2007	9.99	0.89	18.55	135.58	71.36	34.97	33.54
2008	34.97	78.50	37.10	96.34	120.60	24.26	11.42
2009	14.28	92.77	112.04	186.25	44.24	59.23	9.28
2010	18.56	33.54	127.73	137.01	7.13	32.11	34.96
2011	39.96	58.51	73.50	183.40	158.42	8.56	73.50
Mean	25.46	56.05	76.24	161.04	81.94	73.62	36.63

Table 7. Mean aboveground herbage production for domesticated grass in lbs/ac during the growing season on the silty ecological sites of pastures 1 and 2 on the Schnell Recreation Area, 2006-2011.

Domesticated Grass	30 May	15 Jun	30 Jun	15 Jul	15 Aug	15 Sep	15 Oct
2006	1066.48	1114.64	898.78	738.58	803.52	832.06	1228.82
2007	823.32	1197.42	1238.10	1347.01	1196.62	1204.20	1085.86
2008	241.56	489.53	333.61	427.09	159.14	387.84	136.66
2009	382.13	530.21	665.43	705.75	606.92	411.04	531.63
2010	516.29	613.34	839.55	661.51	704.32	718.24	746.79
2011	882.01	1226.68	943.38	1226.32	1396.16	1583.13	1570.99
Mean	651.97	861.97	819.81	851.04	811.11	856.09	883.46

Table 8. Mean aboveground herbage production for domesticated grass in lbs/ac during the growing season on the silty ecological sites of pastures 1 and 2 on the Schnell Recreation Area, 2006-2011.

Native Grass	30 May	15 Jun	30 Jun	15 Jul	15 Aug	15 Sep	15 Oct
2006	53.53	234.06	228.71	296.50	225.15	218.72	185.90
2007	131.57	258.06	573.73	308.73	583.82	351.90	188.96
2008	200.52	199.46	293.29	259.04	260.47	181.97	87.42
2009	265.46	406.40	399.26	591.94	689.34	357.16	400.33
2010	301.50	269.75	563.39	474.19	223.71	322.19	272.60
2011	270.81	569.10	810.29	564.82	675.78	628.32	450.28
Mean	203.90	322.81	478.11	415.87	443.05	343.38	264.25

Table 9. Mean annual domesticated grass herbage biomass (lbs/ac) and basal cover (%) on the silty ecological sites of three treatments on the Schnell Recreation Area, 2006-2011.

Management Treatment	2006	2007	2008	2009	2010	2011
Nongrazed 4						
Domesticated Herbage	1858.69ax	1841.65ax	1389.86ay	1840.92ax	1534.00axy	2570.98ax
Basal Cover	10.55	17.30	12.55	6.75	17.45	13.20
Pasture 3						
Domesticated Herbage	1481.20ay	1333.54byz	754.42bz	1425.24by	1276.75ayz	2364.75ax
Basal Cover	4.80	10.75	12.55	17.30	21.65	15.90
Pastures 1 & 2						
Domesticated Herbage	936.07bx	1211.54bx	322.31bz	575.16cy	713.96by	1324.44bx
Basal Cover	4.80	5.35	4.08	6.20	6.88	6.88

Means in the same column and followed by the same letter (a, b, c) are not significantly different (P<0.05).
Means in the same row and followed by the same letter (x, y, z) are not significantly different (P<0.05).

Table 10. Mean annual native grass herbage biomass (lbs/ac) and basal cover (%) on the silty ecological sites of three treatments on the Schnell Recreation Area, 2006-2011.

Management Treatment	2006	2007	2008	2009	2010	2011
Nongrazed 4						
Native Grass Herbage	148.31axy	275.93axy	150.39aby	337.17ax	301.14axy	228.95axy
Basal Cover	1.25	1.50	2.20	0.85	2.20	0.75
Pasture 3						
Native Grass Herbage	136.30ax	49.15bx	61.37ax	83.97bx	62.08bx	92.65ax
Basal Cover	0.10	0.70	0.25	0.00	0.00	0.00
Pastures 1 & 2						
Native Grass Herbage	231.51ay	377.53ax	213.61by	474.07axz	354.31ayz	616.43bx
Basal Cover	2.55	8.65	6.80	7.60	7.65	6.45

Means in the same column and followed by the same letter (a, b, c) are not significantly different (P<0.05).
Means in the same row and followed by the same letter (x, y, z) are not significantly different (P<0.05).

Table 11. Mean annual domesticated and native grass herbage biomass (lbs/ac) and basal cover (%) on the silty ecological sites of three treatments on the Schnell Recreation Area, 2006-2011.

	Domesticated Grass		Native Grass	
Management Treatment	Herbage lbs/ac	Basal Cover %	Herbage lbs/ac	Basal Cover %
Nongrazed 4	1847.69a	12.97a	240.42a	1.46b
Pasture 3	1434.37b	13.83a	80.92b	0.18c
Pastures 1 & 2	847.25c	5.70b	377.91a	6.62a

Means in the same column and followed by the same letter (a, b, c) are not significantly different ($P<0.05$).

Table 12. Rhizosphere weight (kg) per cubic meter of soil at the Schnell Recreation Area, 2006-2011.

Nongrazed NR 4		May	Jul	Oct	Mean
2006	kg/m^3	52.23	74.41	66.09	64.24x
2007	kg/m^3	55.20	93.19	85.06	77.82x
2008	kg/m^3	69.35	70.62	72.05	70.67z
2009	kg/m^3	82.54	83.22	-	82.88z
2010	kg/m^3	87.74	96.54	76.27	86.85z
2011	kg/m^3	123.07	131.65	136.94	130.56z

Table 13. Rhizosphere weight (kg) per cubic meter of soil at the Schnell Recreation Area, 2006-2011.

Pasture NR 3		May	Jul	Oct	Mean
2006	kg/m^3	51.25	58.51	52.65	54.14x
2007	kg/m^3	37.53	84.52	92.96	71.67x
2008	kg/m^3	92.89	105.98	85.78	94.88y
2009	kg/m^3	128.35	97.74	-	113.05y
2010	kg/m^3	107.17	177.92	134.72	139.94y
2011	kg/m^3	164.54	167.55	169.06	167.05y

Table 14. Rhizosphere weight (kg) per cubic meter of soil at the Schnell Recreation Area, 2006-2011.

Pastures NR 1 & 2		May	Jul	Oct	Mean
2006	kg/m^3	91.36	86.10	72.38	83.28x
2007	kg/m^3	73.26	93.03	110.39	92.23x
2008	kg/m^3	109.24	129.72	128.86	122.61x
2009	kg/m^3	157.01	123.63	-	140.32x
2010	kg/m^3	200.02	193.86	155.13	183.00x
2011	kg/m^3	179.25	244.07	219.70	214.34x

Values in the “Mean” column of these three tables, in the same “year” row, and followed by the same letter are not significantly different ($P<0.05$).

Table 15. Grass relative composition (%) and rhizosphere weight (kg/m^3) for the nongrazed control 4 and grazed pasture NR 3 at the Schnell Recreation Area, 2006-2011.

	Grass Basal Cover Relative Composition (%)					
	Nongrazed Control NR 4			Grazed Pasture NR 3		
	Smooth Bromegrass	Kentucky Bluegrass	% Difference	Smooth Bromegrass	Kentucky Bluegrass	% Difference
2006	0.0	76.95	100.00	52.17	31.30	-40.00
2007	0.0	78.08	100.00	46.94	40.82	-13.04
2008	0.0	65.31	100.00	37.77	52.52	39.05
2009	0.0	80.53	100.00	44.44	49.32	10.98
2010	0.0	83.10	100.00	41.79	50.53	20.91
2011	0.28	85.96	100.00	28.82	64.71	124.53

	Rhizosphere Weight (kg/m^3)		
	Nongrazed Control NR 4	Grazed Pasture NR 3	% Difference
Pregrazing 2006	52.23	51.25	-1.88
2006	64.24x	54.14x	-15.72
2007	77.82x	71.67x	-7.90
2008	70.67z	94.88y	34.26
2009	82.88z	113.05y	36.40
2010	86.85z	139.94y	61.13
2011	130.56z	167.05y	27.95

Means in the same row and followed by the same letter (x, y, z) are not significantly different ($P<0.05$).

Table 16. Rhizosphere weight (kg/m^3) for the nongrazed control 4 and grazed pastures 1 & 2 during six years of twice-over rotation management, 2006-2011.

	Nongrazed Control 4 kg/m^3	Grazed Pastures 1 & 2 kg/m^3	% Difference
Pregrazing 2006	52.23	77.99	49.32
2006	64.24x	83.28x	29.64
2007	77.82x	92.22x	18.50
2008	70.67z	122.61x	73.50
2009	82.88z	140.32x	69.31
2010	86.85z	183.00x	110.71
2011	130.56z	214.34x	64.17

Means in the same row and followed by the same letter (x, y, z) are not significantly different ($P<0.05$).

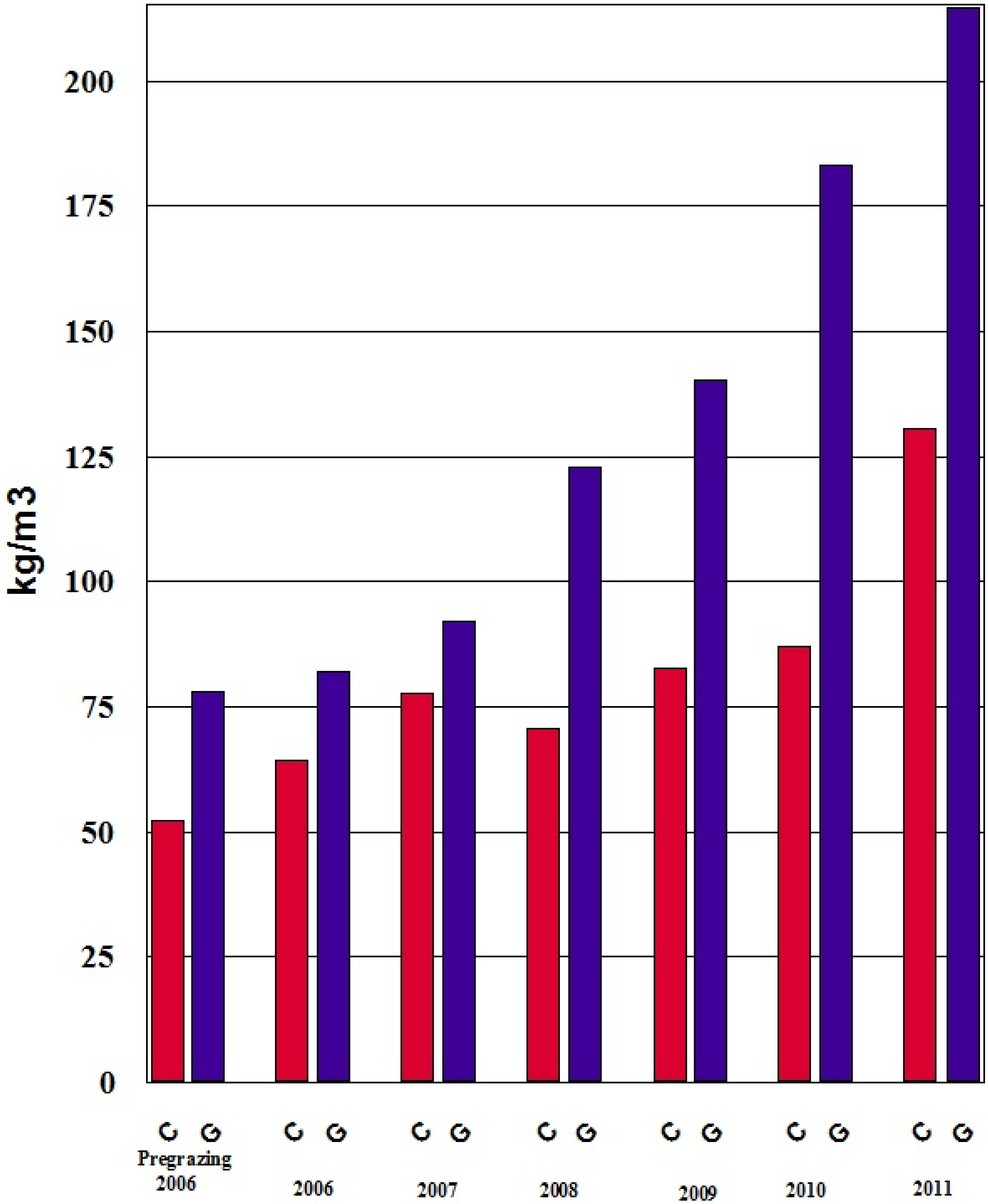

Figure 1. Rhizosphere weight (kg/m3) for the control pasture (red) and grazed pastures (blue) during six years of twice-over rotation management, 2006-2011.

Table 17. Mean domesticated and native grass herbage biomass in lbs/ac during the growing season on the silty ecological sites of three treatments on the Schnell Recreation Area, 2013-2014.

Management Treatment	30 May	15 Jun	30 Jun	15 Jul	15 Aug	15 Sep	15 Oct
Nongrazed 4							
Domesticated	866.67	1736.19	1819.32	2144.37	2419.47	2651.74	2641.75
Native Grass	46.03	28.19	176.98	163.06	398.19	292.94	156.28
Pasture 3							
Domesticated	968.36	1750.46	2137.23	2465.85	1876.42	2308.50	2855.08
Native Grass	26.41	18.91	64.58	43.18	63.16	90.27	58.88
Pastures 1 & 2							
Domesticated	598.00	943.03	633.15	1332.83	1485.72	1014.92	934.81
Native Grass	72.08	269.74	588.19	418.89	313.63	620.48	303.82

Table 18. Mean annual domesticated and native grass herbage biomass (lbs/ac) and basal cover (%) on the silty ecological sites of three treatments on the Schnell Recreation Area, 2013-2014.

	Domesticated Grass		Native Grass	
Management Treatment	Herbage lbs/ac	Basal Cover %	Herbage lbs/ac	Basal Cover %
Nongrazed 4	2235.47a	19.03a	202.61ab	2.18b
Pasture 3	2232.26a	20.20a	56.50b	0.08c
Pastures 1 & 2	1057.41b	9.17b	419.13a	7.65a

Means in the same column and followed by the same letter (a, b, c) are not significantly different (P<0.05).

Table 19. Mean mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac with quantities available in the soil and estimated quantities transformed to organic nitrogen during the growing season on silty ecological sites of the Schnell Recreation Area nongrazed 4, 2013-2014.

Soil Depth (inches)	May	Jun	Jul	Aug	Sep	Oct
NO_3 nitrate						
0-6 Available	13.25	13.00	8.75	6.50	5.25	7.50
0-6 Transformed		-0.25	-4.50	-6.75	-8.00	-5.75
6-12 Available	9.75	6.75	6.00	5.00	4.25	4.25
6-12 Transformed		-3.00	-3.75	-4.75	-5.50	-5.50
12-24 Available	7.69	10.00	19.00	11.00	4.00	4.00
12-24 Transformed		+2.31	+11.31	+3.31	-3.69	-3.69
0-24 Available	30.69	29.75	33.75	22.50	13.50	15.75
0-24 Transformed		-0.94	+3.06	-8.19	-17.19	-14.94
NH_4 ammonium						
0-6 Available	19.99	11.83	16.24	12.40	13.79	13.63
0-6 Transformed		-8.16	-3.75	-7.59	-6.20	-6.36
6-12 Available	12.32	11.18	12.24	11.26	10.85	12.16
6-12 Transformed		-1.14	-0.08	-1.06	-1.47	-0.16
12-24 Available	8.24	13.14	16.16	12.07	12.40	3.65
12-24 Transformed		+4.90	+7.92	+3.83	+4.16	-4.59
0-24 Available	40.55	36.15	44.64	35.73	37.04	29.44
0-24 Transformed		-4.40	+4.09	-4.82	-3.51	-11.11
$NO_3 + NH_4$						
0-6 Available	33.24	24.83	24.99	18.90	19.04	21.13
0-6 Transformed		-8.41	-8.25	-14.34	-14.20	-12.11
6-12 Available	22.07	17.93	18.24	16.26	15.10	16.41
6-12 Transformed		-4.14	-3.83	-5.81	-6.97	-5.66
12-24 Available	15.93	23.14	35.16	23.07	16.40	7.65
12-24 Transformed		+7.21	+19.23	+7.14	+0.47	-8.28
0-24 Available	71.24	65.90	78.39	58.23	50.54	45.19
0-24 Transformed		-5.34	+7.15	-13.01	-20.70	-26.05

Table 20. Mean mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac with quantities available in the soil and estimated quantities transformed to organic nitrogen during the growing season on silty ecological sites of the Schnell Recreation Area pasture 3, 2013-2014.

Soil Depth (inches)	May	Jun	Jul	Aug	Sep	Oct
NO_3 nitrate						
0-6 Available	13.25	9.50	6.25	5.25	7.00	5.00
0-6 Transformed		-3.75	-7.00	-8.00	-6.25	-8.25
6-12 Available	6.00	8.00	4.75	3.75	5.25	4.00
6-12 Transformed		+2.00	-1.25	-2.25	-0.75	-2.00
12-24 Available	9.07	7.25	8.00	3.00	7.00	3.00
12-24 Transformed		-1.82	-1.07	-6.07	-2.07	-6.07
0-24 Available	28.32	24.75	19.00	12.00	19.25	12.00
0-24 Transformed		-3.57	-9.32	-16.32	-9.07	-16.32
NH_4 ammonium						
0-6 Available	14.12	15.91	11.02	11.10	14.36	12.97
0-6 Transformed		+1.79	-3.10	-3.02	+0.24	-1.15
6-12 Available	11.42	13.30	9.88	9.79	10.04	10.37
6-12 Transformed		+1.88	-1.54	-1.63	-1.38	-1.05
12-24 Available	10.60	12.57	10.28	10.45	12.40	3.30
12-24 Transformed		+1.97	-0.32	-0.15	+1.80	-7.30
0-24 Available	36.14	41.78	31.18	31.34	36.80	26.64
0-24 Transformed		+5.64	-4.96	-4.80	+0.66	-9.50
$NO_3 + NH_4$						
0-6 Available	27.37	25.41	17.27	16.35	21.36	17.97
0-6 Transformed		-1.96	-10.10	-11.02	-6.01	-9.40
6-12 Available	17.42	21.30	14.63	13.54	15.29	14.37
6-12 Transformed		+3.88	-2.79	-3.88	-2.13	-3.05
12-24 Available	19.66	19.82	18.28	13.45	19.40	6.30
12-24 Transformed		+0.16	-1.38	-6.21	-0.26	-13.36
0-24 Available	64.45	66.53	50.18	43.34	56.05	38.64
0-24 Transformed		+2.08	-14.27	-21.11	-8.40	-25.81

Table 21. Mean mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac with quantities available in the soil and estimated quantities transformed to organic nitrogen during the growing season on silty ecological sites of the Schnell Recreation Area pastures 1 & 2, 2013-2014.

Soil Depth (inches)	May	Jun	Jul	Aug	Sep	Oct
NO_3 nitrate						
0-6 Available	29.63	10.75	7.63	15.75	9.63	7.75
0-6 Transformed		-18.88	-22.00	-13.88	-20.00	-21.88
6-12 Available	11.38	4.88	4.88	4.38	4.25	4.63
6-12 Transformed		-6.50	-6.50	-7.00	-7.13	-6.75
12-24 Available	9.50	6.00	5.00	8.00	5.50	4.00
12-24 Transformed		-3.50	-4.50	-1.50	-4.00	-5.50
0-24 Available	50.50	21.63	17.51	28.13	19.38	16.38
0-24 Transformed		-28.87	-32.99	-22.37	-31.12	-34.12
NH_4 ammonium						
0-6 Available	20.21	14.44	14.77	16.88	19.00	14.81
0-6 Transformed		-5.77	-5.44	-3.33	-1.21	-5.40
6-12 Available	14.20	10.32	13.96	15.46	12.57	12.52
6-12 Transformed		-3.88	-0.24	+1.26	-1.63	-1.68
12-24 Available	14.45	11.67	14.44	11.99	13.30	3.88
12-24 Transformed		-2.78	-0.01	-2.46	-1.15	-10.57
0-24 Available	48.85	36.42	43.17	44.33	44.87	31.21
0-24 Transformed		-12.43	-5.68	-4.52	-3.98	-17.64
$NO_3 + NH_4$						
0-6 Available	49.83	25.19	22.39	32.63	28.62	22.56
0-6 Transformed		-24.64	-27.44	-17.20	-21.21	-27.27
6-12 Available	25.57	15.20	18.83	19.84	16.82	17.15
6-12 Transformed		-10.37	-6.74	-5.73	-8.75	-8.42
12-24 Available	23.95	17.67	19.44	19.99	18.80	7.88
12-24 Transformed		-6.28	-4.51	-3.96	-5.15	-16.07
0-24 Available	99.35	58.06	60.66	72.46	64.24	47.59
0-24 Transformed		-41.29	-38.69	-26.89	-35.11	-51.76

Table 22. Mean monthly (May-October) mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac with quantities available in the soil and estimated quantities transformed to organic nitrogen during the growing season on silty ecological sites of the Schnell Recreation Area treatments, 2013-2014.

Soil Depth (inches)	Nongrazed 4	Pasture 3	Pastures 1 & 2
NO_3 nitrate			
0-6 Available	9.04	7.71	13.52
0-6 Transformed	-5.05	-6.65	-19.33
6-12 Available	6.00	5.29	5.73
6-12 Transformed	-4.50	-0.85	-6.78
12-24 Available	9.28	6.22	6.33
12-24 Transformed	+1.91	-3.42	-3.80
0-24 Available	24.32	19.22	25.59
0-24 Transformed	-7.64	-10.92	-29.89
NH_4 ammonium			
0-6 Available	14.65	13.25	16.69
0-6 Transformed	-6.41	-1.05	-4.23
6-12 Available	11.67	10.80	13.17
6-12 Transformed	-0.78	-0.74	-1.23
12-24 Available	10.94	9.93	11.62
12-24 Transformed	+3.24	-0.80	-3.39
0-24 Available	37.26	33.98	41.48
0-24 Transformed	-3.95	-2.59	-8.85
$NO_3 + NH_4$			
0-6 Available	23.69	20.96	30.20
0-6 Transformed	-11.46	-7.70	-23.55
6-12 Available	17.67	16.09	18.90
6-12 Transformed	-5.28	-1.59	-8.00
12-24 Available	20.23	16.15	17.96
12-24 Transformed	+5.15	-4.21	-7.19
0-24 Available	61.58	53.20	67.06
0-24 Transformed	-11.59	-13.50	-38.75

Table 23. May available and mean monthly transformed mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac during the growing season on silty ecological sites of the Schnell Recreation Area treatments, 2013-2014.

Soil Depth (inches)	Nongrazed 4		Pasture 3		Pasture 1 & 2	
	May Available	Mean Monthly Transformed	May Available	Mean Monthly Transformed	May Available	Mean Monthly Transformed
NO_3 nitrate						
0-6	13.25	5.05	13.25	6.65	29.63	19.33
6-12	9.75	4.50	6.00	0.85	11.38	6.78
12-24	7.69	+1.91	9.07	3.42	9.50	3.80
0-24	30.69	7.64	28.32	10.92	50.50	29.89
NH_4 ammonium						
0-6	19.99	6.41	14.12	1.05	20.21	4.23
6-12	12.32	0.78	11.42	0.74	14.20	1.23
12-24	8.24	+3.24	10.60	0.80	14.45	3.39
0-24	40.55	3.95	36.14	2.59	48.85	8.85
$NO_3 + NH_4$						
0-6	33.24	11.46	27.37	7.70	49.83	23.55
6-12	22.07	5.28	17.42	1.59	25.57	8.00
12-24	15.93	+5.15	19.66	4.21	23.95	7.19
0-24	71.24	11.59	64.45	13.50	99.35	38.75

Literature Cited

Barker, W.T., and W.C. Whitman. 1988. Vegetation of the Northern Great Plains. Rangelands 10:266-272.

Bloem, J., D.W. Hopkins, and A. Benedetti (eds.). 2006. Microbiological methods for assessing soil quality. CAB International, Oxfordshire, UK. 307p.

Brady, N.C. 1974. The nature and properties of soils. MacMillan Publishing Co., Inc., New York, NY. 639p.

Briske, D.D., and J.H. Richards. 1995. Plant response to defoliation: a physiological, morphological, and demographic evaluation. p. 635-710. *in* D.J. Bedunah and R.E. Sosebee (eds.). Wildland plants: physiological ecology and developmental morphology. Society for Range Management, Denver, CO.

Cook, C.W., and J. Stubbendieck. 1986. Range research: basic problems and techniques. Society for Range Management, Denver, CO. 317p.

Coyne, P.I., M.J. Trlica, and C.E. Owensby. 1995. Carbon and nitrogen dynamics in range plants. p. 59-167. *in* D.J. Bedunah and R.E. Sosebee (eds.). Wildland plants: physiological ecology and developmental morphology. Society for Range Management, Denver, CO.

Crider, F.J. 1955. Root-growth stoppage resulting from defoliation of grass. USDA Technical Bulletin 1102.

Goetz, H. 1975. Availability of nitrogen and other nutrients on four fertilized range sites during the active growing season. Journal of Range Management 28:305-310.

Grant, S.A., G.T. Barthram, L. Torvell, J. King, and H.K. Smith. 1983. Sward management, lamina turnover and tiller population density in continuously stocked *Lolium perenne*-dominated swards. Grass and Forage Science 38:333-344.

Kochy, M. 1999. Grass-tree interactions in western Canada. Ph.D. Dissertation. University of Regina. Regina, Saskatchewan, Canada.

Kochy, M., and S.D. Wilson. 2000. Competitive effects of shrubs and grasses in prairie. Oikos 91:385-395.

Langer, R.H.M. 1972. How grasses grow. Edward Arnold, London, Great Britain.

Legg, J.O. 1975. Influence of plants on nitrogen transformation in soils. pg. 221-227. *in* M.K. Wali (ed.). Prairie: A multiple view. University of North Dakota Press. Grand Forks, ND.

Manske, L.L. 1999. Can native prairie be sustained under livestock grazing? Provincial Museum of Alberta. Natural History Occasional Paper No. 24. Edmonton, Alberta. p.99-108.

Manske, L.L. 2007a. Effects on vegetation, endomycorrhizal fungi, and soil mineral nitrogen from prescribed burning treatments repeated every-other-year in mixed grass prairie invaded by western snowberry. NDSU Dickinson Research Extension Center. Summary Range Research Report DREC 07-3044. Dickinson, ND. 19p.

Manske, L.L. 2011b. Biology of defoliation by grazing. NDSU Dickinson Research Extension Center. Range Management Report DREC 11-1067b. Dickinson, ND. 25p.

Manske, L.L., and J.A. Urban. 2012. Humane soil beastie catcher: Its fabrication and use. NDSU Dickinson Research Extension Center. Range Research Report DREC 12-1079. Dickinson, ND. 9p.

Manske, L.L. 2014. Grazingland management based on native rangeland ecosystem mechanisms and processes. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 14-3062. Dickinson, ND. 18p.

Mosteller, F., and R.E.K. Rourke. 1973. Sturdy statistics. Addison-Wesley Publishing Co., MA. 395p.

Peltzer, D.A., and M. Kochy. 2001. Competitive effects of grasses and woody plants in mixed grass prairie. Journal of Ecology 89:519-527.

Power, J.F. 1970. Nitrogen management of semiarid grasslands in North America. Proceedings of the XI International Grassland Congress. 1970:468-471.

Shiflet, T.N. (ed.). 1994. Rangeland cover types. Society for Range Management. Denver, CO. 152p.

Whitman, W.C. 1975. Native range fertilization and interseeding study. Annual Report. Dickinson Experiment Station. Dickinson, ND. p. 11-16.

Wight, J.R., and A.L. Black. 1972. Energy fixation and precipitation use efficiency in a fertilized rangeland ecosystem of the Northern Great Plains. Journal of Range Management 25:376-380.

Wight, J.R., and A.L. Black. 1979. Range fertilization: plant response and water use. Journal of Range Management 32:345-349.

Evaluation of the Importance of Mineral Nitrogen Available at or Above Threshold Quantities in Maintaining Productivity at Potential Levels on Mixed Grass Prairie Ecosystems

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Before the 1862 Homestead Act of the United States and the 1872 Dominion Lands Act of Canada, agricultural producers of the world did not own the land where they lived and farmed. Part of the commodities produced were used as rent and part of the crops and animals they raised belonged to the producer. These produced commodities were more important for family survival than the land. The land management practices that were developed were based on the use. All of the renewable natural resources (rangelands, grasslands, croplands, forestlands, and fisheries) are still traditionally managed for their use. As a result, the productivity of the worlds natural resources are gradually deteriorating. Traditional management for the "use" does not take the renewable natural resources back to the "good as new" condition.

Traditional management practices neglect the vital cryptobiotic microorganism component. The microorganisms are the renewable portion of natural resource ecosystems. Microbes cycle essential elements from the unusable organic forms into the usable mineral forms. Reductions in microorganism quantity or activity translates into reduction in usable essential elements. Diminishment in the quantity of available essential elements is the deterioration in productivity of natural resources (Bloem et al. 2006).

The major essential elements in native grassland ecosystems are carbon, hydrogen, nitrogen, and oxygen. The minor essential elements are the macronutrients and the micronutrients, and the presence of sunlight energy is essential.

Radiant light from the sun is the ecosystem input source for energy. Radiant energy from the sun is necessary for photosynthesis. Sunlight is not limiting on rangelands even with about 30% cloud cover, except the intensity of sunlight can be greatly reduced by shading from taller grasses and shrubs (Kochy 1999). The light levels penetrating the leaf canopy can be about 20% of the light levels above the canopy (Peltzer and Kochy 2001).

Atmospheric carbon dioxide (CO_2) is the ecosystem input source for carbon. Atmospheric carbon dioxide which composes about 0.03% of the gasses in the atmosphere is not limiting on rangelands. The carbon dioxide is fixed with hydrogen from soil water during photosynthesis which converts energy from sunlight into chemical energy and assimilates simple carbohydrates. Capturing energy by fixing carbon has a relatively low impact on the plant organisms that possess chlorophyll and has low biological costs to the ecosystem resources (Manske 2011a).

Soil water (H_2O) is infiltrated precipitation water and is the ecosystem input source for hydrogen. Soil water is absorbed through the roots and distributed throughout the plant within the xylem vascular tissue. Water deficiency conditions in western North Dakota have a long-term periodicity rate at 32.7%, for a mean of 2.0 months with water deficiency during the 6.0 month perennial plant growing season (Manske et al. 2010). Water is necessary for plant growth. However, deficiencies in mineral nitrogen limit herbage production more often than water deficiencies in temperate grasslands (Tilman 1990).

Carbon dioxide, water, and nitrogen oxides are the ecosystem input sources for oxygen. Atmospheric oxygen composes about 28% of the gasses in the atmosphere. The oxygen cycle is closely linked to the carbon cycle and the water, or hydrological cycle. Oxygen is vital for all organisms that carry out aerobic respiration. Oxygen is not limiting on rangeland ecosystems.

Wet deposition of nitrogen oxides (NO, N_2O) following lightning discharges is the ecosystem input source for nitrogen (Manske 2009a). The source of nitrogen for plant growth is mineral nitrogen (NO_3 nitrate, NH_4 ammonium) converted from soil organic nitrogen by rhizosphere microorganisms. Low quantities of available soil mineral nitrogen below 100 lbs/ac is the major limiting factor of herbage growth on rangelands

(Wight and Black 1979). The quantity of rhizosphere microorganism biomass is the limiting factor in rangeland ecosystems with low mineral nitrogen available. Biomass and activity of microorganisms in the rhizosphere are limited by access to short chain carbon energy which can be exudated from grass lead tillers with partial defoliation by grazing graminivores when grass tillers are at vegetative growth stages between the three and a half new leaf stage and the flower stage. Transforming nitrogen from organic nitrogen into mineral nitrogen and back to organic nitrogen is complex and has a great impact on many organisms at multiple trophic levels and has high biological costs on the ecosystem resources (Manske 2011a, 2014a).

Failure of traditional management practices to replenish ecosystem available essential elements at quantities equal to or greater than the annual amount of essential elements needed for ecosystem functioning at biological potential levels that have resulted from incremental decreases in microorganism biomass is the primary factor causing deterioration in productivity of natural resources.

The objectives of this study are to show the importance of purposeful management of the quantity of available mineral nitrogen to be at 100 lbs/ac or greater on native grassland ecosystems in order to renew the ecosystem condition back to "good as new" and to maintain vegetation and livestock production at potential levels.

Study Area

The native rangeland study sites were on the Dickinson Research Extension Center (DREC) ranch, operated by North Dakota State University (NDSU), and located in Dunn County, 20 miles north of Dickinson, in western North Dakota, USA.

Long-term mean annual temperature was 42.1° F (5.7° C). January was the coldest month, with a mean temperature of 14.6° F (-9.7° C). July and August were the warmest months, with mean temperatures of 69.6° F (20.9° C) and 68.5° F (20.3° C), respectively. Long-term mean annual precipitation was 17.2 inches (437.7 mm). The amount of precipitation received during the perennial plant growing season (April to October) was 14.5 inches (368.8 mm) and was 84.3% of annual precipitation (Manske 2015a).

Soils were primarily Typic Haploborolls. The native rangeland vegetation was the Wheatgrass-Needlegrass Type (Barker and Whitman 1988, Shiflet 1994) of the mixed grass prairie. The dominant native range grasses were western wheatgrass (Agropyron smithii) (Pascopyrum smithii), needle and thread (Stipa comata) (Hesperostipa comata), blue grama (Bouteloua gracilis), and threadleaf sedge (Carex filifolia).

Management Treatments

Three management treatments were evaluated (1) long-term nongrazed control (NG), (2) 4.5-month seasonlong (4.5-m SL), and (3) 4.5-month twice-over rotation system (4.5-m TOR).

The long-term nongrazed control management treatment was designed with two large replicated exclosures and had not been grazed, mowed, or burned for more than 30 years before the initiation of the research treatments in 1983.

The 4.5-month seasonlong grazing system was designed with two replicated pastures. Each pasture was grazed for 137 days from early June to mid October stocked at 2.86 acres per cow-calf pair per month.

The 4.5-month twice-over rotation grazing system was designed with two replicated systems each with three pastures. Each pasture was grazed two times per growing season. Each system was grazed 137 days from early June to mid October stocked at 2.20 acres per cow-calf pair per month.

Procedure

Temperature and precipitation data was taken from historical climatological data collected at the Dickinson Research Extension Center (DREC) ranch, latitude 47° 14' N, longitude 102° 50' W, Dunn County, near Manning, North Dakota, USA, 2013-2014.

Available soil water (reported as inches of water) was determined by the gravimetric procedure from two replications of soil core samples collected at silty ecological sites of each management treatment with the 1 inch Veihmeyer soil tube at incremental depths of 0-6, 6-12, 12-24, 24-36, and 36-48 inches on monthly periods during April to October, 2013-2014.

Aboveground herbage biomass was collected by the standard clipping method (Cook and Stubbendieck 1986). The herbage biomass was partially defoliated by the selected grazing management on the seasonlong and twice-over treatments. The nongrazed exclosure areas had no defoliation treatments. The reported herbage biomass values represent the residuum vegetation and the regrowth vegetation resulting from the respective treatments. Clipped herbage material was collected monthly (May to October) from five 0.25 m^2 quadrats (frames) at two replicated silty ecological sample sites for each of the study treatments during 2013 and 2014. The herbage material in each frame was hand clipped to ground level and sorted in the field by biotype categories: domesticated grass, cool season grass, warm season grass, sedges, forbs, standing dead, and litter. The herbage of each biotype category from each frame was placed in labeled paper bags of known weight, oven dried at 140° F (60° C), and weighed. Herbage biomass in pounds per acre for each category were determined from the clipping data. The native grass (cool and warm season grass) and domesticated grass herbage biomass weights were reported for this study. The domesticated grass weights were too small to measure (<10 g/¼ m^2).

Plant species basal cover for individual species was determined by the ten-pin point frame method (Cook and Stubbendieck 1986), with 2000 points collected along permanent transect lines at two replicated silty ecological sample sites for each of the study treatments annually during peak growth between mid July and mid August, 2013-2014. Basal cover plant species data were sorted into biotype categories: domesticated grass, cool season grass, warm season grass, upland sedges, forbs, and litter. The native grass (cool and warm season grass) and domesticated grass percent basal cover were reported for this study.

Rhizosphere volume associated with western wheatgrass (Agropyron smithii) roots was determined from two replicated intact soil cores from silty ecological sites on each of the study treatments collected monthly (June to September) during 2002. Plastic PVC pipe 3 inches (7.62 cm) in diameter and 4 inches (10.16 cm) long was forced into sample site soil. Intact soil-plant cores and pipe were excavated and transported to the laboratory. The soil matrix of collected soil cores was carefully removed from between the rhizospheres around the roots of western wheatgrass plants. The exposed rhizospheres were sprayed with a clear acrylic coating to prevent damage during further handling. The length and diameter of the rhizosphere around each root of every plant, including associated tillers, were measured in inches with a vernier caliper, then converted to metric system values. The length and diameter measurements were used to determine the volume of each rhizosphere. Data were analyzed on a per-plant basis, as a total of all plants per replication, and reported as a mean of the two replications per sample period.

Soil weight of silty soil in southwestern North Dakota was determined from average silty soil bulk density from analysis of comparable soils (Anonymous circa early 1980's) at incremental depths of 0-6, 6-12, 12-24, 24-36, and 36-48 inches. Weight of soil organic matter (SOM) was determined from the soil weight of silty soil and percent soil organic matter from analysis conducted by the North Dakota State University Soil Testing Laboratory of soil core samples from four replicated cores on silty ecological sites of each management treatment at incremental depths of 0-6, 6-12, 12-24, 24-36, and 36-48 inches collected during June of 2013 and 2014. Weight of soil organic carbon (SOC) was determined from the soil weight of silty soil and percent soil organic matter multiplied by 0.58 (58% organic carbon content of soil organic matter) (Anonymous nd, NRCS Staff 2009, Pluske et al. 2015) of soil core samples on silty ecological sites of each management treatment at incremental depths of 0-6, 6-12, 12-24, 24-36, and 36-48 inches. Weight of soil organic nitrogen (SON) was determined from the soil weight of silty soil and percent soil organic matter multiplied by 0.058 (estimated 5.8% organic nitrogen content of soil organic matter) of soil core samples on silty ecological sites of each management treatment at incremental depths of 0-6, 6-12, 12-24, 24-36, and 36-48 inches (table 1).

Soil mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), was determined from four replicated soil core samples collected at silty ecological sites inside protective exclosures for the nongrazed treatment, and outside exclosures exposed to selected treatments for the seasonlong and twice-over rotation grazing management systems with the 1 inch Veihmeyer soil tube at incremental depths of 0-6, 6-12, 12-24 inches on monthly periods during May to October of 2013 and 2014. Analysis of soil core samples for available mineral nitrogen (NO_3 and NH_4) was conducted by the North Dakota State University Soil Testing Laboratory. Mean available mineral nitrogen was reported as pounds per acre.

Transformation (immobilization) of nitrate (NO_3) and of ammonium (NH_4) was determined by the net mineralization measurement of the nitrogen balance equation of a soil-plant system (table 1) (Bloem et al. 2006). The quantity of mineral nitrogen in a soil is the net difference between the total quantity of organic nitrogen mineralized by soil microorganisms and the quantity of mineral nitrogen immobilized by plants and soil microbes (Brady 1974, Legg 1975). The general nitrogen balance equation is simply: the quantity of nitrogen at time 2 minus the quantity of nitrogen at time 1, the difference is the quantity of the transformed nitrogen. Nitrogen quantity at time 1 is the May values. Nitrogen quantity at time 2 is the values at each successive month. Transformed nitrogen is the quantity of uptake by plants and soil microbes and converted into organic nitrogen plus the quantity of nitrogen loss by leaching or volatilization. Loss by leaching on Northern Plains prairies is negligible (Power 1970, Brady 1974, Wight and Black 1979, Coyne et al. 1995). Loss by volatilization during 2013 and 2014 with high soil water content would also be negligible. Transformation of ammonium (NH_4) could include some conversion to nitrate (NO_3). Most of the transformed nitrogen would be the quantity converted into organic nitrogen by plants and soil microbes.

A standard t-test was used to analyze differences among means (Mosteller and Rourke 1973).

Results

Precipitation during the 2013 and 2014 growing seasons at the Dickinson Research Extension Center (DREC) ranch was generally favorable with the mean two year growing season precipitation at 20.46 inches (140.87% of LTM). None of the growing season months of 2013 experienced water deficiency conditions and during 2014, July, September, and October had water deficiency conditions, however, the total growing season of 2014 received 19.35 inches of precipitation (133.26% of LTM). Mean precipitation of 2013 and 2014 growing season months indicated that means of July had water deficiency conditions and means of May, August, and October had wet conditions with mean monthly precipitation at 4.48 inches (216.58% of LTM) (table 2) (Manske 2015a).

Available soil water at increments down to 48 inches during the growing season months, April to October, in 2013 and 2014 was high on the nongrazed, seasonlong, and twice-over management treatments. The twice-over treatment had significantly greater soil water than that on the nongrazed treatment. The quantity of soil water on the seasonlong treatment was not significantly different than that on the nongrazed and twice-over treatments. A strong trend with greater soil water can be seen during June to October on the twice-over treatment and the available soil water on the seasonlong treatment showed a trend of lower quantities each growing season month, May to September. During the grazing season, June to October, the soil column, 0 to 48 inches, on twice-over treatment had the greatest quantity of soil water; 23.1% greater than that on the seasonlong and 54.0% greater than that on the nongrazed treatments. The seasonlong treatment had 25.1% greater soil water than that on the nongrazed treatment. The twice-over treatment had the greatest quantity of soil water in the top 12 inches; 19.6% greater than that on the seasonlong and 51.5% greater than that on the nongrazed (table 3). The rhizosphere microorganisms on the nongrazed, seasonlong, and twice-over treatments should have had adequate soil water to flourish at all of the incremental depths down to 48 inches, during each growing season month, April to October, and during both years, 2013-2014 (table 3).

Precipitation during August of 2013 and 2014 was wet with a monthly mean of 5.83 inches (298.72% of LTM) (table 2). During August, soil water at the 0 to 12 inch soil depth on the twice-over treatment was 25.6% greater than that on the seasonlong and 67.8% greater than that on the nongrazed treatments. Soil water at the 12-48 inch soil depth on the twice-over treatment was 50.7% greater than that on the nongrazed treatment. The twice-over treatment had 43.9% greater soil water than that on the seasonlong treatment at the 12 to 48 inch soil depth during August. These data indicate that a high percentage of the high August rainfall infiltrated into the soil on the twice-over treatment. The infiltrated soil water on the twice-over treatment moved downward slowly remaining in the 0-48 inch soil column. A much smaller percentage of the high August rainfall infiltrated into the soil on the nongrazed and seasonlong treatments and did not move downward into the 12 to 48 inch soil depths (table 3).

The quantity of soil water during the growing season was greatest on the twice-over treatment because of the greater amount of aggregation of the soil particles that result from the greater biomass of both endomycorrhizal fungi and ectomycorrhizal fungi (Manske 2011a). Endomycorrhizal fungi secrete great quantities of adhesive polysaccharides that bond soil particles around active grass roots forming the structural environment for symbiotic rhizosphere microorganisms, and this adhesive polysaccharid also binds soil particles into aggregates resulting in increased soil pore spaces, increased water holding capacity, and increased rooting depth.

Ectomycorrhizal fungi secrete large amounts of adhesive polysaccharids forming water-stable aggregates in soil that are water permeable but not water soluable causing increased soil aggregation at increased soil profile depths resulting in improved soil structure and quality, increased soil oxygenation, increased water infiltration and water holding capacity, and decreased erodibility (Caesar-TonThat and Cochran 2000, Caesar-TonThat et al. 2001a, Caesar-TonThat et al. 2001b, Caesar-TonThat 2002, Manske and Caesar-TonThat 2003, Manske 2007).

The aboveground native grass herbage biomass was greatest on the nongrazed treatment. During the grazing season, herbage biomass on the nongrazed treatment was 66.0% greater than the grazed herbage biomass remaining on the seasonlong and 16.7% greater than the grazed herbage biomass remaining on twice-over treatments (tables 4 and 5). The herbage biomass remaining after grazing was 42.3% greater on the twice-over treatment than that on the seasonlong treatment. The stocking rate on the twice-over treatment was 28.6% greater than that on the seasonlong treatment and the quantity of herbage removed by grazing cows would have been 28.6% greater per acre on the twice-over treatment than that on the seasonlong treatment.

During 2013 and 2014, the production of native grass herbage biomass was greater on the twice-over treatment than that on the seasonlong treatment. Native grass lead tiller herbage growth on the twice-over treatment was 23.9% greater, the quantity removed by grazing was 28.6% greater, the herbage residuum after grazing was 22.5% greater, the regrowth of secondary tiller herbage was 160.0% greater, and the herbage residuum and regrowth after grazing was 42.3% greater than those on the seasonlong treatment. The quantity of herbage removed by grazing and the quantity of herbage regrowth after grazing was 70.9% greater on the twice-over treatment than that on the seasonlong treatment.

Some domesticated grass herbage biomass grew on all three treatments, however, collection of that quantity by the standard herbage clipping method was too small to measure (table 5). The basal cover of domesticated grasses was also very low with only a small quantity of plants located by the ten-pin point frame method (table 5).

Basal cover of native grasses was significantly greater on the twice-over treatment than that on the seasonlong and nongrazed treatments (table 5). Native grass basal cover on the twice-over treatment was 193.6% greater than that on the nongrazed and 27.3% greater than that on the seasonlong treatments. Basal cover of native grasses was significantly greater on the seasonlong treatment than that on the nongrazed treatment (table 5). Native grass basal cover on the seasonlong treatment was 130.6% greater than that on the nongrazed treatment.

The quantity of native grass herbage biomass was greater on the twice-over treatment because of the greater rhizosphere microorganism biomass and the greater activation of the defoliation resistance mechanisms that increased the quantity of major and minor essential elements being cycled by the biogeochemical processes from organic forms into available mineral forms. Greater quantities of available mineral nitrogen and fixed carbon promoted greater growth and development of plant material through the compensatory physiological processes. Greater grass herbage biomass was produced that permitted a greater stocking rate with greater quantities of herbage removed by grazing livestock that was replaced by greater quantities of regrowth leaving greater quantities of vegetation after each of the two grazing periods (Manske 2011a, 2014a).

The quantity of native grass basal cover was greater on the twice-over treatment because the greater quantities of available mineral nitrogen and fixed carbon promoted greater numbers of secondary tillers from axillary buds to grow and develop through the processes of vegetative reproduction (Manske 2011a, 2014a).

The rhizosphere microorganism biomass cannot easily be measured directly. With painstaking care, volume of the rhizosphere cylinders around active perennial grass roots can be quantitatively measured. During the growing season of 2002, accurate replicated rhizosphere volume measurements were collected (Gorder, Manske, Stroh 2004). The rhizosphere volume per cubic meter of soil was not different during June on the nongrazed, seasonlong, and twice-over treatments, and the rhizosphere volume on the nongrazed and twice-over treatments were not different during July. The rhizosphere volume on the nongrazed and seasonlong treatments were not different during August and September (table 6 and figure 1). The first grazing period on the twice-over treatment pasture 3 was 15 days during early July to mid July. This grazing period stimulated the rhizosphere microorganisms to increase in quantity and biomass causing the volume to increase from 3900 cm^3/m^3 during July to 7200 cm^3/m^3 during August for an 85.6% increase in volume. The rhizosphere volume on the twice-over treatment pasture 3 decreased slightly during low precipitation occurring in September. The rhizosphere volume during August and September on the twice-over treatment was significantly greater than those on the nongrazed and seasonlong treatments (table 6 and figure 1). The rhizosphere volume on the twice-over treatment was 265.9% greater than that on the seasonlong and was 200.3% greater than that on the nongrazed treatments during August.

The western wheatgrass tiller density associated with the rhizosphere sample cores during the grazing season of 2002 was 1754.25 tillers/m^2 on the twice-over treatment and was significantly greater than the tiller density of 657.84 tillers/m^2 on the seasonlong and was significantly greater than the tiller density of 794.89 tillers/m^2 on the nongrazed treatments. The tiller density on the twice-over treatment was 166.7% greater than that on the seasonlong and was 120.7% greater than that on the nongrazed trreatments (Gorder, Manske, Stroh 2004).

The great increase in rhizosphere volume on the twice-over treatment after July was caused by partial defoliation by grazing cows during 15 days from early July to mid July that removed 25% to 33% of the leaf biomass of lead tillers at vegetative phenological growth stages between the three and a half new leaf stage and the flower stage which activated exudation of short chain carbon energy from the grass tillers through the roots and released into the rhizosphere where microorganisms could rapidly increase in number and biomass by ingesting the needed energy that had been limiting growth and development.

The greater rhizosphere volume on the twice-over treatment had a greater rhizosphere microorganism biomass that mineralized a greater quantity of organic nitrogen into mineral nitrogen. The greater quantity of available mineral nitrogen was the essential nutrient needed to support a greater grass tiller density, a greater production of grass herbage biomass, and a greater production grass regrowth following grazing (Manske 2011a, 2014a).

Percent soil organic matter (SOM) was determined at the NDSU soil testing laboratory from soil cores collected on silty ecological sites. Determination of the weight of the soil organic matter required the determination of the weight of silty soil at each incremental depth from average soil bulk density data (Anonymous circa early 1980's) of all incremental depths (table 7).

Percent (%) and weight (lbs/ac, tons/ac) of soil organic matter was greatest on the twice-over treatment. Soil organic matter (SOM) on the twice-over (3.20%, 231.5 tons/ac) treatment had 11.7% greater percent and 13.5% greater weight than that on the seasonlong (2.82%, 203.9 tons/ac) and had 54.1% greater percent and 116.9% greater weight than that on the nongrazed (1.47%, 106.7 tons/ac) treatments. Soil organic matter on the seasonlong treatment had 47.9% greater percent and 91.1% greater weight than that on the nongrazed treatment (table 8).

Soil organic carbon (SOC) composes 58% of the soil organic matter (Anonymous nd, NRCS Staff 2009, Pluske et al. 2015). The weight of soil organic carbon is 58% of the weight of soil organic matter. The weight of soil organic carbon (SOC) was greater on the twice-over (1.85%, 134.1 tons/ac) treatment than that on the seasonlong (1.63%, 118.2 tons/ac) and nongrazed (0.86%, 62.0 tons/ac) treatments. The weight of soil organic carbon was greater on the seasonlong treatment than that on the nongrazed treatment (table 9).

Soil organic nitrogen (SON) has been estimated to compose 5.8% of the soil organic matter. The weight of soil organic nitrogen (SON) was greater on the twice-over (0.185%, 13.4 tons/ac) treatment than that on the seasonlong (0.163%, 11.8 tons/ac) and nongrazed (0.086%, 6.2 tons/ac) treatments. The weight of soil organic nitrogen was greater on the seasonlong treatment than that on the nongrazed treatment (table 10).

The quantity of mineral nitrogen available in a soil is the net difference between the total quantity of organic nitrogen mineralized by soil microorganisms and the quantity of mineral nitrogen immobilized by plants and soil microbes (Brady 1974, Legg 1975). The quantity of available mineral nitrogen varies cyclically with changes in soil temperature, soil microorganism biomass, and plant phenological growth and development during the growing season (Whitman 1975). The relationships between soil microorganism activity and phenology of plant growth activity results in a dynamic cycle of available mineral nitrogen (Goetz 1975). When mineralization activity by soil microbes is greater than plant growth activity, the quantity of available mineral nitrogen increases. When transformation (immobilization) of mineral nitrogen by plant and soil microbe growth activity is greater than mineralization activity, the quantity of available mineral nitrogen decreases.

The available mineral nitrogen cycle model for a typical growing season would have three peaks and three valleys (Whitman 1975). The first peak of mineral nitrogen would occur in mid May. As plant growth rates increase in June, transformation would increase with available mineral nitrogen at a low value during late June to early July. Mineral nitrogen would increase and reach a second peak during late July or early August. Fall tillers and fall tiller buds start development in mid August and would cause a decrease in mineral nitrogen until mid October. A third peak would occur shortly after mid October. And when liquid water becomes unavailable with winter soil freeze up, available mineral nitrogen would decline for a third low period.

Nitrate (NO_3) cycle on the nongrazed treatment had a peak during May at the 0-6, and 6-12, inch soil depths (table 11). Generally, available mineral nitrate and transformation were low during the growing season at all soil depths. There was a decrease in available mineral nitrate and an increase in transformation during July and again during October at all soil depths. There was an increase in available mineral nitrate and a decrease in transformation during September at the 0-6 inch soil depth (table 11).

Ammonium (NH_4) cycle on the nongrazed treatment had a peak during May at the 0-6 inch soil depth (table 11). Generally, available mineral ammonium was moderate during the growing season at all soil depths and transformation was moderate at the 0-6 inch soil depth and was extremely low or accumulating at the 6-12 and 12-24 inch soil depths. There was a decrease in available mineral ammonium and an increase in transformation during August at the 0-6 inch soil depth. There was an increase in available mineral ammonium and a decrease in transformation during October at the 0-6 inch soil depth (table 11).

Nitrate (NO_3) cycle on the seasonlong treatment had a peak during May at the 0-6 inch soil depth and had minor peaks at the 6-12 and 12-24 inch soil depths (table 12). Generally, available mineral nitrate was relatively high and transformation was moderate during the growing season at all soil depths. There was a decrease in available mineral nitrate and an increase in transformation during July and again during October at the 0-24 inch soil depth. There was an increase in available mineral nitrate and a decrease in transformation during August and September at the 0-6 and 12-24 inch soil depths (table 12).

Ammonium (NH_4) cycle on the seasonlong treatment had a peak during May at the 0-6 inch soil depth. Generally, available mineral ammonium was low during the growing season at all soil depths and transformation was low at the 0-6 inch soil depth, very low or accumulating at the 6-12 inch soil depth, and very low at the 12-24 inch soil depth. Available mineral ammonium and transformation changed little during July, August, and September at all soil depths (table 12).

Nitrate (NO_3) cycle on the twice-over treatment had a peak during May at the 0-6, 6-12, and 12-24 inch soil depths (table 13). Generally, available mineral nitrate was low and transformation was high during July, August, September, and October at all soil depths. There was a decrease in available mineral nitrate and an increase in transformation during July and again during October at all soil depths. There was an increase in available mineral nitrate and a decrease in transformation during September at the 0-6 and 6-12 inch soil depths (table 13).

Ammonium (NH_4) cycle on the twice-over treatment had a peak during May at the 0-6 inch soil depth. Generally, available mineral ammonium was high and transformation was high during the growing season at all soil depths. There was an increase in available mineral ammonium and a decrease in transformation during August and again during October at all soil depths. There was a decrease in available mineral ammonium and an increase in transformation during July and again during September at the 0-6 and 12-24 inch soil depths (table 13).

The reference high peak available mineral nitrogen ($NO_3 + NH_4$) occurs during mid May. The greatest available mineral nitrogen was on the twice-over treatment and the lowest was on the nongrazed treatment. Available mineral nitrogen on the twice-over treatment was 19.4% greater than that on the seasonlong and was 75.6% greater than that on the nongrazed treatments. Available mineral nitrogen on the seasonlong treatment was 47.1% greater than that on the nongrazed treatment (table 14). The twice-over treatment was the only treatment with mineral nitrogen available at quantities greater than 100 lbs/ac. During May, both nitrate (NO_3) and ammonium (NH_4) were available at greater quantities at each soil depth on the twice-over treatment than those on the seasonlong and nongrazed treatments.

During May (table 14), available mineral nitrate (NO_3) was greatest on the twice-over treatment. Available mineral nitrate on the twice-over treatment was 16.9% greater than that on the seasonlong and was 166.9% greater than that on the nongrazed treatments. Available mineral nitrate on the seasonlong treatment was 128.4% greater than that on the nongrazed treatment.

During May (table 14), available mineral ammonium (NH_4) was greatest on the twice-over treatment. Available mineral ammonium on the twice-over treatment was 21.3% greater than that on the seasonlong and was 40.5% greater than that on the nongrazed treatments. Available mineral ammonium on the seasonlong treatment was 15.8% greater than that on the nongrazed treatment.

During the growing season, available mineral nitrogen ($NO_3 + NH_4$) was greatest on the twice-over treatment. Available mineral nitrogen on the twice-over treatment was 9.8% greater than that on the seasonlong and was 41.7% greater than that on the nongrazed treatments. Available mineral nitrogen on the seasonlong treatment was 29.0% greater than that on the nongrazed treatment (table 15).

During the growing season (table 15), available mineral nitrate (NO_3) on the twice-over treatment was 85.6% greater than that on the nongrazed treatment. Available mineral nitrate on the seasonlong treatment was 1.9% greater than that on the twice-over treatment and was 89.2% greater than that on the nongrazed treatment.

During the growing season (table 15), available mineral ammonium (NH_4) on the twice-over treatment was 14.9% greater than that on the seasonlong and was 30.2% greater than that on the nongrazed treatments. Available mineral ammonium on the seasonlong treatment was 13.3% greater than that on the nongrazed treatment. Mineral ammonium tended to accumulate at the 6 to 24 inch soil depths on the nongrazed treatment (table 15).

Transformation of mineral nitrogen was greatest on twice-over treatment. Transformation of mineral nitrogen on the twice-over treatment was 48.0% greater than that on the seasonlong and was 276.8% greater than that on the nongrazed treatments. Transformation of mineral nitrogen on the seasonlong treatment was 154.5% greater than that on the nongrazed treatment (table 15). Mineral nitrogen from the lower soil depths was tranformed in greater quantities on the twice-over treatment. Transformed mineral nitrogen on the twice-over treatment from the 6 to 24 inch soil depths was 76.0% greater than that used on the seasonlong and was 194.2% greater than that used on the nongrazed treatments (table 15).

Transformation of mineral nitrate (NO_3) on the twice-over treatment was 37.7% greater than that on the seasonlong and was 307.0% greater than that on the nongrazed treatments. Transformation of mineral nitrate on the seasonlong treatment was 195.7% greater than that on the nongrazed treatment (table 15).

Transformation of mineral ammonium (NH_4) on the twice-over treatment was 95.5% greater than that on the seasonlong and was 203.0% greater than that on the nongrazed treatments. Transformation of mineral ammonium on the seasonlong treatment was 55.0% greater than that on the nongrazed treatment (table 15).

The quantity of available mineral nitrogen was greater than 100 lbs/ac on the twice-over treatment and the quantity of available mineral nitrate and mineral ammonium during May and during the growing season was greatest on the twice-over treatment because the rhizosphere microorganism biomass was greater. The rhizosphere microorganism biomass was greater on the twice-over treatment as a result of two grazing periods that coordinated grazing activity with grass phenological growth that resulted in greater quantities of short chain carbon energy to be exudated into the rhizosphere for the microorganisms from the roots of partially defoliated vegetative lead tillers during the first grazing period between early June and mid July. The increased energy increased the microbe biomass; the greater microbe biomass mineralized greater quantities of organic nitrogen into mineral nitrogen. Microbial digestion produces ammonia and ammonium. The ammonia can readily be hydrolyzed into stable ammonium. Some of the ammonium can be oxidized during nitrification to produce nitrate. These processes result in greater quantities of available mineral nitrate and mineral ammonium (Manske 1999, 2009a, 2009b, 2011a, 2014a, 2015b).

The quantity of mineral nitrate and mineral ammonium transformed (immobilized) was greatest on the twice-over treatment because the high activity of the defoliation resistance mechanisms was producing greater quantities of herbage biomass, vegetative tillers, and regrowth leaves and shoots and the greater quantity of soil microorganisms were maintaining and increasing their biomass requiring the use of greater quantities of mineral nitrogen (Manske 2011a, 2014a).

The cow and calf weight performance on the twice-over treatment has been greater for 21 years than that on the seasonlong treatment (table 16). Calves on the twice-over treatment have accumulated 13.9% greater weight, gained 14.1% greater weight per day, and gained 33.8% greater weight per acre than the calves on the seasonlong treatment. Cows on the twice-over treatment have accumulated 81.7% greater weight, gained 131.0% greater weight per day, and gained 183.7% greater weight per acre than the cows on the seasonlong treatment.

The quantity of cow and calf weight performance was greatest on the twice-over treatment because of the increased production and quality of the herbage biomass, vegetative tillers, and regrowth biomass. The increased herbage biomass permitted the greater stocking rate without harming the vegetation. The greater vegetative tillers and regrowth biomass provide greater forage of higher quality that permitted the cows to produce milk closer to their genetic potential that permitted the calves to grow closer to their genetic potential during the period from mid July to mid October when traditional practices provide forage with quality below the livestock requirements (Manske 2008a, 2008b, 2008c, 2014b).

Discussion

The “good as new” condition for native grassland ecosystems in the Northern Plains produces herbage biomass, vegetative tillers, and regrowth of stems and leaves at potential levels and cow and calf weight performance at near genetic potentials. These potential levels of production are reached when soil mineral nitrogen is available at threshold quantities of 100 lbs/ac or greater.

The weight of soil organic nitrogen was 13.4 tons/ac on the twice-over treatment, 11.8 tons/ac on the seasonlong treatment, and 6.2 tons/ac on the nongrazed treatment. Soil organic nitrogen in the silty ecological sites was adequately abundant on all three treatments. Plants cannot use organic nitrogen. The reference high peak level of mineral nitrogen during mid May (at time 1) was available at 102.6 lbs/ac on the twice-over treatment, 85.9 lbs/ac on the seasonlong treatment, and 58.4 lbs/ac on the nongrazed treatment. The twice-over treatment was the only management strategy with mineral nitrogen available at or above the threshold quantity of 100 lbs/ac. Both the nongrazed and seasonlong treatments had deficiencies in the quantity of available mineral nitrogen.

Conversion of soil organic nitrogen into 100 lbs/ac of available mineral nitrogen is a complex process, requires a great biomass of rhizosphere microorganisms, and has high biological costs for the ecosystem. The primary producer trophic level in the rhizosphere are achlorophyllous saprophytes and cannot fix carbon for energy. The rhizosphere microorganism biomass and activity are limited by access to short chain carbon energy (Manske 2011a, 2014a). Greater quantities of short chain carbon compounds are produced during photosynthesis than healthy grass tillers need for growth and development (Coyne et al. 1995). Some of this surplus short chain carbon energy can be moved from the grass tiller through the roots into the rhizosphere with partial defoliation by large grazing graminivores.

The twice-over rotation strategy has two grazing periods on each pasture of a three to six pasture system. The periods of partial defoliation by grazing are coordinated with the grass tillers phenological growth stages. Grazing on each of the pastures during the first period (1 June to 15 July) removes 25% to 33% of the leaf weight of grass lead tillers at vegetative growth stages between the three and a half new leaf stage and the flower stage that fully activates the defoliation resistance mechanisms (Manske 1999, 2011a, 2014a).

Full activation of the compensatory physiological processes within grass plants accelerates growth rates of replacement leaves and shoots, increases photosynthetic capacity of remaining mature leaves that increases the quantity of available fixed carbon, improves water (precipitation) use efficiency, and increases restoration of biological and physiological processes enabling rapid and complete recovery of partially defoliated grass tillers.

Full activation of the asexual processes of vegetative production increases secondary tiller development from axillary buds, increases initiated tiller density during the grazing season, and increases herbage biomass production and improves herbage nutritional quality.

Full activation of the external symbiotic rhizosphere organism activity occurs with the exudation of adequate quantities of surplus short chain carbon energy that increases microorganism numbers and biomass that increases rhizosphere volume, increases ecosystem biogeochemical cycling of essential elements, increases soil aggregation that improves water infiltration and increases soil water holding capacity, increases soil organic matter, soil organic carbon, and soil organic nitrogen, increases mineralization of soil organic nitrogen that increases the quantity of available mineral nitrogen, and improves belowground resource uptake competitiveness.

During the second grazing period (mid July to mid October) each pasture is grazed for double the number of days it was grazed the during first period. Because the greater herbage biomass resulting from the increased growth of replacement herbage and the increased tiller density (basal cover) and greater nutritional quality from the great number of secondary vegetative tillers that have developed past the three and a half new leaf stage, the cow and calf weight performance is improved. The stocking rates are increased without harming the vegetation. The lactating cows gain weight and produce milk close to their genetic potential and their calves gain weight at or near their genetic potential during the entire grazing season.

Grass plants, soil organisms, and graminivores have developed numerous complex symbiotic mechanisms and processes. The twice-over rotation grazing strategy is designed to fully activate and maintain these defoliation resistance mechanisms so that the ecosystem biogeochemical processes function at potential levels, that the biological requirements for all of the biotic organisms are met, and that the native grassland ecosystems are fully renewed back to the “good as new” condition.

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Table 1. Weight of soil, soil organic matter (SOM), soil organic carbon (SOC), and soil organic nitrogen (SON) in pounds per acre per incremental depth and the quantity of transformed nitrogen.

Mathematical Formula
Soil weight per increment of soil depth per acre
Soil bulk density in g/cm^3 X depth of soil in cm X 100,000,000 cm^2/1 hectare
X 1 ha/2.471 ac X 1 lb/453.5924 g = soil weight in lbs/ac
Weight of soil organic matter (SOM)
Weight of soil in lbs/ac X % SOM/100 = weight of SOM in lbs/ac
Weight of soil organic carbon (SOC)
Weight of soil in lbs/ac X (% SOM/100 X 0.58) = weight of SOC in lbs/ac
Weight of soil organic nitrogen (SON)
Weight of soil in lbs/ac X (% SOM/100 X 0.058) = weight of SON in lbs/ac
Net mineralization measurements of the nitrogen balance equation
$M = \Delta NH_4 + \Delta NO_3 + \Delta$ Plant + loss
$\Delta N = NT2 - NT1$ = Transformed nitrogen

Table 2. Precipitation in inches during the growing season at the DREC ranch, 2013 and 2014.

	Apr	May	Jun	Jul	Aug	Sep	Oct	Growing Season
2013								
inches/month	1.05	7.55	2.23	2.13	2.81	2.44	3.35	21.56
% of LTM	73.43	274.55	68.83	90.64	144.10	169.44	248.15	148.48
wet/dry		wet			wet	wet	wet	wet
2014								
inches/month	1.41	3.73	3.38	0.37	8.84	1.03	0.59	19.35
% of LTM	98.60	135.64	104.32	15.74	453.33	71.53	43.70	133.26
wet/dry		wet		dry	wet	dry	dry	wet
2013-2014								
inches/month	1.23	5.64	2.81	1.25	5.83	1.74	1.97	20.46
% of LTM	86.01	205.09	86.57	53.19	298.72	120.49	145.93	140.87
wet/dry		wet		dry	wet		wet	wet

Table 3. Mean inches of soil water at incremental depths during the growing season on the silty ecological sites of the three grazing management treatments, 2013-2014.

Soil Depth (inches)	Apr	May	Jun	Jul	Aug	Sep	Oct
Nongrazed							
0-6	1.27	1.21	0.66	0.60	0.84	0.92	1.02
6-12	0.91	0.91	0.64	0.43	0.62	0.73	0.81
12-24	1.59	1.42	1.37	0.94	1.15	1.32	1.45
24-36	2.02	1.81	1.73	1.48	1.47	1.43	1.75
36-48	1.74	1.68	1.64	1.40	1.40	1.39	1.49
0-48	7.53	7.03	6.04	4.85	5.48	5.79	6.52
Seasonlong							
0-6	1.43	1.55	1.12	0.83	1.13	0.94	1.34
6-12	1.08	1.05	0.93	0.71	0.82	0.71	1.01
12-24	2.25	2.16	2.16	1.66	1.54	1.39	1.92
24-36	1.53	2.02	2.23	2.13	1.42	1.45	1.91
36-48	1.85	1.86	2.03	2.06	1.25	1.63	1.66
0-48	8.14	8.63	8.45	7.37	6.14	6.10	7.83
Twice-over							
0-6	1.61	1.34	1.19	0.99	1.38	1.17	1.56
6-12	1.37	1.28	1.17	0.86	1.07	1.15	1.34
12-24	2.77	2.53	2.51	1.82	2.14	2.09	2.46
24-36	1.52	2.28	2.52	1.99	2.00	1.99	2.12
36-48	1.39	2.00	2.58	2.18	1.92	2.01	2.03
0-48	8.66	9.43	9.95	7.83	8.52	8.39	9.49

Table 4. Native grass herbage biomass in lbs/ac during the growing season on the silty ecological sites of three grazing management treatments, 2013-2014.

Grazing Management Year	May	Jun	Jul	Aug	Sep	Oct
Nongrazed						
2013	1036.50		2211.57			3226.66
2014	802.21		2870.46			3264.72
Mean	919.36		2541.02			3245.69
Seasonlong						
2013	522.36		1401.63			1212.53
2014	458.84	910.74	1976.08	1019.26	1658.53	1697.18
Mean	490.60	910.74	1688.86	1019.26	1658.53	1454.86
Twice-over						
2013	469.01	1177.20	2030.79	1935.44	2030.19	1754.27
2014	784.48	1310.05	2453.99	1658.53	2642.10	2167.84
Mean	626.75	1243.63	2242.39	1796.99	2336.15	1961.06

Table 5. Mean annual domesticated and native grass herbage biomass (lbs/ac) and basal cover (%) on the silty ecological sites of the three grazing management treatments, 2013-2014.

	Domesticated Grass		Native Grass	
Management Treatment	Herbage lbs/ac	Basal Cover %	Herbage lbs/ac	Basal Cover %
Nongrazed	-	0.17	2235.36	9.64z
Seasonlong	-	0.00	1346.45	22.23y
Twice-over	-	0.37	1916.04	28.30x

Means in the same column and followed by the same letter (x, y, z) are not significantly different (P<0.05).

Table 6. Rhizosphere volume in cubic centimeters per cubic meter of soil (cm^3/m^3), 2002.

Grazing Management	May	Jun	Jul	Aug	Sep	Oct
Nongrazed		1725.24a	2804.61a	2391.97b	2438.47b	
Seasonlong		1800.93a	642.21b	1963.02b	1802.97b	
Twice-over		3214.75a	3867.54a	7183.27a	6586.06a	

Means in the same column and followed by the same letter are not significantly different (P<0.05).
Data from Gorder, Manske, and Stroh, 2004.

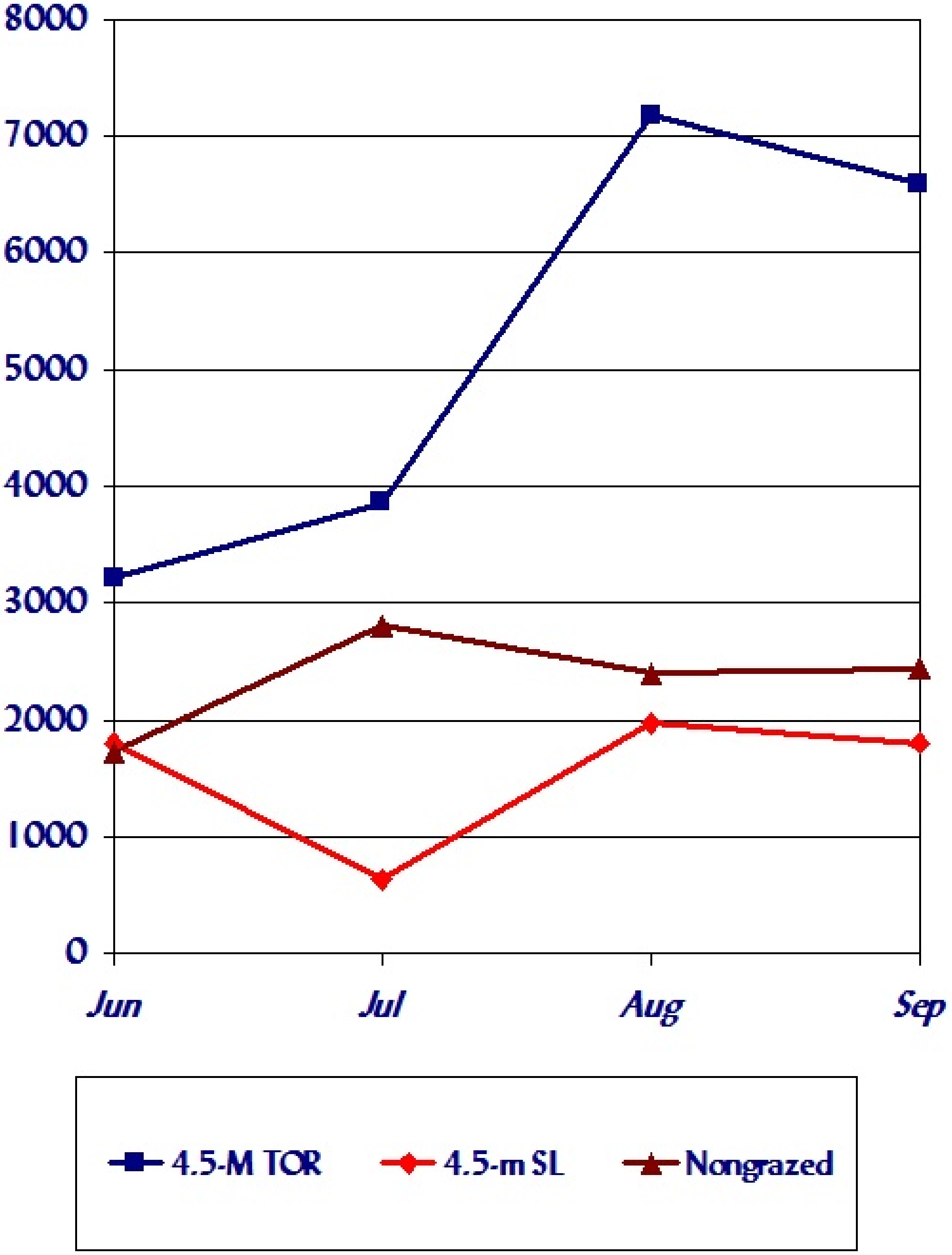

Figure 1. Rhizosphere volume (cm^3) per cubic meter of soil

Table 7. Generalized soil bulk density and soil weight at incremental depths on silty ecological sites of rangeland in southwestern North Dakota.

Soil Depth	Soil Bulk Density	Soil Weight	
(inches)	(g/cm^3)	(lbs/ac)	(tons/ac)
0-6	1.15	1,560,194.37	780.10
6-12	1.30	1,763,697.98	881.85
12-24	1.30	3,536,678.58	1,768.34
24-36	1.33	3,618,294.24	1,809.15
36-48	1.47	3,999,167.32	1,999.58
0-48		14,478,032.49	7,239.02

Average silty soil bulk density from Anonymous. circa early 1980's.
NDSU Soils Department.

Table 8. Soil organic matter (SOM) at incremental depths as percent, pounds per acre, and tons per acre during June on silty ecological sites of the grazing management treatments, 2013-2014.

	Soil Depth (inches)					
	0-6	6-12	12-24	24-36	36-48	0-48
Nongrazed						
SOM						
%	3.08	1.89	1.45	1.15	0.98	1.47
lbs/ac	48,053.99	33,333.89	51,281.84	41,610.38	39,191.84	213,471.94
tons/ac	24.03	16.67	25.64	20.81	19.60	106.74
Seasonlong						
SOM						
%	6.07	3.38	2.55	2.26	2.04	2.82
lbs/ac	94,703.80	59,612.99	90,185.30	81,773.45	81,583.01	407,858.55
tons/ac	47.35	29.81	45.09	40.89	40.79	203.93
Twice-over						
SOM						
%	5.98	4.19	3.38	2.56	2.09	3.20
lbs/ac	93,299.62	73,898.95	119,539.74	92,628.33	83,582.60	462,949.24
tons/ac	46.65	36.95	59.77	46.31	41.79	231.47

Table 9. Soil organic carbon (SOC) at incremental depths as percent, pounds per acre, and tons per acre during June on silty ecological sites of the grazing management treatments, 2013-2014.

	Soil Depth (inches)					
	0-6	6-12	12-24	24-36	36-48	0-48
Nongrazed						
SOC						
%	1.79	1.10	0.84	0.67	0.57	0.86
lbs/ac	27,927.48	19,400.68	29,708.10	24,242.57	22,795.25	124,074.08
tons/ac	13.96	9.70	14.85	12.12	11.40	62.04
Seasonlong						
SOC						
%	3.52	1.96	1.48	1.31	1.18	1.63
lbs/ac	54,918.84	34,568.48	52,342.84	47,399.65	47,190.17	236,419.98
tons/ac	27.46	17.28	26.17	23.70	23.60	118.21
Twice-over						
SOC						
%	3.47	2.43	1.96	1.48	1.21	1.85
lbs/ac	54,138.74	42,857.86	69,318.90	53,550.75	48,389.92	268,256.17
tons/ac	27.07	21.43	34.66	26.78	24.19	134.13

Table 10. Soil organic nitrogen (SON) at incremental depths as percent, pounds per acre, and tons per acre during June on silty ecological sites of the grazing management treatments, 2013-2014.

	Soil Depth (inches)					
	0-6	6-12	12-24	24-36	36-48	0-48
Nongrazed						
SON						
%	0.179	0.110	0.084	0.067	0.057	0.086
lbs/ac	2,792.74	1,940.07	2,970.81	2,424.26	2,279.53	12,407.41
tons/ac	1.40	0.97	1.49	1.21	1.14	6.20
Seasonlong						
SON						
%	0.352	0.196	0.148	0.131	0.118	0.163
lbs/ac	5,491.88	3,456.85	5,234.28	4,739.97	4,719.02	23,642.00
tons/ac	2.75	1.73	2.62	2.37	2.36	11.82
Twice-over						
SON						
%	0.347	0.243	0.196	0.148	0.121	0.185
lbs/ac	5,413.87	4,285.79	6,931.89	5,355.08	4,838.99	26,825.62
tons/ac	2.71	2.14	3.47	2.68	2.42	13.41

Table 11. Mean mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac with quantities available in the soil and estimated quantities transformed to organic nitrogen during the growing season on silty ecological sites of the long-term nongrazed prairie, 2013-2014.

Soil Depth (inches)	May	Jun	Jul	Aug	Sep	Oct
NO_3 nitrate						
0-6 Available	9.75	5.00	2.25	3.13	5.50	3.13
0-6 Transformed		-4.75	-7.50	-6.62	-4.25	-6.62
6-12 Available	3.00	2.38	2.00	2.25	2.25	2.25
6-12 Transformed		-0.62	-1.00	-0.75	-0.75	-0.75
12-24 Available	3.50	2.88	2.00	2.50	4.50	3.50
12-24 Transformed		-0.62	-1.50	-1.00	+1.00	0.00
0-24 Available	16.25	10.26	6.25	7.88	12.25	8.88
0-24 Transformed		-5.99	-10.00	-8.37	-4.00	-7.37
NH_4 ammonium						
0-6 Available	16.48	13.42	12.85	11.06	12.97	15.46
0-6 Transformed		-3.06	-3.63	-5.42	-3.51	-1.02
6-12 Available	13.10	12.45	12.36	13.18	13.87	15.05
6-12 Transformed		-0.65	-0.74	+0.08	+0.77	+1.95
12-24 Available	12.57	12.65	14.68	9.87	12.65	13.38
12-24 Transformed		+0.08	+2.11	-2.70	+0.08	+0.81
0-24 Available	42.15	38.52	39.89	34.11	39.49	43.89
0-24 Transformed		-3.63	-2.26	-8.04	-2.66	+1.74
$NO_3 + NH_4$						
0-6 Available	26.23	18.42	15.10	14.19	18.47	18.59
0-6 Transformed		-7.81	-11.13	-12.04	-7.76	-7.64
6-12 Available	16.10	14.83	14.36	15.43	16.12	17.30
6-12 Transformed		-1.27	-1.74	-0.67	+0.02	+1.20
12-24 Available	16.07	15.53	16.68	12.37	17.15	16.88
12-24 Transformed		-0.54	+0.61	-3.70	+1.08	+0.81
0-24 Available	58.40	48.78	46.14	41.99	51.74	52.77
0-24 Transformed		-9.62	-12.26	-16.41	-6.66	-5.63

Table 12. Mean mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac with quantities available in the soil and estimated quantities transformed to organic nitrogen during the growing season on silty ecological sites of the seasonlong grazing system, 2013-2014.

Soil Depth (inches)	May	Jun	Jul	Aug	Sep	Oct
NO_3 nitrate						
0-6 Available	19.38	12.88	3.25	6.32	6.42	4.57
0-6 Transformed		-6.50	-16.13	-13.06	-12.96	-14.81
6-12 Available	7.57	7.38	3.07	2.76	4.63	2.32
6-12 Transformed		-0.19	-4.50	-4.81	-2.94	-5.25
12-24 Available	10.16	9.00	3.25	5.00	4.75	4.25
12-24 Transformed		-1.16	-6.91	-5.16	-5.41	-5.91
0-24 Available	37.11	29.26	9.57	14.08	15.80	11.14
0-24 Transformed		-7.85	-27.54	-23.03	-21.31	-25.97
NH_4 ammonium						
0-6 Available	18.87	17.32	14.34	14.46	14.42	17.18
0-6 Transformed		-1.55	-4.53	-4.41	-4.45	-1.69
6-12 Available	14.59	15.77	13.81	13.24	13.77	16.62
6-12 Transformed		+1.18	-0.78	-1.35	-0.82	+2.03
12-24 Available	15.37	15.24	13.42	13.91	12.72	14.77
12-24 Transformed		-0.13	-1.95	-1.46	-2.65	-0.60
0-24 Available	48.82	48.32	41.57	41.61	40.91	48.57
0-24 Transformed		-0.50	-7.25	-7.21	-7.91	-0.25
NO_3 + NH_4						
0-6 Available	38.24	30.19	17.59	20.78	20.84	21.74
0-6 Transformed		-8.05	-20.65	-17.46	-17.40	-16.50
6-12 Available	22.15	23.14	16.87	15.99	18.39	18.94
6-12 Transformed		+0.99	-5.28	-6.16	-3.76	-3.21
12-24 Available	25.53	24.24	16.67	18.91	17.48	19.02
12-24 Transformed		-1.29	-8.86	-6.62	-8.05	-6.51
0-24 Available	85.92	77.57	51.13	55.68	56.71	59.70
0-24 Transformed		-8.35	-34.79	-30.24	-29.21	-26.22

Table 13. Mean mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac with quantities available in the soil and estimated quantities transformed to organic nitrogen during the growing season on silty ecological sites of the twice-over rotation system, 2013-2014.

Soil Depth (inches)	May	Jun	Jul	Aug	Sep	Oct
NO_3 nitrate						
0-6 Available	20.94	10.94	2.44	4.25	7.13	3.63
0-6 Transformed		-10.00	-18.50	-16.69	-13.82	-17.32
6-12 Available	9.13	6.50	2.07	2.57	3.94	2.25
6-12 Transformed		-2.63	-7.07	-6.57	-5.19	-6.88
12-24 Available	13.30	11.63	2.00	4.75	3.75	3.50
12-24 Transformed		-1.67	-11.30	-8.55	-9.55	-9.80
0-24 Available	43.37	29.07	6.51	11.57	14.82	9.38
0-24 Transformed		-14.30	-36.86	-31.80	-28.55	-33.99
NH_4 ammonium						
0-6 Available	22.25	18.46	14.48	16.22	15.48	18.09
0-6 Transformed		-3.79	-7.77	-6.03	-6.77	-4.16
6-12 Available	18.03	18.32	15.20	15.26	15.81	19.83
6-12 Transformed		+0.29	-2.83	-2.77	-2.22	+1.80
12-24 Available	18.93	20.46	14.20	16.89	15.22	16.97
12-24 Transformed		+1.53	-4.73	-2.04	-3.71	-1.96
0-24 Available	59.21	57.23	43.88	48.37	46.51	54.89
0-24 Transformed		-1.98	-15.33	-10.84	-12.70	-4.32
$NO_3 + NH_4$						
0-6 Available	43.19	29.40	16.92	20.47	22.61	21.72
0-6 Transformed		-13.79	-26.27	-22.72	-20.58	-21.47
6-12 Available	27.15	24.82	17.26	17.82	19.75	22.08
6-12 Transformed		-2.33	-9.89	-9.33	-7.40	-5.07
12-24 Available	32.23	32.08	16.20	21.64	18.97	20.47
12-24 Transformed		-0.15	-16.03	-10.59	-13.26	-11.76
0-24 Available	102.57	86.29	50.38	59.93	61.33	64.27
0-24 Transformed		-16.28	-52.19	-42.64	-41.24	-38.30

Table 14. May available and mean monthly transformed mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac during the growing season on silty ecological sites of the three grazing management treatments, 2013-2014.

Soil Depth (inches)	Nongrazed		Seasonlong		Twice-over	
	May Available	Mean Monthly Transformed	May Available	Mean Monthly Transformed	May Available	Mean Monthly Transformed
NO_3 nitrate						
0-6	9.75	-5.45	19.38	12.69	20.94	15.27
6-12	3.00	-0.77	7.57	3.54	9.13	5.67
12-24	3.50	-0.42	10.16	4.91	13.30	8.17
0-24	16.25	-7.15	37.11	21.14	43.37	29.10
NH_4 ammonium						
0-6	16.48	-3.33	18.87	3.33	22.25	5.70
6-12	13.10	+0.28	14.59	+0.05	18.03	1.15
12-24	12.57	+0.08	15.37	1.36	18.93	2.18
0-24	42.15	-2.98	48.82	4.62	59.21	9.03
$NO_3 + NH_4$						
0-6	26.23	-9.28	38.24	16.01	43.19	20.97
6-12	16.10	-0.49	22.15	3.48	27.15	6.80
12-24	16.07	-0.35	25.53	6.27	32.23	10.36
0-24	58.40	-10.12	85.92	25.76	102.57	38.13

Table 15. Mean monthly (May-October) mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), at incremental depths in lbs/ac with quantities available in the soil and estimated quantities transformed to organic nitrogen during the growing season on silty ecological sites of the three grazing management treatments, 2013-2014.

Soil Depth (inches)	Nongrazed	Seasonlong	Twice-over
NO_3 nitrate			
0-6 Available	4.79	8.80	8.22
0-6 Transformed	-5.95	-12.69	-15.27
6-12 Available	2.36	4.62	4.41
6-12 Transformed	-0.77	-3.54	-5.67
12-24 Available	3.15	6.07	6.49
12-24 Transformed	-0.42	-4.91	-8.17
0-24 Available	10.30	19.49	19.12
0-24 Transformed	-7.15	-21.14	-29.10
NH_4 ammonium			
0-6 Available	13.71	16.10	17.50
0-6 Transformed	-3.33	-3.33	-5.70
6-12 Available	13.34	14.63	17.08
6-12 Transformed	+0.28	+0.05	-1.15
12-24 Available	12.63	14.24	17.11
12-24 Transformed	+0.08	-1.36	-2.18
0-24 Available	39.68	44.97	51.68
0-24 Transformed	-2.98	-4.62	-9.03
$NO_3 + NH_4$			
0-6 Available	18.50	24.90	25.72
0-6 Transformed	-9.28	-16.01	-20.97
6-12 Available	15.69	19.25	21.48
6-12 Transformed	-0.49	-3.48	-6.80
12-24 Available	15.78	20.31	23.60
12-24 Transformed	-0.35	-6.27	-10.36
0-24 Available	49.97	64.45	70.80
0-24 Transformed	-10.12	-25.76	-38.13

Table 16. Mean cow and calf weight performance (in pounds) on native grassland pastures managed by the seasonlong and the twice-over rotation grazing treatments, 1983-2003.

	Seasonlong		Twice-over	
	Cow	Calf	Cow	Calf
Accumulated Wt	47.77	307.54	86.80	350.33
Gain/Day	0.29	2.42	0.67	2.76
Gain/Acre	3.06	26.19	8.68	35.03

Literature Cited

Anonymous. circa early 1980's. Average bulk density from analysis of comparable soils. In memorandum to Branch Experiment Station Personnel from Department of Soils, NDSU, Staff. 3p.

Anonymous. no date. Soil quality for environmental health. http://soilquality.org/indicators/total_organic_carbon.

Barker, W.T., and W.C. Whitman. 1988. Vegetation of the Northern Great Plains. Rangelands 10:266-272.

Bloem, J., D.W. Hopkins, and A. Benedetti (eds.). 2006. Microbiological methods for assessing soil quality. CAB International, Oxfordshire, UK. 307p.

Brady, N.C. 1974. The nature and properties of soils. MacMillan Publishing Co., Inc., New York, NY. 639p.

Caesar-TonThat, T.C., and V. Cochran. 2000. Soil aggregate stabilization by a saprophytic lignin-decomposing basidiomycete fungus. I. Microbiological aspects. Biology and Fertility of Soils 32:374-380.

Caesar-TonThat, T.C., W. Shelver, R.G. Thorn, and V.L. Cochran. 2001a. Generation of antibodies for soil-aggregating basidiomycete detection to determine soil quality. Applied Soil Ecology 18:99-116.

Caesar-TonThat, T.C., D.H. Branson, J.D. Reeder, and L.L. Manske. 2001b. Soil-aggregating basidiomycetes in the rhizosphere of grasses under two grazing management systems. Poster. American Society of Agronomy Annual Meeting. Charlotte, NC.

Caesar-TonThat, T.C. 2002. Soil binding properties of mucilage produced by a basidiomycete fungus in a model system. Mycological Research 106:930-937.

Cook, C.W., and J. Stubbendieck. 1986. Range research: basic problems and techniques. Society for Range Management, Denver, CO. 317p.

Coyne, P.I., M.J. Trlica, and C.E. Owensby. 1995. Carbon and nitrogen dynamics in range plants. p. 59-167. *in* D.J. Bedunah and R.E. Sosebee (eds.). Wildland plants: physiological ecology and developmental morphology. Society for Range Management, Denver, CO.

Goetz, H. 1975. Availability of nitrogen and other nutrients on four fertilized range sites during the active growing season. Journal of Range Management 28:305-310.

Gorder, M.M., L.L. Manske, and T.L. Stroh. 2004. Grazing treatment effects on vegetative tillering and soil rhizospheres of western wheatgrass. NDSU Dickinson Research Extension Center. Range Research Report DREC 04-1056. Dickinson, ND. 13p.

Kochy, M. 1999. Grass-tree interactions in western Canada. Ph.D. Dissertation. University of Regina. Regina, Saskatchewan, Canada.

Legg, J.O. 1975. Influence of plants on nitrogen transformation in soils. pg. 221-227. *in* M.K. Wali (ed.). Prairie: A multiple view. University of North Dakota Press. Grand Forks, ND.

Manske, L.L. 1999. Can native prairie be sustained under livestock grazing? Provincial Museum of Alberta. Natural History Occasional Paper No. 24. Edmonton, Alberta. p.99-108.

Manske, L.L. 2007. Ectomycorrhizal Basidiomycete fungi detected in rhizospheres of mixed grass prairie grasses. NDSU Dickinson Research Extension Center. Summary Range Research Report DREC 07-3047. Dickinson, ND. 3p.

Manske, L.L. 2008a. Cow and calf performance as affected by grazing management. NDSU Dickinson Research Extension Center. Range Research Report DREC 08-1052b. Dickinson, ND. 28p.

Manske, L.L. 2008b. Cow and calf performance on pasture-forage types during Fall, mid October to mid November. NDSU Dickinson Research Extension Center. Range Research Report DREC 08-1054b. Dickinson, ND. 14p.

Manske, L.L. 2008c. Annual nutritional quality curves for graminoids in the Northern Plains. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 08-3014c. Dickinson, ND. 15p.

Manske, L.L. 2009a. Enhancement of the nitrogen cycle improves native rangeland. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 09-3054. Dickinson, ND. 6p.

Manske, L.L. 2009b. Influence of soil mineral nitrogen on native rangeland plant water use efficiency and herbage production. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 09-3053. Dickinson, ND. 3p.

Manske, L.L., S. Schneider, J.A. Urban, and J.J. Kubik. 2010. Plant water stress frequency and periodicity in western North Dakota. NDSU Dickinson Research Extension Center. Range Research Report DREC 10-1077. Dickinson, ND. 11p.

Manske, L.L. 2011a. Biology of defoliation by grazing. NDSU Dickinson Research Extension Center. Range Management Report DREC 11-1067b. Dickinson, ND. 25p.

Manske, L.L. 2011b. Soil mineral nitrogen increased above the threshold quantity of 100 pounds per acre in rangeland ecosystems. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 11-3056. Dickinson, ND. 8p.

Manske, L.L. 2014a. Grazingland management based on native rangeland ecosystem mechanisms and processes. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 14-3062. Dickinson, ND. 18p.

Manske, L.L. 2014b. Diagnostic examination of the value captured from land natural resources by twelve month forage management strategies for range cows. NDSU Dickinson Research Extension Center. Range Research Report DREC 14-1084. Dickinson, ND. 23p.

Manske, L.L. 2015a. Range plant growth related to climatic factors of Western North Dakota, 1982-2014. NDSU Dickinson Research Extension Center. Range Research Report DREC 15-1078d. Dickinson, ND. 19p.

Manske, L.L. 2015b. Biogeochemical Processes of Nitrogen in Rangeland Soils. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 15-3066. Dickinson, ND. 13p.

Mosteller, F., and R.E.K. Rourke. 1973. Sturdy statistics. Addison-Wesley Publishing Co., MA. 395p.

NRCS Staff. 2009. Soil quality indicators: Total organic carbon. USDA Natural Resources Conservation Service. http://www.nrcs.gov/wps/portal/nrcs/detail/soils/health/assessment/

Peltzer, D.A., and M. Kochy. 2001. Competitive effects of grasses and woody plants in mixed grass prairie. Journal of Ecology 89:519-527.

Pluske, W., D. Murphy, and J. Shephard. 2015. Total organic carbon. Australian Governments Natural Heritage Trust. www.soilquality.org.au/

Power, J.F. 1970. Nitrogen management of semiarid grasslands in North America. Proceedings of the XI International Grassland Congress. 1970:468-471.

Shiflet, T.N. (ed.). 1994. Rangeland cover types. Society for Range Management. Denver, CO. 152p.

Tilman, D. 1990. Constraints and tradeoffs: toward a predictive theory of competition and succession. Oikos 58:3-15.

Whitman, W.C. 1975. Native range fertilization and interseeding study. Annual Report. Dickinson Experiment Station. Dickinson, ND. p. 11-16.

Wight, J.R., and A.L. Black. 1979. Range fertilization: plant response and water use. Journal of Range Management 32:345-349.

The Renewability of Grazingland Ecosystems

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Native rangeland and domesticated grassland ecosystems are the major grazingland renewable natural resources in the Northern Plains. The 'natural resources' portion of this term indicates that these are assets produced by processes of nature that can be sources of wealth. The 'renewable' portion of this term is usually not recognized as an extremely important description which declares that grazingland ecosystems have the capability to return to the potential production level of the good-as-new condition following deterioration of ecosystem performance that results from unenlightened management practices. Grazingland ecosystems degrade (decrease) when management causes output of essential elements to be greater than input. Conversely, grazingland ecosystems aggrade (increase) when management causes input of essential elements to be greater than output (McGill and Cole 1981).

The single most important factor that determines when a grazingland ecosystem is degraded or aggraded by a particular type of management practice depends on whether the biomass of soil microorganisms is decreasing or increasing. A large biomass of soil microbes is required to aggrade grazingland ecosystems (Coleman et al. 1983, Schimel, Coleman, and Horton 1985, Cheng and Johnson 1998). The soil microorganisms are the renewable portion of grazingland natural resources.

Grazingland ecosystems aggrade when beneficial management practices cause increased quantities of labile (readily available) simple carbon chain energy from grass tillers at vegetative growth stages to be exudated (released) through the roots into the rhizosphere providing the limiting nutrient necessary for microorganism biomass to increase (Anderson et al. 1981, Curl and Truelove 1986, Whipps 1990, Hamilton and Frank 2001, Manske 2011). The resulting increased biomass of soil microorganisms are able to decompose and mineralize components of the soil organic matter at stimulated rates producing greater quantities of essential elements in plant available mineral (inorganic) forms (Coleman et al. 1983, Klein et al. 1988, Bird et al. 2002, Burrows and Pfleger 2002, Rillig et al. 2002, Driver et al. 2005). The increased quantities of available essential elements provide for increased production of plant herbage biomass. Greater quantities and improved quality of herbage provide for increased quantities of livestock weight production (Manske 2008).

These resulting increased quantities of microbe, plant, and animal biomass supplies greater amounts of residue to become soil organic matter. The quantity of organic matter in a soil can increase at annual increments until the steady state is reached where the inputs and the outputs are at equilibrium levels. The equilibrium level of soil organic matter is variable and affected by differences in soil texture, mean temperature, growing season precipitation, and type of management practice. Soil organic matter equilibrium level increases when the texture of the soils change from coarse textured sandy soils to finer textured silty and clayey soils; it increases as mean temperatures cool; and it increases as growing season precipitation increases (Weaver et al. 1935, Brady 1974, Parton et al. 1987). The equilibrium level of soil organic matter for any specific soil type in a region can be decreased or increased depending on whether the effects from the type of management practice implemented causes a decrease or an increase in the soil microorganism biomass (Dormaar et al. 1977, Campbell and Souster 1982, Naeth et al. 1991, McLaunchlan et al. 2006).

The quantity of soil organic matter in relation to the equilibrium level is a direct indication of soil primary productivity and is a sensitive measure of changes in soil quality and ecosystem functionality (Burke et al. 1989, Gregorich et al. 1994). Soil organic matter is the primary nutrient reservoir of grazingland ecosystems and contains the organic forms of the essential elements, carbon (C), hydrogen (H), nitrogen (N), oxygen (O), phosphorus (P), and sulfur (S); the ionic forms of the macronutrients, calcium (Ca), magnesium (Mg), and potassium (K); and the ionic forms of the micronutrients, boron (B), copper (Cu), molybdenum (Mo), and zinc (Zn) (Brady 1974, Van Veen and Paul 1981, Burke et al. 1989). Essential elements stored in the soil organic matter as unavailable organic forms or as ions adsorbed by colloidal complexes are biologically and chemically immobilized, respectively, and these stable forms are not subjected to great potential losses through volatilization or leaching movement (Legg 1975, Gibson 2009). Availability of the immobilized essential elements is conducted through the ecosystem biogeochemical cycles performed by soil microorganisms (McGill and Cole 1981, Cheng and Johnson 1998, Manske 2012, 2014). The quantity of available essential elements is determined by the rates of soil organic matter decomposition and mineralization that are directly regulated by the biomass of active soil microorganisms and are not affected by the quantity or rate of residue accession (Van Veen and Paul 1981). Increases in the organic matter content of a soil improves the stability of soil aggregates, improves the physical and chemical properties, improves air and water infiltration and water holding capacity, improves soil fertility, and increases cation exchange capacity (Schimel, Coleman, and Horton 1985, Six et al. 1998, 2004).

This report will evaluate the renewability of grazingland ecosystems in silty soils of the mixed grass prairie that are managed by three types of long-term (32 years) management strategies by comparing differences in the quantities of soil organic matter (SOM), soil organic carbon (SOC), and soil organic nitrogen (SON), and to quantify differences in the soil microorganism rhizosphere volume, the mean monthly amounts of soil available mineral nitrogen, and the mean monthly amounts of transformed mineral nitrogen that result from the combined effects of each of the three different management strategies.

Procedure

Three management treatments were evaluated (1) the long-term nongrazed control was not grazed, mowed, or burned, (2) the 4.5-month seasonlong practice was grazed for 137 days from early June to mid October on one pasture stocked at 2.86 acres per cow-calf pair per month, and (3) the 4.5-month twice-over rotation system was grazed for 137 days from early June to mid October on three rotated pastures stocked at 2.20 acres per cow-calf pair per month. Each of the three management treatments had two replications. The land included in this study had received the same previous management of seasonlong grazing for more than 30 years before the initiation of the research treatments in 1983.

Soil weight of silty soil in southwestern North Dakota was determined from average silty soil bulk density from analysis of comparable soils (Anonymous circa early 1980's) at incremental depths of 0-6, 6-12, 12-24, 24-36, and 36-48 inches. Weight of soil organic matter (SOM) was determined from the soil weight of silty soil and percent soil organic matter from analysis conducted by the North Dakota State University Soil Testing Laboratory of soil core samples from four replicated cores on silty ecological sites of each management treatment at incremental depths of 0-6, 6-12, 12-24, 24-36, and 36-48 inches collected during June of 2013 and 2014. Weight of soil organic carbon (SOC) was determined from the soil weight of silty soil and percent soil organic matter multiplied by 0.58 (58% organic carbon content of soil organic matter) (Anonymous nd, NRCS Staff 2009, Pluske et al. 2015) of soil core samples on silty ecological sites of each management treatment at incremental depths of 0-6, 6-12, 12-24, 24-36, and 36-48 inches. Weight of soil organic nitrogen (SON) was determined from the soil weight of silty soil and percent soil organic matter multiplied by 0.058 (estimated 5.8% organic nitrogen content of soil organic matter) of soil core samples on silty ecological sites of each management treatment at incremental depths of 0-6, 6-12, 12-24, 24-36, and 36-48 inches. Data were reported as mean of two years.

Rhizosphere volume associated with grass roots was determined from two replicated intact soil cores from silty ecological sites on each of the study treatments collected monthly (June to September) during 2002, year 20. Plastic PVC pipe 3 inches (7.62 cm) in diameter and 4 inches (10.16 cm) long was forced into sample site soil. Intact soil-plant cores and pipe were excavated and transported to the laboratory. The soil matrix of collected soil cores was carefully removed from between the rhizospheres. The exposed rhizospheres were sprayed with a clear acrylic coating to prevent damage during further handling. The length and diameter of the rhizosphere around each root of every plant were measured in inches with a vernier caliper, then converted to metric system values. The length and diameter measurements were used to determine the volume of each rhizosphere (Gorder, Manske, and Stroh 2004. Data were analyzed and reported as a mean of the two replications per sample period during August and September.

Soil mineral nitrogen, nitrate (NO_3) and ammonium (NH_4), was determined from four replicated soil core samples collected at silty ecological sites inside protective exclosures for the nongrazed treatment, and outside exclosures exposed to selected treatments for the seasonlong and twice-over rotation grazing management systems with the 1 inch Veihmeyer soil tube at incremental depths of 0-6, 6-12, 12-24 inches on monthly periods during May to October of 2013 and 2014. Analysis of soil core samples for available mineral nitrogen (NO_3 and NH_4) was conducted by the North Dakota State University Soil Testing Laboratory. Two year mean available mineral nitrogen was reported as pounds per acre.

Transformation (immobilization) of nitrate (NO_3) and of ammonium (NH_4) was determined by the net mineralization measurement of the nitrogen balance equation of a soil-plant system (Bloem et al. 2006). The quantity of mineral nitrogen in a soil is the net difference between the total quantity of organic nitrogen mineralized by soil microorganisms and the quantity of mineral nitrogen immobilized by plants and soil microbes (Brady 1974, Legg 1975). The general nitrogen balance equation is simply: the quantity of nitrogen at time 2 minus the quantity of nitrogen at time 1, the difference is the quantity of the transformed nitrogen. Nitrogen quantity at time 1 is the May values. Nitrogen quantity at time 2 is the values at each successive month. Transformed nitrogen is the quantity of uptake by plants and soil microbes and converted into organic nitrogen plus the quantity of nitrogen loss by leaching or volatilization. Loss by leaching on Northern Plains prairies is negligible (Power 1970, Brady 1974, Wight and Black 1979, Coyne et al. 1995). Loss by volatilization during 2013 and 2014 with high soil water content would also be negligible. Transformation of ammonium (NH_4) could include some conversion to nitrate (NO_3). Most of the transformed nitrogen would be the quantity converted into organic nitrogen by plants and soil microbes.

Results

The quantities of soil organic matter (SOM), soil organic carbon (SOC), and soil organic nitrogen (SON) on the silty ecological sites should have been the same on all three management treatments at the start of this study in 1983. Differences in the quantities of these soil organic parameters after 32 years of uninterrupted application of each of the three treatments would be caused by the combined effects that each management strategy had on the soil microorganism biomass and the ecosystem biogeochemical processes and the resulting simultaneous affects on the soil quality and ecosystem productivity.

The quantity of soil organic matter (SOM) to the 48 inch soil depth on the nongrazed treatment was 106.7 tons/ac. The quantity of SOM on the seasonlong treatment was 91.1% greater than that on the nongrazed treatment. The quantity of SOM on the twice-over treatment was 117.0% greater than that on the nongrazed treatment. The twice-over treatment had accumulated a 54.1% greater quantity of SOM than had been accumulated on the seasonlong treatment (figure 1).

The quantity of soil organic carbon (SOC) to the 48 inch soil depth on the nongrazed treatment was 62.0 tons/ac. The quantity of SOC on the seasonlong treatment was 56.2 tons/ac greater than that on the nongrazed treatment. In 32 years, the seasonlong treatment had accumulated 1.8 tons/ac/yr of SOC more than the annual amount that accumulated on the nongrazed treatment. The quantity of SOC on the twice-over treatment was 72.1 tons/ac greater than that on the nongrazed treatment. In 32 years, the twice-over treatment had accumulated 2.3 tons/ac/yr of SOC more than the annual amount that accumulated on the nongrazed treatment. The twice-over treatment had accumulated 0.5 tons/ac/yr of SOC more than the annual amount that accumulated on the seasonlong treatment (figure 2).

The quantity of soil organic nitrogen (SON) to the 48 inch soil depth on the nongrazed treatment was 12,407.4 lbs/ac. The quantity of SON on the seasonlong treatment was 11,234.6 lbs/ac greater than that on the nongrazed treatment. In 32 years, the seasonlong treatment had accumulated 351.1 lbs/ac/yr of SON more than the annual amount that accumulated on the nongrazed treatment. The quantity of SON on the twice-over treatment was 14,418.2 lbs/ac greater than that on the nongrazed treatment. In 32 years, the twice-over treatment had accumulated 450.6 lbs/ac/yr of SON more than the annual amount that accumulated on the nongrazed treatment. The twice-over treatment had accumulated 99.5 lbs/ac/yr of SON more than the annual amount that accumulated on the seasonlong treatment (figure 3).

The soil microorganism biomass cannot be easily measured in pounds per acre. The volume of the rhizosphere cylinders around active perennial grass roots were painstakingly measured on the three management treatments during the growing season of 2002, study year 20. The volume of the rhizosphere is directly proportional to the activity and biomass of the microorganisms (Manske 2011). The rhizosphere volume on the nongrazed treatment during August and September was 2,415.2 cm^3/m^3. The rhizosphere volume on the seasonlong treatment was 532.2 cm^3/m^3 smaller than, but not significantly different ($P < 0.05$) from, the volume on the nongrazed treatment, for a reduction of 22%. The rhizosphere volume on the twice-over treatment was 4,469.5 cm^3/m^3 larger than, and significantly greater ($P < 0.05$) than, the volume on the nongrazed treatment. The rhizosphere volume on the twice-over treatment was 5,001.7 cm^3/m^3 larger than, and significantly greater ($P < 0.05$) than, the volume on the seasonlong treatment. The rhizosphere volume on the twice-over treatment was 185.1% greater than that on the nongrazed treatment and was 265.6% greater than that on the seasonlong treatment (figure 4).

Available soil mineral nitrogen is the major limiting factor of herbage growth on grazingland ecosystems (Wight and Black 1979). A minimum quantity of mineralization of soil organic matter that supplies 100 pounds of mineral nitrogen per acre is required to sustain herbage production at biological potential levels on grazinglands of the Northern Plains (Wight and Black 1972).

The quantity of available soil mineral nitrogen ($NO_3 + NH_4$) to the 24 inch soil depth on the nongrazed treatment was 58.4 lbs/ac, 41.6% lower than the minimum quantity required. The quantity of available soil mineral nitrogen on the seasonlong treatment was 47.1% greater than that on the nongrazed treatment and 14.1% lower than the minimum quantity required. The quantity of available soil mineral nitrogen on the twice-over treatment was 75.7% greater than that on the nongrazed treatment, was 19.4% greater than that on the seasonlong treatment, and was 2.6% greater than the minimum quantity required (figure 5).

The quantity of available soil mineral nitrogen ($NO_3 + NH_4$) varies with changes in soil microorganism biomass and plant phenological growth and development during the growing season (Whitman 1975) and is the net difference between the total quantity of soil organic nitrogen (SON) mineralized by soil microorganisms and the quantity of mineral nitrogen immobilized (transformed) by plants and soil microbes (Brady 1974, Legg 1975). Greater quantities of soil mineral nitrogen transformed indicates greater productivity on the grazingland ecosystem. The quantity of soil mineral nitrogen ($NO_3 + NH_4$) to the 24 inch soil depth that was transformed (immobilized) on the nongrazed treatment was 10.1 lbs/ac during a mean growing season month. The quantity of soil mineral nitrogen transformed on the seasonlong treatment was 155.4 % greater than that on the nongrazed treatment. The quantity of soil mineral nitrogen transformed on the twice-over treatment was 277.2% greater than that on the nongrazed treatment and was 48.0% greater than that on the seasonlong treatment (figure 5).

Discussion

The threshold quantity of a minimum of 100 pounds of mineral nitrogen per acre is required to sustain herbage production at biological potential levels in grazingland ecosystems (Wight and Black 1972, 1979) was the prerequisite standard value necessary to define the proper perspective from which to evaluate these data sets.

The soil microorganism biomass and biological activity on the ecosystems managed with the nongrazed and seasonlong treatments were not large enough to mineralize the minimum threshold quantity of mineral nitrogen. Grazingland ecosystems managed by traditional concepts are not renewable to the good-as-new condition and continue to slowly degrade at various rates because traditional concepts manage for the “use” of only the primary ecosystem components that produce short-term capture of wealth. Traditional concepts do not place importance on the functionality of ecosystem biogeochemical processes. The grazingland ecosystems managed with the nongrazed and seasonlong treatments were degrading at different rates as a result of inferior microorganism biomass and deficient performance of the biogeochemical processes causing reduced quantities of available essential elements and reduced ecosystem productivity.

The nongrazed treatment is not grazed by livestock and quantities of simple carbohydrates greater than the typical low leakage rate are not exudated into the rhizosphere. The nongrazed treatment removed grazing defoliation by livestock for the purpose of resting the grazingland ecosystems as a restoration management practice developed from traditional concepts that do not go deeper than the soil surface. This misguided practice was based on a naive assumption that the observed vigor depletion was caused by livestock grazing, rather than by poor management of livestock grazing, and that the lost vigor could be restored by resting. Ironically, removal of livestock grazing does not rest an ecosystem and enhance the soil microorganism biomass and does not invigorate the degrading ecosystem. The biomass of soil microorganisms on a nongrazed ecosystem can fluctuate with the soil moisture levels but cannot increase above the mass that can be supported by the small quantities of available energy provided in the normal carbohydrate leakage rate and the small amount of short carbon chains and water soluble crude protein remaining within the recently dead plant residue. As a result, the microorganism biomass on the nongrazed ecosystems remains small. Grazingland ecosystems managed by the nongrazed treatment are not renewable.

The seasonlong treatment was managed by traditional concepts that consider the aboveground plants, the livestock, and sometimes wildlife habitat; unfortunately, the importance of the soil microorganism biomass is not even acknowledged. The grazing periods on the seasonlong treatment are not coordinated with grass tiller phenological growth stages and partial defoliation by the grazing livestock does not cause large quantities of simple carbohydrates to be exudated, but does cause quantities greater than typical leakage rates to be exudated into the rhizosphere. Any enhancement of biological activity below the soil surface is purely unintentional. When characteristic seasonlong stocking rates cause quantities greater than 50% of the leaf weight to be removed from grass tillers at vegetative growth stages, the remaining leaf area is insufficient to photosynthesize adequate quantities of carbohydrates to meet the demand for average growth. Without large quantities of exudated simple carbohydrates, the microorganism biomass on seasonlong managed ecosystems remains mediocre. Grazingland ecosystems managed by the seasonlong treatment are not renewable.

The soil microorganism biomass and biological activity on the grazingland ecosystems managed with the twice-over rotation strategy were great enough to mineralize more than 100 pounds of mineral nitrogen per acre. Most of the soil microorganisms that occupy intact soils inhabit the narrow zone around active perennial plant roots; the rhizosphere. Soil microorganism populations existing in the rhizosphere are limited in production by access to energy from simple carbon chains. The grazing periods on the twice-over rotation system are coordinated with the grass tiller phenological growth and development. Partial defoliation by grazing livestock that removes 25% to 33% of the aboveground leaf weight on about 60% to 80% of the grass tillers at vegetative growth stages between the three and a half new leaf stage and the flower stage intentionally causes large quantities of grass leaf photosynthesized simple carbohydrates to be exudated through the roots into the rhizosphere. This great increase in availability of simple carbon compounds in the ecosystems managed with the twice-over rotation system elevate microbe activity and are responsible for the large expansion in the microorganism biomass.

The grazingland ecosystems managed with the twice-over rotation strategy were aggrading and were progressing well along the ecosystem renewal processes towards the good-as-new condition. The rhizosphere volume and, accordingly, the soil microorganism biomass on the twice-over managed ecosystems were enormous and were able to perform the biogeochemical processes at elevated rates greatly increasing the annual accumulation of soil organic matter and annually capturing much greater quantities of essential elements than the amount removed from the ecosystem. The input accumulation of essential elements were greater than the output quantities showing that grazingland ecosystems possibly do not or cannot function exactly according to the steady state at equilibrium levels. The large biomass of microorganisms were able to decompose and mineralize components of the soil organic matter at rapid rates providing great quantities of available essential elements including mineral nitrogen and the macronutrients and micronutrients. The transformation of these large quantities of essential elements back to their organic forms greatly increased ecosystem productivity. The renewal processes on the grazingland ecosystems managed with the twice-over rotation strategy were superior to the biological activity on the ecosystems managed by the seasonlong and nongrazed treatments.

Grazingland ecosystems managed with the twice-over rotation strategy are renewable. These complex ecosystems consist of biotic (living) and abiotic (nonliving) components. The biotic components are the grass plants, soil microorganisms, and grazing livestock that have biological and physiological requirements and have developed complex symbiotic relationships. The abiotic components include radiant energy from sunlight, the major essential elements of carbon, hydrogen, nitrogen, and oxygen with separate but closely linked biogeochemical cycles that transform the elements between organic forms and inorganic forms, and the minor essential elements of the macronutrients and micronutrients. Renewal of grazingland ecosystems requires numerous biogeochemical processes performed by a large biomass of soil microorganisms. A large biomass of active soil microorganisms require large quantities of grass plant produced energy from simple carbon chains. Movement of large quantities of short carbon chain energy from vegetative grass tillers to the soil microorganisms in the rhizosphere requires partial defoliation by grazing livestock coordinated with grass tiller phenological growth stages as accomplished on grazingland ecosystems managed with the twice-over rotation strategy. The cryptobiotic soil microorganisms are vital to the functionality and productivity of grazingland ecosystems; and a large biomass of soil microorganisms are necessary for grazingland ecosystem renewability to the good-as-new condition.

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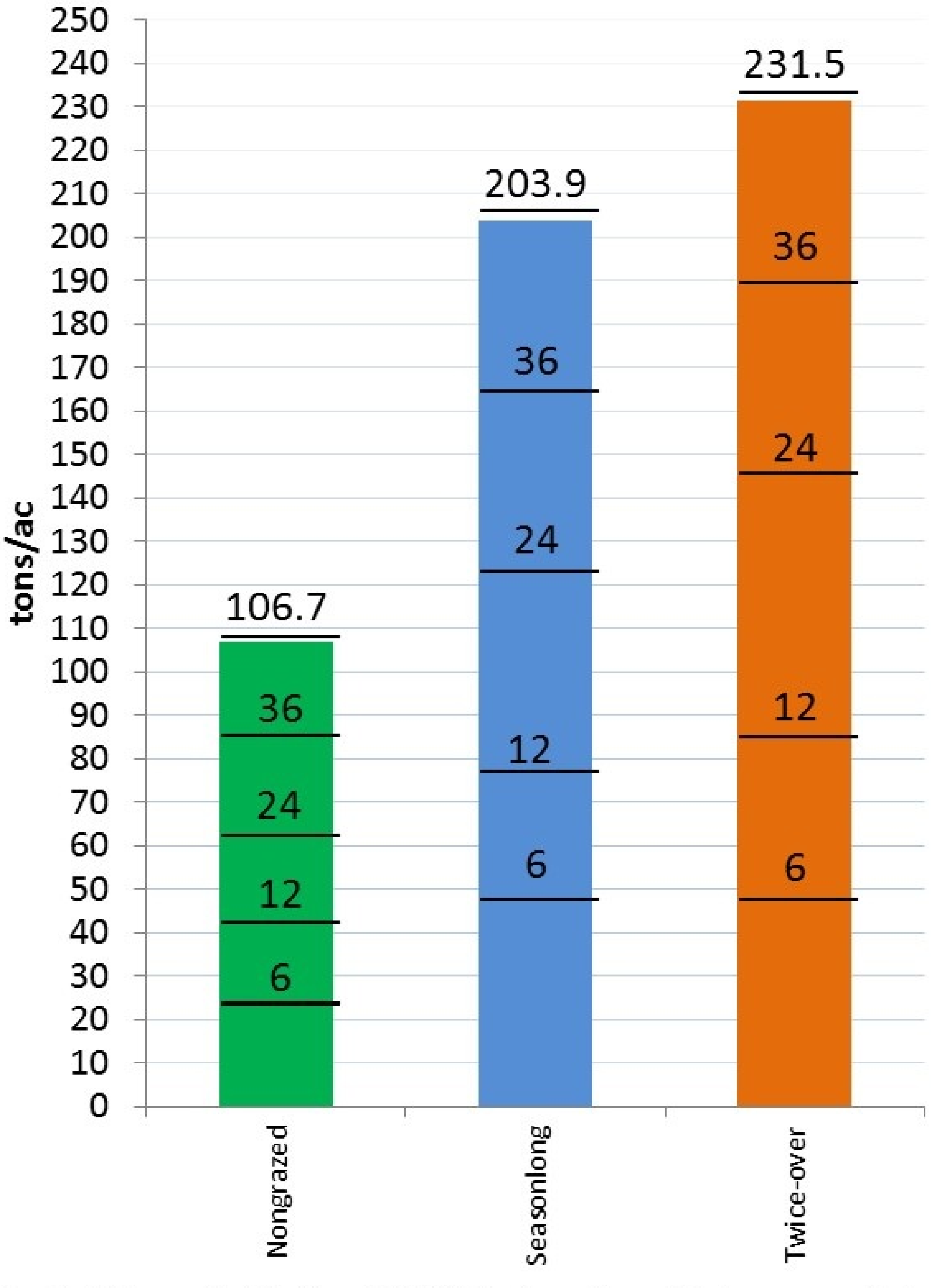

Figure 1. Soil Organic Matter (SOM) in tons/ac at 5 incremental depths to 48 inches on Nongrazed, Seasonlong, and Twice-over rotation management strategies after 32 years.

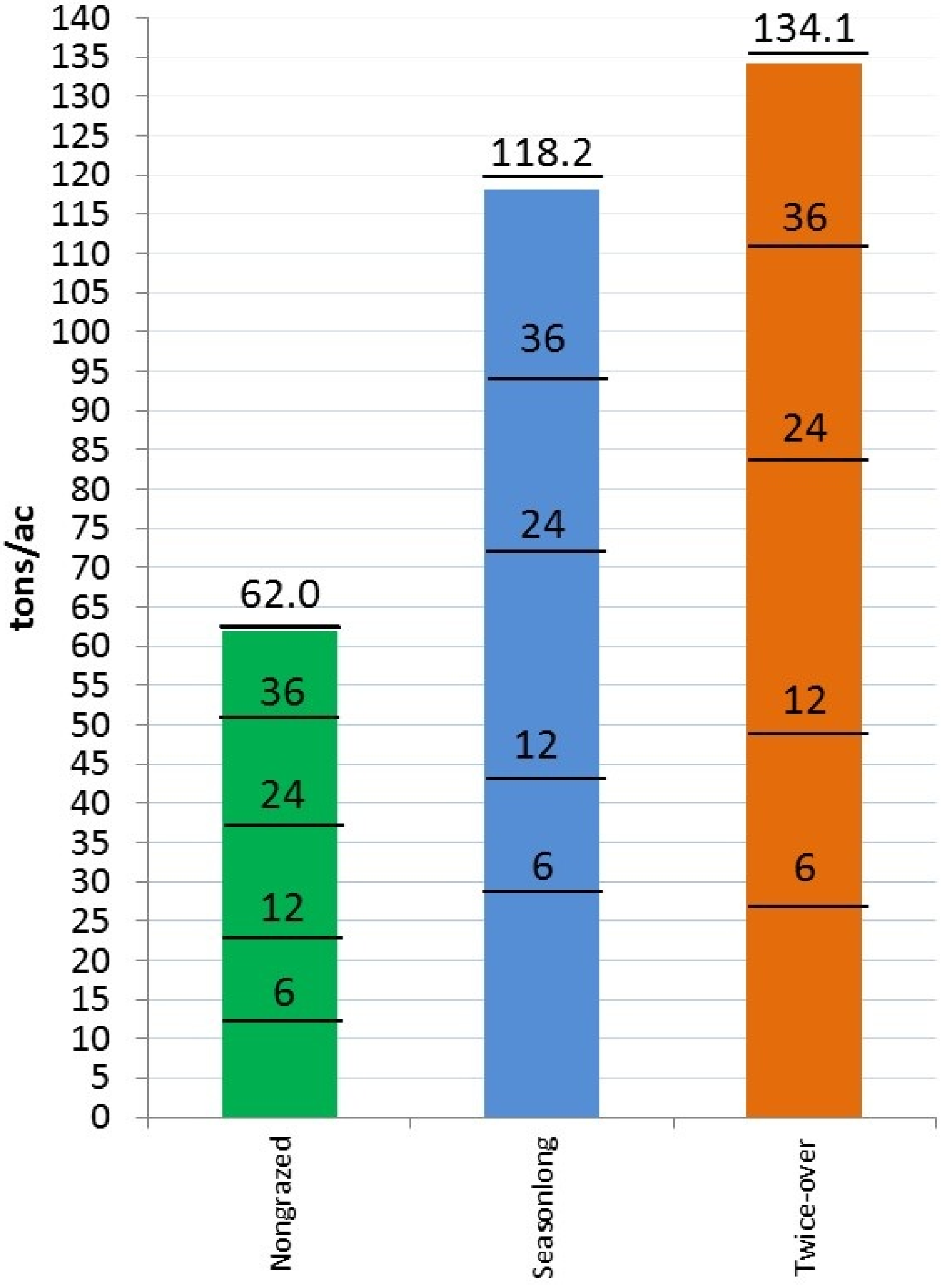

Figure 2. Soil Organic Carbon (SOC) in tons/ac at 5 incremental depths to 48 inches on Nongrazed, Seasonlong, and Twice-over rotation management strategies after 32 years.

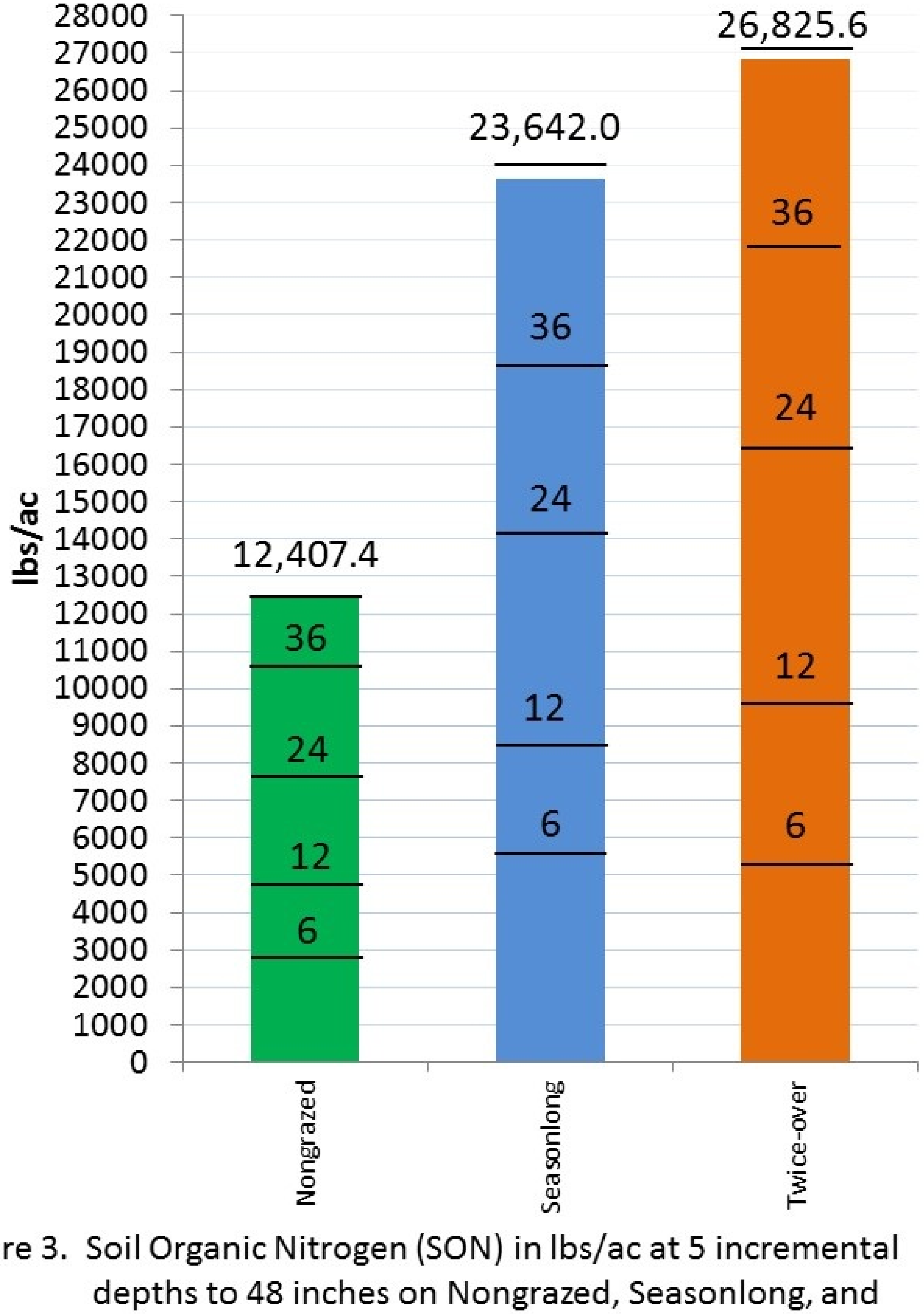

Figure 3. Soil Organic Nitrogen (SON) in lbs/ac at 5 incremental depths to 48 inches on Nongrazed, Seasonlong, and Twice-over rotation management strategies after 32 years.

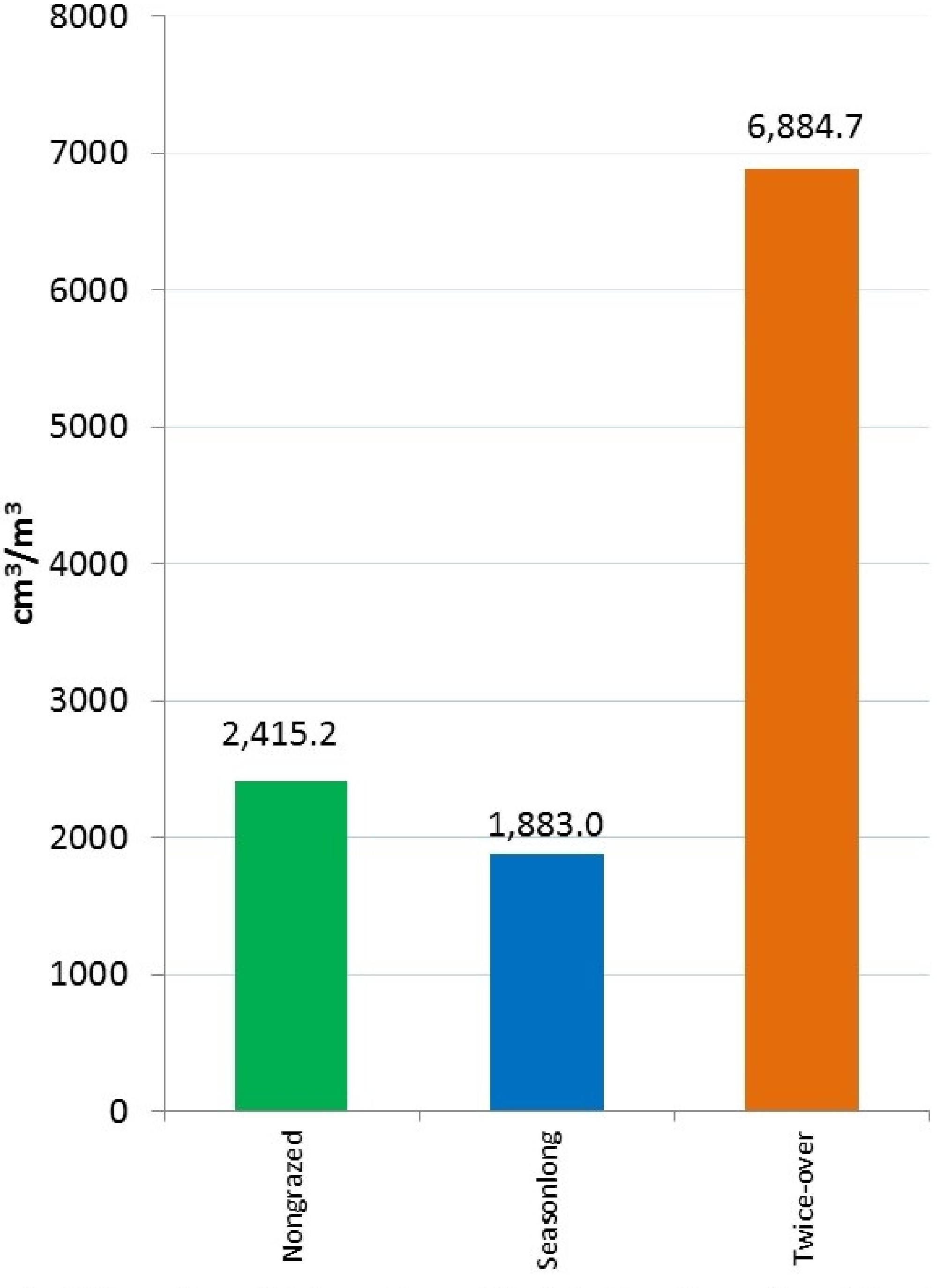

Figure 4. Rhizosphere Volume in cm^3/m^3 during August and September on Nongrazed, Seasonlong, and Twice-over rotation management strategies, 2002.

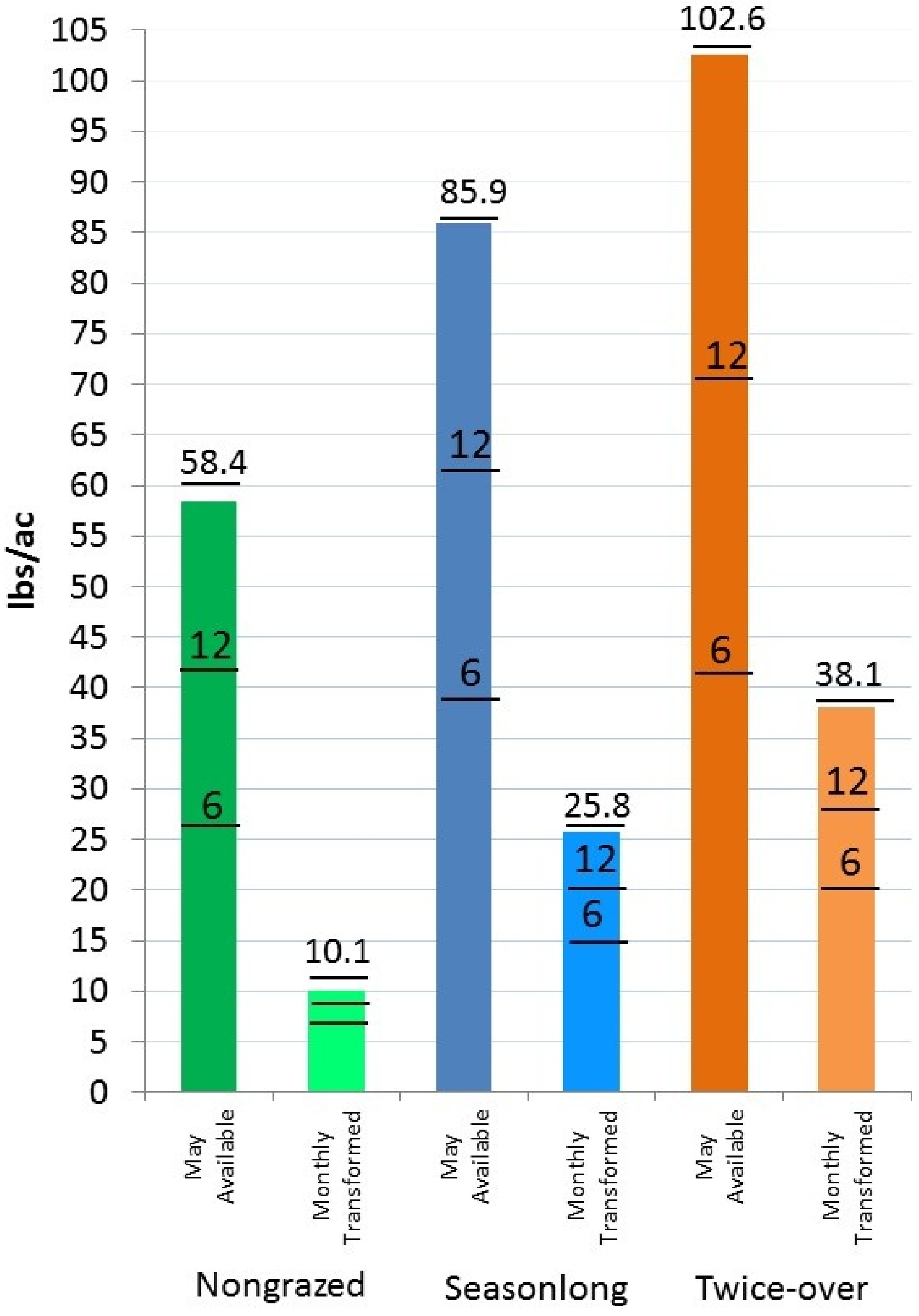

Figure 5. Mineral Nitrogen ($NO_3 + NH_4$) monthly available and transformed in lbs/ac at 3 incremental depths to 24 inches on Nongrazed, Seasonlong, and Twice-over rotation management strategies, 2013-2014.

Literature Cited

Anderson, R.V., D.C. Coleman, C.V. Cole, and E.T. Elliott. 1981. Effect of nematodes *Acrobeloides sp.* and *Mesodiplogaster lheritieri* on substrate utilization and nitrogen and phosphorus mineralization. Ecology 62:549-555.

Anonymous. circa early 1980's. Average bulk density from analysis of comparable soils. In memorandum to Branch Experiment Station Personnel from Department of Soils, NDSU, Staff. 3p.

Anonymous. no date. Soil quality for environmental health. http://soilquality.org/indicators/total_organic_carbon.

Bird, S.B., J.E. Herrick, M.M. Wander, and S.F. Wright. 2002. Spatial heterogeneity of aggregate stability and soil carbon in semi-arid rangeland. Environmental Pollution 116:445-455.

Bloem, J., D.W. Hopkins, and A. Benedetti (eds.). 2006. Microbiological methods for assessing soil quality. CAB International, Oxfordshire, UK. 307p.

Brady, N.C. 1974. The nature and properties of soils. MacMillan Publishing Co., Inc., New York, NY. 639p.

Burke, I.C., C.M. Yonker, W.J. Parton, C.V. Cole, K. Flach, and D.S. Schimel. 1989. Texture, climate, and cultivation effects on soil organic matter content in U.S. grassland soils. Soil Science Society of America Journal 53(3):800-805.

Burrows, R.L., and F.L. Pfleger. 2002. Arbuscular mycorrhizal fungi respond to increasing plant diversity. Canadian Journal of Botany 80:120-130.

Campbell, C.A. and W. Souster. 1982. Loss of organic matter and potentially mineralizable nitrogen from Saskatchewan soils due to cropping. Canadian Journal of Soil Science 62:651-656.

Cheng, W. and D.W. Johnson. 1998. Elevated CO_2, rhizosphere processes, and soil organic matter decomposition. Plant and Soil 202:167-174.

Coleman, D.C., C.P.P. Reid, and C.V. Cole. 1983. Biological strategies of nutrient cycling in soil ecosystems. Advances in Ecological Research 13:1-55.

Coyne, P.I., M.J. Trlica, and C.E. Owensby. 1995. Carbon and nitrogen dynamics in range plants. p. 59-167. *in* D.J. Bedunah and R.E. Sosebee (eds.). Wildland plants: physiological ecology and developmental morphology. Society for Range Management, Denver, CO.

Curl, E.A., and B. Truelove. 1986. The rhizosphere. Springer-Verlag, New York, NY.

Dormaar, J.F., A. Johnston, and S. Smoliak. 1977. Seasonal variation in chemical characteristics of soil organic matter of grazed and ungrazed mixed grass prairie and fescue grasslands. Journal of Range Management 30(3):195-198.

Driver, J.D., W.E. Holben, and M.C. Rillig. 2005. Characterization of glomalin as a hyphal wall component of arbuscular mycorrhizal fungi. Soil Biology and Biochemistry 37:101-106.

Gibson, D.J. 2009. Grasses and grassland ecology. Oxford University Press Inc., New York, NY. 305p.

Gorder, M.M., L.L. Manske, and T.L. Stroh. 2004. Grazing treatment effects on vegetative tillering and soil rhizospheres of western wheatgrass. NDSU Dickinson Research Extension Center. Range Research Report DREC 04-1056. Dickinson, ND. 13p.

Gregorich, E.G., M.R. Carter, D.A. Angers, C.M. Monreal, and B.H. Ellert. 1994. Towards a minimum data set to assess soil organic matter quality in agricultural soils. Canadian Journal of Soil Science 74:367-385.

Hamilton, E.W., and D.A. Frank. 2001. Can plants stimulate soil microbes and their own nutrient supply? Evidence from a grazing tolerant grass. Ecology 82:2397-2402.

Klein, D.A., B.A. Frederick, M. Biondini, and M.J. Trlica. 1988. Rhizosphere microorganism effects on soluble amino acids, sugars, and organic acids in the root zone of *Agropyron cristatum, A. smithii,* and *Bouteloua gracilis*. Plant and Soil 110:19-25.

Legg, J.O. 1975. Influence of plants on nitrogen transformation in soils. pg. 221-227. *in* M.K. Wali (ed.). Prairie: A multiple view. University of North Dakota Press. Grand Forks, ND.

Manske, L.L. 2008. Cow and calf performance as affected by grazing management. NDSU Dickinson Research Extension Center. Range Research Report DREC 08-1052b. Dickinson, ND. 28p.

Manske, L.L. 2011. Biology of defoliation by grazing. NDSU Dickinson Research Extension Center. Range Management Report DREC 11-1067b. Dickinson, ND. 25p.

Manske, L.L. 2012. Degradation and biological restoration of mixed grass prairie ecosystems. NDSU Dickinson Research Extension Center. Summary Range Research Report DREC 12-3058. Dickinson, ND. 16p.

Manske, L.L. 2014. Grazingland management based on native rangeland ecosystem mechanisms and processes. NDSU Dickinson Research Extension Center. Summary Range Management Report DREC 14-3062. Dickinson, ND. 18p.

McGill, W.B., and C.V. Cole. 1981. Comparative aspects of cycling of organic C, N, S, and P through soil organic matter. Geoderma 26:267-286.

McLauchlan, K.K., S.E. Hobbie, and W.M. Post. 2006. Conversion from agriculture to grassland builds soil organic matter on decadal timescales. Ecological Applications 16(1):143-153.

Naeth, M.A., A.W. Bailey, D.J. Pluth, D.S. Chanasyk, and R.T. Hardin. 1991. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland ecosystems of Alberta. Journal of Range Management 44(1):7-12.

NRCS Staff. 2009. Soil quality indicators: Total organic carbon. USDA Natural Resources Conservation Service. http://www.nrcs.gov/wps/portal/nrcs/detail/soils/health/assessment/

Parton, W.J., D.S. Schimel, C.V. Cole, and D.S. Ojima. 1987. Analysis of factors controlling soil organic matter levels in great plains grasslands. Soil Science Society of America 51(5):1173-1179.

Pluske, W., D. Murphy, and J. Shephard. 2015. Total organic carbon. Australian Governments Natural Heritage Trust. www.soilquality.org.au/

Power, J.F. 1970. Nitrogen management of semiarid grasslands in North America. Proceedings of the XI International Grassland Congress. 1970:468-471.

Rillig, M.C., S.F. Wright, and V.T. Eviner. 2002. The role of arbuscular mycorrhizal fungi and glomalin in soil aggregation: comparing effects of five plant species. Plant and Soil 238:325-333.

Schimel, D.S., D.C. Coleman, and K.A. Horton. 1985. Soil organic matter dynamics in paired rangeland and cropland toposequences in North Dakota. Geoderma 36:201-214.

Six, J., E.T. Elliot, K. Paustian, and J.W. Doran. 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. Soil Science Society of America 62:1367-1377.

Six, J., H. Bossuyt, S. Degryze, and K. Denef. 2004. A history of research on the link between (micro) aggregates, soil biota, and soil organic matter dynamics. Soil & Tillage Research 79:7-31.

Van Veen, J.A., and E.A. Paul. 1981. Organic carbon dynamics in grassland soils. 1. Background information and computer simulation. Canadian Journal of Soil Science 61(2):185-201.

Weaver, J.E., V.H. Hougen, and M.D. Weldon. 1935. Relation of root distribution to organic matter in prairie soil. Botanical Gazette 96(3):389-420.

Whipps, J.M. 1990. Carbon economy. p. 59-97. *in*

J.M. Lynch (ed.). The rhizosphere. John Wiley and Sons, New York, NY.

Whitman, W.C. 1975. Native range fertilization and interseeding study. Annual Report. Dickinson Experiment Station. Dickinson, ND. p. 11-16.

Wight, J.R., and A.L. Black. 1972. Energy fixation and precipitation use efficiency in a fertilized rangeland ecosystem of the Northern Great Plains. Journal of Range Management 25:376-380.

Wight, J.R., and A.L. Black. 1979. Range fertilization: plant response and water use. Journal of Range Management 32:345-349.