# Evaluation of Burning and Grazing as Restoration Treatments of Degraded Rangelands



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## **Evaluation of Burning and Grazing as Restoration Treatments of Degraded Rangelands**

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#### **Restoration of Degraded Prairie Ecosystems**

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All prairie ecosystems that are not functioning at potential biological levels are degraded by some degree. Degradation of prairie ecosystems is caused by management practices that are antagonistic to the defoliation resistance mechanisms in perennial grasses. Restoration of degraded prairie ecosystems requires that managers understand the defoliation resistance mechanisms within grass plants, the symbiotic rhizosphere organisms' relationship with ecosystem biogeochemical processes, and the activation of these processes with defoliation by grazing animals.

Grass plants and large grazing herbivores evolved together. During the coevolutionary period, grasses developed biological processes that help plants withstand and recover from defoliation by grazing (Manske 2000a, 2007). Collectively, these processes are the Defoliation Resistance Mechanisms (McNaughton 1979, 1983; Coleman et al. 1983; Briske 1991; Briske and Richards 1995; Manske 1999). These mechanisms are: compensatory physiological processes within grasses (McNaughton 1979, 1983; Briske 1991); vegetative reproduction of secondary tillers from axillary buds (Mueller and Richards 1986; Richards et al. 1988; Murphy and Briske 1992; Briske and Richards 1994, 1995); and symbiotic rhizosphere organism activity (Coleman et al. 1983, Ingham et al. 1985). The defoliation resistance mechanisms accelerate growth rates of replacement leaves and shoots, increase photosynthetic capacity of remaining mature leaves, increase allocation of carbon and nitrogen, increase secondary tiller development from axillary buds, and increase conversion of soil organic nitrogen into plant usable mineral nitrogen.

Defoliation by grazing that removes 25% to 33% of the leaf area of perennial grass tillers at phenological growth stages between the three and a half new leaf stage and the flower (anthesis) stage is needed to trigger these beneficial mechanisms (Manske 1999, 2007), however, a threshold quantity of 100 pounds per acre or greater of mineral nitrogen processed by the rhizosphere organisms is required for full activation of the defoliation resistance mechanisms in grazed grass tillers (Manske 2009, 2010).

Perennial grass plants and rhizosphere organisms interact in the narrow zone of soil around grass roots. The rhizosphere contains bacteria, protozoa, nematodes, mites, springtails, and endomycorrhizal fungi (Elliot 1978, Anderson et al. 1981, Harley and Smith 1983, Curl and Truelove 1986, Whipps 1990, Campbell and Greaves 1990). The grass plants release carbon compounds (Campbell and Greaves 1990), including sugars, through the roots into the rhizosphere, and the soil microorganisms release mineral nitrogen that the plants absorb (Ingham et al. 1985, Clarholm 1985, Biondini et al. 1988, Frederick and Klein 1994, Frank and Groffman 1998). The endomycorrhizal fungi also provide phosphorus, other mineral nutrients, and water that the plant needs for growth (Moorman and Reeves 1979, Harley and Smith 1983, Allen and Allen 1990, Koide 1993, Marschner and Dell 1994, Smith and Read 1997). Activity of the soil microorganisms increases with the availability of carbon compounds in the rhizosphere (Curl and Truelove 1986, Whipps 1990), and the elevated microorganism activity results in an increase in mineral nitrogen available to the grass plant (Coleman et al. 1983, Klein et al. 1988, Burrows and Pfleger 2002, Rillig et al. 2002, Bird et al. 2002, Driver et al. 2005).

Grazing lead tillers between the three and a half new leaf stage and the flower stage (Manske 1999) increases the quantity of carbon compounds the defoliated plants release into the rhizosphere (Hamilton and Frank 2001). The increase in mineral nitrogen produced by elevated rates of microorganism activity allows the plant to accelerate growth and recover more quickly from defoliation. This beneficial activity does not occur when grazing is conducted during the middle and late growth stages of grass plants (Manske 2000a, 2007).

The cause of degradation of prairie ecosystems is antagonistic management practices that are not coordinated with plant phenological growth stages and do not meet the biological requirements of the perennial grass plants and the rhizosphere organisms. Ecosystem deterioration starts with management caused reductions in the quantity of plant carbon exudates released into the rhizosphere; reduced carbon exudates causes a decrease in rhizosphere organism biomass and activity; reduced rhizosphere organism vitality causes a reduction in the quantity of soil organic nitrogen converted into mineral nitrogen. Decreases in the amount of available mineral nitrogen in the ecosystem causes reductions in grass herbage biomass production and causes decreases in plant density (basal cover). When the quantity of mineral nitrogen drops below 100 pounds per acre, the defoliation resistance mechanisms fail to be activated and plant recovery from grazing is incomplete (Manske 2009, 2010). In addition, the water use efficiency processes cease in grass plants growing in ecosystems with less than 100 lbs/ac available mineral nitrogen causing herbage biomass production to be reduced by 49.6% (Wight and Black 1972, 1979). As degradation continues, numerous large bare spaces between grass plants are created in the plant community. These open spaces are ideal habitat for growth of opportunistic "weedy" plant species. The composition of plant species changes with decreases in the desirable species and increases in less desirable species, and later with increases in undesirable species. The change in plant composition from desirable to undesirable species is actually the symptom of ecosystem degradation; the fundamental degradation is the diminishment of ecosystem biogeochemical processes and the reduction of available mineral nitrogen below 100 lbs/ac. The degree of plant species change lags behind the degree of ecosystem biogeochemical degradation.

The greatest antagonistic effects to prairie ecosystems occur from long-term nondefoliation (idle) management that withholds defoliation from a grassland ecosystem. Nondefoliation management results in an accumulation of standing dead leaves that shade lower leaves, increasing the rate of leaf senescence and reducing the rate of photosynthesis, that causes a decrease in the supply of carbohydrates, resulting in a reduction in growth of new leaves and roots (Langer 1972, Briske and Richards 1995). Shading increases grass tiller mortality and reduces grass density (Grant et al. 1983). Reduction of root biomass (Whitman 1974, Brand and Goetz 1986) reduces active root length for interaction with rhizosphere organisms and causes a decrease in absorption of water and nutrients from the soil. Lack of defoliation greatly reduces the quantity of plant carbon exudated into the rhizosphere reducing organism biomass and activity causing a reduction in conversion of soil organic nitrogen into mineral nitrogen.

Under nondefoliation management, dead leaves remain standing for several years. Standing dead plant material not in contact with soil does not decompose through microbial activity. The dead material breaks down slowly as a result of leaching and weathering. A thick mulch layer builds up that modifies soil temperatures, inhibits water infiltration, and retains increasing portions of ecosystem nutrients reducing the quantities of soil organic carbon and nitrogen. Reduced sunlight from shading and reduced quantities of hydrogen, carbon, and nitrogen in the soil, severely reduces the rates of nutrient cycles and biogeochemical processes causing further reductions in leaf growth, tiller leaf area, and grass herbage biomass production. Plant community degradation by nondefoliation promotes changes in composition towards increases in shade-tolerant and shade-adapted replacement species (Manske 2008a).

Prairie ecosystems consist of three components: grass plants, rhizosphere microorganisms, and large grazing herbivores. All three components must be present at sufficient quantities in order for the ecosystem biogeochemical processes to function properly during each growing season. Removal of the large herbivores for one growing season starts the degeneration processes. Degradation of prairie ecosystems caused by nondefoliation management occurs slowly, however, the degree of deterioration to ecosystem biogeochemical processes and plant community structure descends magnitudes greater than any degradation caused by antagonistic grazing management practices (Manske 2008a).

Traditional grazing management practices that are not coordinated with plant phenological growth stages and do not meet the biological requirements of the perennial grass plants and the rhizosphere organisms are detrimental to prairie ecosystems. The deferred grazing, 6.0 month seasonlong, and 4.5 month seasonlong management strategies are antagonistic to rhizosphere organism activity, ecosystem biogeochemical processes, and the nitrogen cycle by causing retardation in the quantity of plant carbon exudation into the rhizosphere. The rhizosphere volume diminishes to levels with insufficient organism biomass and activity to convert soil organic nitrogen into available mineral nitrogen at the threshold rates of 100 lbs/ac. The low rhizosphere volume on the deferred grazing strategy converts an extremely low quantity of mineral nitrogen at 31.2 lbs/ac. The low rhizosphere volume of 49.8 ft<sup>3</sup>/ac on the 6.0-m seasonlong grazing strategy converts a low quantity of mineral nitrogen at 62.0 lbs/ac. The low rhizosphere volume of 67.6

ft<sup>3</sup>/ac on the 4.5-m seasonlong grazing strategy converts a low quantity of mineral nitrogen at 76.7 lbs/ac (Manske 2008b). The defoliation resistance mechanisms are not activated on prairie ecosystems managed with traditional grazing practices that process mineral nitrogen at rates less than 100 lbs/ac, even if the grass tillers are defoliated at the proper phenological growth stages (Manske 2009, 2010). Without help from compensatory physiological processes and from vegetative reproduction of secondary tillers from axillary buds, desirable native grass tillers cannot fully recover from defoliation by grazing; the grass density and grass herbage biomass production progressively decrease and less desirable replacement species backfill as open spaces develop.

Management strategies that defer grazing until after the flowering stage were intended to enhance sexual reproduction and increase the quantity of seeds produced. However, deferred grazing causes a decrease in native grass plant density (Sarvis 1941, Manske et al. 1988). Most young grass plants in grassland ecosystems start not as seedlings but as vegetative tillers that grow from axillary buds on the crowns of an established plant. These vegetative tillers make up the majority of the plant population because they have a competitive advantage over seedlings. Tillers initially draw support from the root systems of parent tillers, while seedlings must rely on their own less-developed structures.

Tiller development from axillary buds is regulated by lead tillers (Briske and Richards 1995), through a process called lead tiller dominance. The lead tillers produce an inhibitory hormone that prevents the growth hormone from activating growth within axillary buds (Briske and Richards 1995). Reduction of the amount of the inhibitory hormone in the plant allows the growth hormone to activate cell growth in multiple axillary buds (Briske and Richards 1994). With that inhibitory hormone reduced, the growth hormone stimulates vegetative reproduction (Murphy and Briske 1992, Briske and Richards 1994), and secondary tillers develop from the axillary buds (Langer 1972). Grazing that removes 25% to 33% of the young leaf tissue from the aboveground portion of lead tillers after the three and a half new leaf stage and before the flower stage increases activation of vegetative tillers from axillary buds (Manske 2007).

All grass species in the Northern Plains have strong lead tiller dominance except Kentucky bluegrass and meadow bromegrass, which have low levels of inhibitory hormones and relatively higher levels of tiller development. Plants with these growth characteristics have greater demand for water than grasses with strong lead tillers and cease growth processes during minor water deficiency periods.

Beneficial grass plant response to grazing depends on the timing of defoliation. Grazing grass plants prior to the three and a half new leaf stage negatively affects grass growth (Manske 2000b). Early seasonal growth of grass plants depends on carbohydrates stored in the roots, rhizomes, and stem bases (Trlica 1977), and prematurely grazed plants are unable to replenish adequate amounts of carbohydrates to support active growth (Coyne et al. 1995, Manske 1999). Grazing after the three and a half new leaf stage and before the flower stage allows plants to establish sufficient leaf area to produce adequate photosynthetic assimilates to meet leaf growth requirements and allows all leaf bud primordia in the apical meristem to develop into leaf buds (Manske 1999).

If no defoliation occurs before the flower stage, as on a deferred grazing strategy, the lead tiller inhibits vegetative tiller development until the inhibitory hormone production naturally declines during the flower stage. This hormone reduction permits one axillary bud to grow and develop into a secondary tiller, which in turn produces inhibitory hormones that prevent growth of the other six to eight axillary crown buds (Mueller and Richards 1986). These dormant axillary buds are never activated and become senescent with the lead tiller. The lack of defoliation of lead tillers prior to the flower stage diminishes recruitment of vegetative tillers, leading to decreased plant density and reduced rhizosphere organism activity; this reduction results in decreased conversion of soil organic nitrogen into mineral nitrogen. No evidence has been found to suggest that grazing the lead tiller after it has reached the flower stage has beneficial stimulatory effects on vegetative tiller development or rhizosphere organism activity (Manske 2000a).

Late season grazing after mid October and early season grazing before the three and a half new leaf stage are antagonistic to native perennial grasses. Late grazing is not harmful to senescent lead tillers that produced seeds that growing season. Usually around 25% of the tiller population reaches the reproductive phenological growth stage, however, it can vary from 5% to 50% of the tiller population. The other tillers that did not produce seeds will overwinter and resume active growth during the subsequent growing season. Late season grazing of carryover tillers causes decreased tiller numbers, reduced total basal area, and reduced quantities of herbage biomass produced (Olson and Richards 1988, Coyne et al. 1995). Winter survival of carryover tillers depends on having adequate carbohydrate reserves which is closely related to the amount of active leaf material remaining on each tiller during the winter hardening process, that occurs between mid August and mid October. The crown, portions of the root system, and some leaf tissue remain active and maintain physiological processes throughout the winter using stored carbohydrates. Depletion of the carbohydrates reserves before spring causes tiller death, "winter kill".

During early spring, portions of the carryover tillers' leaves from the previous year that have intact cell walls regreen with chlorophyll and provide crucial photosynthetic product for new leaf growth (Briske and Richards 1995). New growing leaves draw carbohydrates from the carryover older leaves until maintenance and growth requirements can be met by the new leaves (Langer 1972, Coyne et al. 1995). Removal of leaf matreial by grazing from grass tillers not yet at the three and a half new leaf stage deprives the new tillers of foliage needed for photosynthetic product and causes a demand on the low levels of carbohydrate reserves that results in a reduction of the herbage biomass production well below potential quantities later in the growing season (Campbell 1952, Rogler et al. 1962, Manske 2000b).

The twice-over rotation grazing management system is the biologically effective management strategy that is coordinated with grass phenological growth stages and meets the biological requirements of the perennial grass plants and soil organisms by applying defoliation treatment to grass plants at the appropriate growth stages that activate the defoliation resistance mechanisms and stimulate the symbiotic rhizosphere microorganisms biomass and activity (Manske 1999, 2000a; Gorder, Manske, Stroh 2004). The increased rhizosphere activity results in an increased quantity of available mineral nitrogen. The high rhizosphere volume of 227.1 ft<sup>3</sup>/ac on the twiceover rotation grazing strategy converts a high quantity of mineral nitrogen at 177.8 lbs/ac (Manske 2009, 2010).

The twice-over rotation grazing management system uses three to six pastures. Every pasture is grazed during two periods per growing season. Each pasture is grazed for 7 to 17 days during the first period, the 45-day interval from 1 June to 15 July. The length of the first period on each pasture is the same percentage of 45 days as the percentage of the total season's grazeable forage each pasture contributes (Manske 2000a). During the second period, after mid July and before mid October, each pasture is grazed for double the number of days it was grazed during the first period. Livestock are removed in mid October.

The coordinated defoliation improves plant health and stimulates biological and ecological processes within grass plants and the ecosystem so that beneficial changes to plant growth, soil organisms, and biogeochemical cycles in the ecosystem result (Manske 2000a). During the first grazing period, grasses are between the three and a half new leaf and flower stages, the stages of plant development at which grazing stimulates the defoliation resistance mechanisms that increase tillering from axillary buds and enhance rhizosphere organism activity increasing the conversion of soil organic nitrogen into mineral nitrogen. Increased vegetative reproduction by tillering contributes to the development of greater plant basal cover and to the production of greater grass herbage weight; increased activity of the soil organisms in the rhizosphere supplies the plant with greater quantities of nutrients to support additional growth (Manske 2000a).

Restoration of degraded prairie ecosystems requires implementation of a biologically effective grazing management strategy that meets the biological requirements of the perennial grass plants and rhizosphere organisms, and that is coordinated with perennial grass phenological growth stages. Proper defoliation during the first grazing period removes 25% to 33% of the leaf area of grass tillers between the three and a half new leaf stage and the flower stage. This defoliation by grazing treatment increases the quantity of plant carbon exudates released into the rhizosphere causing an increase in the biomass and activity of the rhizosphere organisms that results in an increase in the quantity of soil organic nitrogen converted into mineral nitrogen. Increases in the amount of available mineral nitrogen processed by rhizosphere organisms in the degraded prairie ecosystem results in increases in grass herbage biomass production and increases in plant density (basal cover) of the desirable native grass species. When the quantity of mineral nitrogen increases to 100 pounds per acre or greater, the defoliation resistance mechanisms can be fully activated that accelerate growth rates of replacement leaves and shoots, increase photosysnthetic capacity of remaining mature leaves, increase allocation of carbon and nitrogen, and increase secondary tiller development from axillary buds. The water use efficiency processes also are activated in grass plants resulting in an increase of 50.4% in herbage biomass production per inch of precipitation received. As

restoration continues, native perennial grass species composition increases in the plant community. The native grasses out compete the less desirable and the undesirable species for space and nutrient resources because of the advantages provided by the symbiotic relationship with the rhizosphere organisms. The restoration of the native plant species composition lags behind the restoration of the prairie ecosystem biogeochemical processes and the recovery of the nitrogen cycle.

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#### Effects from prescribed burning treatments on mixed grass prairie

Llewellyn L. Manske PhD Range Scientist North Dakota State University Dickinson Research Extension Center Report DREC 07-3044

#### Introduction

Restoration of degraded grassland ecosystems requires the reactivation of the complex biological and ecological processes within native plants and within the rhizosphere organisms that convert soil organic nitrogen into inorganic nitrogen.

The primary cause of deterioration in grassland ecosystems is management practices that are antagonistic to the rhizosphere organism population. Decreases in rhizosphere organism biomass result in reductions in the quantity of organic nitrogen converted into inorganic nitrogen; this conversion is one of the primary functions of rhizosphere organisms. Decreases in the amount of inorganic nitrogen in an ecosystem cause reductions in grass biomass production and decreased native plant density (basal cover), creating larger and more numerous bare spaces between grass plants. These open spaces in the plant community provide ideal habitat for growth of opportunistic "weedy" plant species that are not dependent on the nitrogen converted by rhizosphere organisms. Once established, most opportunistic weedy species have mechanisms that aid in widening the species' distribution; the spread of the weeds indicate further degradation of the grassland ecosystem.

Additions of mineral (inorganic) nitrogen fertilizers to native grassland soils are antagonistic to rhizosphere organism populations, causing greater ecosystem degradation and pushing the plant species composition to be dominated by domesticated coolseason grasses like smooth bromegrass, crested wheatgrass, and Kentucky bluegrass. Other "quick fix" practices that treat only symptoms of the problem and do not correct the cause of the problem also result in further degradation of the ecosystem.

The solution for restoration of degraded grassland ecosystems is to correct the cause of the problem rather than just treat the symptoms of the problem. Grazing management coordinated with grass plant phenological development has been shown to stimulate rhizosphere organism increases in biomass and activity levels (Gorder, Manske, and Stroh 2004, Manske 2005), resulting in increased quantities of inorganic nitrogen (Coleman et al. 1983).

This study was conducted to investigate the possibilities of using prescribed burning treatments in the restoration of degraded mixed grass prairie ecosystems.

#### Study Area

The study area was the Lostwood National Wildlife Refuge, located in Burke and Mountrail counties in northwestern North Dakota between 48° 50' and 48° 30' north latitude and 102° 40' and 102° 20' west longitude. The landscape is glacial terminal moraine of the Missouri Coteau. Topography is rolling to steep hills interspersed with shallow lakes and prairie wetlands. Soils are primarily fine-loamy, mixed Typic Haploboralls and fine-loamy, mixed Typic Argiboralls. Some areas have sandy or gravelly substratum. Native vegetation is the Wheatgrass-Needlegrass Type (Barker and Whitman 1988) of the mixed grass prairie.

The region was homesteaded between 1910 and 1930. During that period, about 25% of the upland was plowed and used as cropland. Most likely, grazing livestock had access to the remainder of the area. Naturalists' surveys of the region conducted in 1913-1915 described one lone grove of trees located at the southeastern corner of the refuge's lower long lake (Smith 1997). The trees were cut down for their wood, and the lake became known as lower Lostwood Lake.

Because of economic troubles of the time, the federal government developed relief programs to repurchase failed homestead land during the mid to late 1930's. The homestead acres repurchased under land utilization projects were designated for three specific purposes. The acres identified for grazing use and economic development from livestock agriculture became the Little Missouri National Grasslands, acres identified for recreational use became Theodore Roosevelt National Park, and acres identified for wildlife use became Lostwood National Wildlife Refuge.

The Lostwood Wildlife Refuge consists of 26,904 acres (10,896 ha) with 5,381 acres (2,179 ha) of wetlands and 5,577 acres (2,259 ha) of wilderness (Smith 1997). Early management of the refuge was based on the concept of preserving wildlife habitat with little or no disturbance (idle), and about 15% of the refuge has never been burned, grazed, or mowed.

All grazing on the refuge was stopped between 1935 and 1940. After 1940, about 59% of the refuge was grazed periodically using deferred seasonlong management with some areas grazed only one time and other areas grazed as many as 22 times over a 35-year period. Between 1940 and 1975, about 26% of the refuge was annually grazed with seasonlong management for 4.5 to 5.0 months at low to moderate stocking rates, primarily during July through November (Smith 1988). This deferred-type management that delays grazing until after the flowering stage of grasses is known to decrease grass tiller density (Sarvis 1941, Manske et al. 1988).

After 1935, some of the land parcels previously used as cropland by homesteaders (about 8% of the refuge) were allowed to "go back" by natural revegetation through secondary succession. The remaining cropland parcels were managed as cropland until the mid 1950's, when about 15% of the refuge was reseeded with domesticated cool-season grasses, primarily smooth bromegrass and crested wheatgrass (Smith 1988).

The available records indicate that the inhabitants of the region had suppressed all fires from sometime in the late 1800's and that refuge land had not been burned in over 80 years by wildfire or prescribed fire until a prescribed burning program was started in 1978 (Smith 1985b).

There were few trees on the refuge during the 1930's and 1940's, but by 1985, there were over 540 expanding aspen groves covering about 475 acres (192 ha) interspersed across the landscape located at the edges of seasonal wetlands, with about 300 aspen groves completely occupying previous wetland basins (Smith 1997).

The shrub cover on Lostwood Wildlife Refuge increased from about 5% during the mid 1930's to greater than 50% in 1979. This change, however, did not occur at a uniform rate. The shrub composition in the plant community did not change much during the first 20 years. A substantial increase in shrub cover occurred between 1953 and 1969 and, between 1969 and 1979, the western snowberry colonies expanded rapidly and invaded extensive areas of degraded grassland; as a result, over half of the refuge upland was transformed into shrubland (Smith 1988). Kentucky bluegrass was the dominant grass associated with the western snowberry colonies. Large portions of the western snowberry colonies were extremely dense and had no herbaceous understory. Decadent centers of old western snowberry colonies had been reinvaded by smooth bromegrass, quackgrass (Smith 1985a), and Canada thistle (Smith 1985b.). Native grasses and forbs were still present in low quantities in some areas but were greatly suppressed.

Refuge manager Karen Smith initiated an every-other-year prescribed burning strategy that was conducted from 1978 to 2002. The every-other-year burn regime was designed to reduce the invading western snowberry and exotic grasses and renovate the prairie ecosystem. Annual burns were not possible because of insufficient production of plant biomass for fuel (Smith 1985a). The refuge was subdivided into prescribed burn management units that used trails or mowed swaths as fire breaks. Several parcels of the refuge received no burning treatments and were used as reference control areas. The prescribed burns were conducted during four seasons: early spring (mid-late April), 1 replication; spring (May-mid June), 3 replications; early summer (mid June-July), 7 replications; and mid summer (early-mid August), 4 replications. The number of repeated every-other-year burns was 1 burn, 4 replications; 2 burns, 4 replications; 3 burns, 4 replications; and 4 burns, 3 replications. Control treatments, 6 replications, had no burning (wildfires or prescribed burns) for over 100 years, since sometime during the late 1800's. In 1990, Manske (1992) evaluated the effects of every-other-year prescribed burning after thirteen years of treatments (1978-1990). This report is a summary of that study.

#### Procedures

Field data were collected on permanent landscape transects that included the plant communities on the summit, shoulder, back, foot, and toe slopes from 15 prescribed burn management units with an average size of 530.5 acres (214.85 ha) and 6 control management units of no burning with an average size of 436.8 acres (176.90 ha). A standard paired plot t-test was used to analyze differences between means (Mosteller and Rourke 1973).

Aboveground herbage biomass was collected during peak growth in mid to late July by the standard clipping method (Cook and Stubbendieck 1986). The herbage material from three  $0.25 \text{ m}^2$  quadrats (frames) located along the landscape transects for each treatment was sorted in the field by biotype categories: grasses, sedges, forbs, shrubs, and standing dead. The herbage of each biotype category from each frame was placed in labeled paper bags of known weight, oven dried at  $176^{\circ}$  F (80° C), and weighed.

Plant species composition was determined during peak growth, between mid July and mid August, by the plant shoot cover method (% shoot frequency) (Cook and Stubbendieck 1986), with one hundred  $0.1 \text{ m}^2$  quadrats placed systematically along the landscape transects of each treatment.

Endomycorrhizal fungal infection in roots was evaluated for blue grama, western wheatgrass, smooth bromegrass, and western snowberry. Three replicated soil cores 4 inches (10.2 cm) in diameter and 4 inches (10.2 cm) in depth were collected for each of the four plant species; samples from nearly level loam soils along the permanent landscape transects of each control and prescribed burn treatment were taken with a golf cup cutter. Roots were washed over sieving, and current year's roots were removed from plant crowns by clipping. Root samples of each replicate were stored in individual vials and preserved in a solution of glycerin and lactic acid. In the laboratory, root samples were cleared and stained to enhance mycorrhizal structures using procedures described by Phillips and Hayman (1970) and modified by Kormanik and McGraw (1982). Fungal colonization in the root samples was scanned through a Nikon 107733 type 104 microscope, and percent fungal infection was assessed using a nonsystematic modification of the grid-intersect method (Giovannetti and Mosse 1980), with presence or absence (P/A) of fungal structures recorded for 100 intersected root segments.

Changes in soil microorganism activity were monitored by change in the quantity of soil inorganic (mineral) nitrogen in July and August. Five replicated soil cores 1 inch (2.54 cm) in diameter and 6 inches (15.24 cm) in depth were collected from nearly level loam soils along the permanent landscape transects of each control and prescribed burn treatment and air dried. In the laboratory, subsamples of the soil cores were evaluated for total incubated mineralizable nitrogen (N) with procedures outlined by Keeney (1982). Inorganic forms of nitrogen were extracted from soil samples by adding a reagent, 2 M KCl, at the rate of 50 ml/10 g of soil; samples were then shaken for one hour. The extract was analyzed for ammonium (NH<sub>4</sub>) and nitrate (NO<sub>3</sub>) by steam distillation (Keeney and Nelson 1982). Soil nitrite (NO<sub>2</sub>) was not analyzed because it is seldom present in detectable quantities and no methods had been developed that were adequately sensitive to obtain reliable estimates of nitrite.

Gravimetric soil water data (Cook and Stubbendieck 1986) were collected on both the summit slopes and foot slopes of each landscape transect in July and August using a 1-inch (2.54 cm) Veihmeyer soil tube at depths of 0-6, 6-12, and 12-24 inches (0-15.24, 15.24-30.48, and 30.48-60.96 cm) and oven dried at  $212^{\circ}$  F (100° C). Weather data for the region were collected at the Des Lacs NWR weather station.

#### Results

The Lostwood Wildlife Refuge region has cold winters and hot summers typical of continental climates. January was the coldest month, and July and August were the warmest months. Plants experience temperature stress during months with mean monthly temperatures below  $32.0^{\circ}$  F ( $0.0^{\circ}$  C). From November through March each year, plants in northwestern North Dakota cannot conduct active growth because mean temperatures are below  $32.0^{\circ}$  F ( $0.0^{\circ}$  C).

The long-term (1936-1989) annual precipitation at the Lostwood Wildlife Refuge region is 16.50 inches (419.10 mm). The growing-season precipitation (April through October) is 13.80 inches (350.52 mm), 83.64% of the annual precipitation. The seasonal period during which the greatest precipitation occurs is spring–April, May, and June–with 6.62 inches (168.15 mm), 40.12% of the annual precipitation. June has the greatest monthly precipitation received during the 3-month period of May, June, and July accounts for 46.67% of the annual precipitation (7.70 inches, or 195.58 mm). The precipitation received during the 5-month period of November through March averages 2.70 inches (68.58 mm), 16.36% of the annual precipitation. The seasonal period during which the least precipitation occurs is winter–January, February, and March–with 1.69 inches (42.93 mm), 10.24% of the annual precipitation.

Annual precipitation during the study period (1978-1990) averaged 15.36 inches (390.14 mm), 93.10% of the long-term mean (LTM), and growingseason precipitation, April though October, averaged 12.35 inches (313.69 mm), 89.48% of the LTM growing-season precipitation (table 1). The growingseason precipitation was greater than the long-term mean during 1978, 1982, 1984, 1985, 1986, and 1990, and lower than the long-term mean during 1979, 1980, 1981, 1983, 1987, 1988, and 1989 (table 1). The growing-season precepitation during 1979, 1983, 1987, and 1988 was 48.2%, 62.2%, 60.2%, and 69.6% of the long-term mean, respectively (table 1). The water deficiencies during 1979, 1983, 1987, and 1988 caused water stress in perennial plants so that herbage biomass production in the region was restricted.

The quantity of soil water during the growing season (July and August), to 24 inches (60.96 cm) in depth, was not significantly different among the no burn control treatments and the number of repeated every-other-year burn treatments (table 2) and the seasonal period of every-other-year burn treatments (table 3).

The total current year's production of aboveground biomass was not different after one, two, three, and four repeated prescribed burns compared to the biomass of the unburned treatment (table 4). However, the composition of the aboveground biomass changed remarkably. The contribution of grasses to the total biomass changed from 24.2% on treatments with no burns to 65.6% after four burns (table 4), an increase of 171.1%. Grass biomass decreased 24.7% after one burn and increased 109.3% after four burns (table 4). The contribution from sedges changed from 13.2% on treatments with no burns to 11.1% after four burns. Sedge biomass increased 61.6% after one burn and decreased 35.1% after four burns (table 4). The contribution from forbs changed from 15.0% on treatments with no burns to 20.3% after four burns. After one burn, the forb contribution to total aboveground biomass was 139.7% greater than that on the unburned treatments. After two and three burns, the weedy forbs decreased and the ecological status of perennial forbs improved. The forb contribution to the total biomass production after four burns was 35.3% greater than that on the unburned treatments. Forb biomass increased 78.0% after one burn and increased 4.4% after four burns (table 4). The biomass contribution from shrubs changed from 47.5% on treatments with no burns to only 3.0% after four burns (table 4), a 93.7% decrease. Shrub biomass decreased 83.1% after one burn and decreased 95.1% after four burns (table 4).

Native grass shoot frequency increased significantly as a result of repeated burning. The average increase after one, two, and three burns was 79.6%; after four burns, native grass shoot frequency increased 94.7% (table 5). The quantity of basal cover area for native grasses, however, was not well developed even after four burns. Sedge shoot frequency increased an average of 58.4% after repeated burning. Introduced grass shoot frequency decreased an average of 49.4% after one, two, and three burns and decreased 65.1% after four burns. Four burns were required to reduce introduced grasses significantly (table 5). Kentucky bluegrass shoot frequency decreased an average of 36.2% after one, two, three, and four burns. Quackgrass shoot frequency decreased an average of 84.0% after one and two burns and decreased an average of 90.9% after three and four burns. Smooth bromegrass shoot frequency decreased an average of 90.0% after one and two burns and decreased an average of 96.7% after three and four burns.

Perennial forb shoot frequency increased 39.3% after one burn (table 5) and increased an average of 7.5% after additional repeated burns of two, three, and four times. Early succession and weedy forb shoot frequency increased 8.2% after one burn, decreased an average of 7.5% after two and three burns, and decreased 50.9% after four burns (table 5). Four burns were required to reduce weedy forbs significantly (table 5).

Shrub shoot frequency decreased 36.4% after one burn, decreased an average of 46.1% after two and three burns, and decreased 58.2% after four burns (table 5). Four burns were required to reduce shrubs significantly (table 5).

Western snowberry shoot frequency decreased 62.7% after one burn, decreased an average of 55.8% after two and three burns, and decreased 64.0% after four burns. Shoot frequency of western snowberry changed little from repeated burning after the first burn. However, the aboveground biomass produced by the shrubs was greatly reduced after the third and fourth burns.

All burns cause some damage to plants, but the seasonal period that prescribed burns are conducted affects the biomass production and shoot frequency of plant biotypes differently. Effective prescribed burns are conducted during appropriate seasonal periods so that the greatest reduction to the undesirable plants is caused and the damage to the desirable plants is minimized.

Grass biomass greatly increased after spring (May-mid June) and mid summer (early-mid August) burns but decreased after early summer (mid June-July) burns (table 6). Grass shoot frequency increased significantly after burns conducted during all seasonal periods. The greatest increases occurred after spring (May-mid June) and mid summer (earlymid August) burns (table 7). Shoot frequency of native cool-season grasses increased significantly after burns conducted during all seasonal periods (table 8). Shoot frequency of western wheatgrass increased significantly after early spring (mid-late April) burns and decreased after spring (May-mid June) burns (table 8). Shoot frequency of native warm-season grasses increased significantly after burns conducted during spring (May-mid June) (table 8). Blue grama shoot frequency increased significantly after spring (May-mid June) burns (table 8). Shoot frequency of introduced grasses decreased significantly after spring (May-mid June) burns (table 7). Kentucky bluegrass shoot frequency decreased significantly after spring (May-mid June) burns (table 8) and increased after early spring (mid-late April) burns (table 8). Smooth bromegrass and quackgrass shoot frequency decreased after burns conducted during all seasonal periods. Smooth bromegrass shoot frequency decreased most after early spring (mid-late April) burns (table 8). Quackgrass shoot frequency decreased most after spring (May-mid June) burns (table 8).

Sedge biomass increased after burns conducted during early spring (mid-late April) and early summer (mid June-July) and decreased after spring (May-mid June) and mid summer (early-mid August) burns (table 6). Shoot frequency of sedges increased significantly after spring (May-mid June) burns (table 7).

Forb biomass increased after burns conducted during all seasonal periods. The greatest increases occurred after early spring (mid-late April) and spring (May-mid June) burns (table 6). Shoot frequency of perennial forbs increased after early spring (mid-late April) and early summer (mid June-July) burns and decreased slightly after spring (Maymid June) burns (table 7). Shoot frequency of weedy forbs increased significantly after early spring (midlate April) burns and decreased after spring (May-mid June), early summer (mid June-July), and mid summer (early-mid August) burns (table 7).

Shrub biomass decreased after burns conducted during all seasonal periods. The greatest decreases occurred after early spring (mid-late April), spring (May-mid June), and mid summer (early-mid August) burns (table 6). Shoot frequency of shrubs decreased significantly after early spring (mid-late April), spring (May-mid June), and mid summer (early-mid August) burns (table 7). Shrub shoot frequency did not decrease significantly after early summer (mid June-July) burns (table 7). Silverberry shoot frequency decreased significantly after early spring (mid-late April) and mid summer (early-mid August) burns (table 9). Western rose shoot frequency decreased significantly after early spring (mid-late April) and spring (May-mid June) burns (table 9) and increased slightly after early summer (mid June-July) and mid summer (early-mid August) burns. Shoot frequency of western snowberry decreased significantly after early spring (mid-late April), spring (May-mid June), and mid summer (early-mid August) burns (table 9). Western snowberry shoot frequency did not decrease significantly after early summer (mid June-July) burns (table 9).

Endomycorrhizal fungi do not colonize the entire root. Fungal colonization occurs at the portions of current year's roots that are biologically active. Previous years' roots, mature root portions, and young growing root portions do not host fungal structures. Percent fungal infection of root segments is primarily a factor of the proportion of biologically active root portions to the amount of mature and young root portions included in the sample. Identification of biologically active root portions from mature root portions is difficult in the field, with the naked eve or low-power hand lenses. Basically, the less than 100% fungal infection in the blue grama, western wheatgrass, and western snowberry root samples in tables 10 and 11 should be considered to indicate the percent biologically active root portions within the root sample. This evaluation, however, is not applicable to the smooth bromegrass root samples. Even though the smooth bromegrass root samples contained a small amount of young and mature root portions, most of the samples consisted of biologically active root portions and the percent fungal infection in tables 10 and 11 should be

considered to be close to the percent fungal infection in smooth bromegrass samples. However, almost all of the fungal infection observed in the smooth bromegrass samples was restricted to the root hairs. Very few smooth bromegrass samples had fungal colonization within the root tissue. Even with the large differences in proportions of biologically active root portions within the root samples, the percent fungal infection in blue grama, western wheatgrass, and western snowberry root samples was significantly greater than the percent fungal infection in smooth bromegrass root samples on the number of repeated every-other-year prescribed burn treatments, the seasonal period of every-other-year prescribed burn treatments, and the control no burn treatments. Smooth bromegrass had virtually no fungal infection in the biologically active root tissue and relatively low fungal infection in the root hairs. The percent fungal infection in the roots of blue grama, western wheatgrass, western snowberry, and smooth bromegrass was not changed significantly by the number of repeated every-other-year prescribed burn treatments and the seasonal period of every-otheryear prescribed burn treatments from the percent fungal infection in the respective plant species on the control no burn treatments (tables 10 and 11). The quantity of endomycorrhizal fungal colonization in plant roots was not stimulated by the prescribed burning treatments.

Grassland soils have abundant quantities of nitrogen; however, most of it is in the organic form and unavailable for direct use by plants. Grassland plants can use nitrogen only in the inorganic (mineral) form. Soil microorganisms of the rhizosphere convert soil organic nitrogen into inorganic nitrogen (Ingham et al. 1985). Grassland ecosystems with greater biomass of rhizosphere organisms convert greater quantities of organic nitrogen into inorganic nitrogen (Coleman et al. 1983). The quantities of inorganic nitrogen (NH<sub>4</sub>-NO<sub>3</sub>) in the soils on the number of repeated every-other-year prescribed burn treatments and the seasonal period of every-otheryear prescribed burn treatments were not changed significantly from the quantities of inorganic nitrogen in the soils on the control no burn treatments (tables 10 and 11). The rhizosphere microorganism biomass and activity levels were not stimulated by the prescribed burning treatments, and the quantity of organic nitrogen converted into inorganic nitrogen was not stimulated by the prescribed burning treatments.

#### Discussion

Western snowberry aerial stems are sensitive to fire. The top growth is usually removed completely if sufficient fine fuel is present, and even if the stems are not completely consumed by the fire, they usually die to ground level. The belowground rhizomes and rhizome crowns with clusters of aerial stems are usually not damaged by fire. The belowground parts have large quantities of buds that have the potential to develop into new aerial sucker stems. Spring burns result in great quantities of sucker stems, which become visible about two weeks following the burn, and because carbohydrate stores can be completely replenished by the new plant material in one growing season, spring burns decrease shrub biomass but do not decrease stem frequency, even after numerous years of repeated burns. Prescribed burns conducted during early summer (mid June-July) cause the least reduction in western snowberry shoot frequency and shrub biomass production. These early summer burns coincide with western snowberry's major carbohydrate replenishment period, which occurs from full leaf expansion stage through most of the flowering stage, from early June to mid July.

Prescribed burns conducted during early spring (mid-late April), spring (May-mid June), and mid summer (early-mid August) result in decreased western snowberry shoot frequency and shrub biomass production. These burns coincide with the first two carbohydrate drawdown periods of western snowberry. The first carbohydrate drawdown period occurs during early spring, from mid April to early June (9 June), when the plants are in rapid growth to full leaf expansion stages. The second carbohydrate drawdown period occurs during the major portion of fruit fill stage, from mid July to mid August. Burns conducted during carbohydrate drawdown periods have greater success at western snowberry reduction than burns conducted during carbohydrate replenishment periods.

Kentucky bluegrass is increased by earlier burns and decreased by later spring burns. Weedy forbs are increased greatly by spring burns. However, four repeated burns conducted every-otheryear significantly reduce the undesirable plants of introduced grasses, early succession and weedy forbs, and shrubs from mixed grass prairie habitat. Native grasses, sedges, and perennial forbs are not reduced by repeated every-other-year burning and benefit from the reduction in competition for sunlight from the taller shrubs. Repeated every-other-year prescribed burning did not increase native grass basal cover (density), prescribed burning did not increase endomycorrhizal fungal infection of prairie plant roots, and prescribed burning did not increase the quantity of soil organic nitrogen converted into inorganic nitrogen. Stimulation of vegetative reproduction in grasses, stimulation of rhizosphere organism biomass and activity levels, and stimulation of biogeochemical cycling are the key physiological and ecological processes that grassland managers must activate in order to improve native plant health and vigor and to restore degraded ecosystems. Grassland fires do not improve mixed grass prairie ecosystems biologically or ecologically.

Historically, fire had been an environmental factor on mixed grass prairie, with an estimated fire return interval of 5 to 10 years on the moist regions and around 25 years on the dry regions (Wright and Bailey 1982, Bragg 1995). Most lightning-set fires occurred in July and August, and a large portion of the Indian-set fires occurred between July and early November (Higgins 1986). The Northern Plains mixed grass prairie has probably had considerably more late-season fires, occurring after mid July, than spring or early summer fires.

Spring burns during late April or May are severely detrimental to native cool-season grasses because of the removal of the valuable growth of the fall tillers and overwintering secondary tillers. June and early July burns are usually detrimental to native grass plants and hurt western snowberry plants only a little. Prescribed fire during August causes the least damage to native cool- and warm-season grasses and perennial forbs. An August fire removes all or most of the top growth of western snowberry and results in fewer sucker shoots the following year than a spring burn. August burns can be nearly nondetrimental to desirable plants when the soil is not dry, and August burns can cause considerable damage to the undesirable woody plants. Late April and May prescribed burns are less likely to escape control measures compared to August burns; however, the growth pattern and biological requirements of herbaceous vegetation in the mixed grass prairie match the August burns more closely (figure 1).

#### Conclusions

Grassland ecosystems in the mixed grass prairie degrade when managed with long-term idle (no defoliation) treatments and low to moderately stocked deferred grazing treatments. Ecosystem degradation does not occur at a uniform (linear) rate across time. The rate of decline begins slowly and accelerates progressively. The change in plant composition to greater abundance of nonrhizosphere species is basically a symptom, and the degree of plant species change lags behind the degree of ecosystem degradation.

Treatments that are designed to remove undesirable plant species do not restore ecosystem functions. Treatments that stimulate rhizosphere organism populations and enhance ecosystem biogeochemical cycles restore degraded grassland ecosystem processes; this restoration is followed by the improvement in plant composition to a greater abundance of rhizosphere species.

Prescribed burning can be used to kill western snowberry aerial stems to ground level and reduce shrub stem frequency, but fire alone will not eliminate western snowberry. Four repeated everyother-year burns can reduce introduced grasses, early succession and weedy forbs, and undesirable shrubs and woody plants.

Prescribed burning, however, cannot restore degraded grassland ecosystems because fire does not stimulate vegetative reproduction by tillering, a process that results in increased grass basal cover; fire does not stimulate endomycorrhizal fungal colonization of grass roots; fire does not stimulate rhizosphere organism biomass and activity levels; and fire does not stimulate conversion of soil organic nitrogen into inorganic nitrogen.

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	Apr	May	Jun	Jul	Aug	Sep	Oct	Growing Season	Annual Total
Long-term mean 1936-1989	1.28	2.02	3.32	2.36	1.94	1.87	1.01	13.80	16.50
1978	0.58	5.13	2.76	1.45	0.84	2.55	0.64	13.95	14.96
% of LTM	45.31	253.96	83.13	61.44	43.30	136.36	63.37	101.09	90.67
1979	1.59	1.75	0.61	1.32	0.04	0.84	0.50	6.65	8.72
% of LTM	124.22	86.63	18.37	55.93	2.06	44.92	49.50	48.19	52.85
1980	1.81	0.08	1.41	1.63	3.12	2.22	2.08	12.35	15.59
% of LTM	141.41	3.96	42.47	69.07	160.82	118.72	205.94	89.49	94.48
1981	0.41	0.50	4.06	2.77	2.71	1.31	0.65	12.41	15.34
% of LTM	32.03	24.75	122.29	117.37	139.69	70.05	64.36	89.93	92.97
1982	0.38	2.23	1.82	3.12	1.22	2.06	4.64	15.47	19.56
% of LTM	29.69	110.40	54.82	132.20	62.89	110.16	459.41	112.10	118.55
1983	0.19	1.30	1.44	2.42	1.37	1.04	0.82	8.58	12.19
% of LTM	14.84	64.36	43.37	102.54	70.62	55.61	81.19	62.17	73.88
1984	5.74	0.36	1.70	1.29	0.98	2.54	4.10	16.71	19.48
% of LTM	448.44	17.82	51.20	54.66	50.52	135.83	405.94	121.09	118.06
1985	0.69	4.77	1.81	0.99	3.98	2.07	1.60	15.91	18.40
% of LTM	53.91	236.14	54.52	41.95	205.15	110.70	158.42	115.29	111.52
1986	1.71	2.59	1.77	5.85	0.67	1.95	0.76	15.30	18.04
% of LTM	133.59	128.22	53.31	247.88	34.54	104.28	75.25	110.87	109.33
1987	0.10	1.80	0.89	3.83	0.73	0.66	0.30	8.31	12.16
% of LTM	7.81	89.11	26.81	162.29	37.63	35.29	29.70	60.22	73.70
1988	0.05	2.36	2.73	1.67	0.30	2.31	0.19	9.61	13.19
% of LTM	3.91	116.83	82.23	70.76	15.46	123.53	18.81	69.64	79.94
1989	0.70	1.30	3.31	2.66	0.80	1.59	0.48	10.84	16.37
% of LTM	54.69	64.36	99.70	112.71	41.24	85.03	47.52	78.55	99.21
1990	1.26	2.39	2.41	4.24	2.28	0.57	0.03	14.44	15.69
% of LTM	100.78	118.32	72.59	179.66	117.53	30.48	2.97	104.64	95.09
1978-1990	1.17	2.04	2.06	2.56	1.46	1.67	1.29	12.35	15.36
% of LTM	91.41	101.14	61.91	108.34	75 50	89 30	127.88	89 48	93 10

 Table 1. Precipitation in inches for growing-season months and the annual total precipitation for 1978-1990, Lostwood Wildlife Refuge, North Dakota.

	Depth	No Burns	One Burn	Two Burns	Three Burns	Four burns
	inches	5 reps	3 reps	3 reps	4 reps	2 reps
Summit Slope						
July	0-24	3.00a	3.47a	3.11a	2.37a	2.53
August	0-24	2.31b	2.79b	2.17b	2.30b	2.35b
Foot Slope						
July	0-24	3.47c	3.47c	3.24c	2.88c	4.42c
August	0-24	2.58d	3.02d	2.45d	2.69d	3.35d

 Table 2. Inches of soil water, 0-24 inches in depth, on summit and foot slopes with deep loam soils on the number of repeated every-other-year burn treatments.

Means in the same row and followed by the same letter are not significantly different (p<0.05).

 Table 3. Inches of soil water, 0-24 inches in depth, on summit and foot slopes with deep loam soils on the seasonal period of every-other-year burn treatments.

	Depth in inches	No Burns 5 reps	Early Spring (mid-late Apr) 1 rep	Spring (May-mid Jun) 1 rep	Early Summer (mid Jun-Jul) 6 reps	Mid Summer (early-mid Aug) 4 reps
Summit Slope						
July	0-24	3.00a	-	-	3.03a	2.94a
August	0-24	2.31b	2.30b	-	2.51b	2.56b
Foot Slope						
July	0-24	3.47c	-	-	3.36c	3.69c
August	0-24	2.58d	-	-	2.82d	3.01d

Data from Manske 1992

		No Burns	One Burn	Two Burns	Three Burns	Four Burns
Plant Biotypes		6 reps	4 reps	4 reps	4 reps	3 reps
Grass Biomass	(lbs/ac)	411.61a	310.12a	762.75a	512.87a	861.51a
% change	(%)		-24.7	85.3	24.6	109.3
Sedge Biomass	(lbs/ac)	224.59b	362.93b	74.34b	238.58b	145.81b
% change	(%)		61.6	-66.9	6.2	-35.1
Forb Biomass % change	(lbs/ac) (%)	255.33c	454.35c 78.0	445.14c 74.3	587.41c 130.1	266.49c 4.4
Shrub Biomass % change	(lbs/ac) (%)	806.83d	136.00d -83.1	237.09d -70.6	52.00d -93.6	39.57d -95.1
Total Live Biomass	(lbs/ac)	1698.36e	1263.39e	1519.19e	1390.87e	1313.38e
% change	(%)		-25.6	-10.6	-18.1	-22.7

 Table 4. Live biomass production of plant biotypes on the number of repeated every-other-year burn treatments and percent change from nonburned control.

		No Burns	One Burn	Two Burns	Three Burns	Four Burns
Plant Biotypes		6 reps	4 reps	4 reps	4 reps	3 reps
Native Grass Shoot frequency	(%)	107.0a	194.3b	183.3ab	198.8b	208.3b
% change	(%)		81.6	71.3	85.8	94.7
Sedge Shoot frequency	(%)	56.7c	95.5d	97.0d	77.8cd	89.0cd
% change	(%)		68.4	71.1	37.2	57.0
Introduced Grass Shoot frequency	(%)	86.7e	46.3ef	31.8ef	53.5ef	30.3f
% change	(%)		-46.6	-63.3	-38.3	-65.1
Perennial Forbs Shoot frequency	(%)	120.5g	167.8h	125.5gh	137.5gh	125.7gh
% change	(%)		39.3	4.1	14.1	4.3
Weedy Forbs Shoot frequency	(%)	85.5i	92.5i	80.3ij	78.0ij	42.0j
% change	(%)		8.2	-6.1	-8.8	-50.9
Shrubs Shoot frequency	(%)	111.7k	71.0kl	58.5kl	62.0kl	46.71
% change	(%)		-36.4	-47.6	-44.5	-58.2

 Table 5. Shoot frequency of plant biotypes on the number of repeated every-other-year burn treatments and percent change from nonburned control.

Plant Biotypes		No Burns 6 reps	Early Spring (mid-late Apr) 1 rep	Spring (May-mid Jun) 3 reps	Early Summer (mid Jun-Jul) 7 reps	Mid Summer (early-mid Aug) 4 reps
Grass						
Biomass	(lbs/ac)	411.61a	571.59a	748.93a	347.88a	918.49a
% change	(%)		38.9	82.0	-15.5	123.1
Sedge	(11	224 501	2(( 70)	40.001	21 ( 201	102 221
Biomass	(lbs/ac)	224.596	366./96	48.886	316.296	103.336
% change	(%)		63.3	-78.2	40.8	-54.0
Forb	(lbs/ac)	255 330	771.40c	587.21c	451 17c	263.07c
Diomass	(105/ac)	255.550	//1.400	567.210	431.170	203.970
% change	(%)		202.1	130.0	76.7	3.4
<b>CI</b> 1						
Shrub Biomass	(lbs/ac)	806.83d	0.0d	0.0d	226.43d	58.52d
0/ shares	(0/)		100.0	100.0	71.0	02.8
76 change	(70)		-100.0	-100.0	-/1.9	-92.8
Total Live						
Biomass	(lbs/ac)	1698.36e	1709.78e	1385.02e	1341.77e	1344.18e
% change	(%)		0.7	-18.5	-21.0	-20.9

 Table 6. Live biomass production of plant biotypes on the seasonal period of every-other-year burn treatments and percent change from nonburned control.

Data from Manske 1992

Plant Biotypes		No Burns 6 reps	Early Spring (mid-late Apr) 1 rep	Spring (May-mid Jun) 3 reps	Early Summer (mid Jun-Jul) 7 reps	Mid Summer (early-mid Aug) 4 reps
Native Grass Shoot frequency	(%)	107.0a	189.0b	219.7b	182.9b	200.5b
% change	(%)		76.6	105.3	70.9	87.4
Sedge Shoot frequency % change	(%) (%)	56.7c	39.0c -31.2	97.3d 71.6	93.4cd 64.7	90.8cd 60.1
Introduced Grass Shoot frequency % change	(%) (%)	86.7e	73.0e	23.7f -72 7	43.4ef	42.3ef
Perennial Forbs Shoot frequency % change	(%) (%)	120.5g	157.0g 30.3	116.7g -3.2	154.6g 28.3	127.8g 6.1
Weedy Forbs Shoot frequency	(%) (%)	85.5h	129.0i	43.3h	79.3h	78.8h
Shrubs Shoot frequency	(%) (%)	111.7j	50.9 15.01	-49.4 22.01	-7.5 81.7jk	-7.8 63.3k

Table 7.	Shoot frequency of plant biotypes on the seasonal period of every-other-year burn treatments and
	percent change from nonburned control.

Plant Biotypes		No Burns 6 reps	Early Spring (mid-late Apr) 1 rep	Spring (May-mid Jun) 3 reps	Early Summer (mid Jun-Jul) 7 reps	Mid Summer (early-mid Aug) 4 reps
Native Grass						
Cool Season Grass Shoot frequency	(%)	89.2a	177.0b	166.0b	168.9b	169.0b
% change	(%)		98.4	86.1	89.3	89.5
Warm Season Grass Shoot frequency	(%)	17.8c	12.0c	53.7d	14.0c	31.5cd
% change	(%)		-32.7	201.0	-21.5	76.7
Western wheatgrass Shoot frequency	(%)	15.5e	42.0c	13.7e	16.4e	19.5e
% change	(%)		170.1	-11.8	6.0	25.8
Blue grama Shoot frequency % change	(%) (%)	8.0f	4.0f -50.0	39.3g 391.6	7.4f -7.1	14.8fg 84.4
Introduced Grass						
Smooth bromegrass Shoot frequency	(%)	17.5h	0.0h	2.3h	0.3h	2.3h
% change	(%)		-100.0	-86.7	-98.3	-87.1
Quackgrass Shoot frequency % change	(%) (%)	7.8i	1.0i -87.2	0.0i -100.0	0.4i -94.5	2.8i -64.7
Kentucky bluegrass Shoot frequency % change	(%) (%)	59.5j	72.0j 21.0	21.3k -64.2	42.0j -29.4	37.3jk -37.3

 Table 8. Shoot frequency of grasses on the seasonal period of every-other-year burn treatments and percent change from nonburned control.

Data from Manske 1992

Plant Biotypes		No Burns 6 reps	Early Spring (mid-late Apr) 1 rep	Spring (May-mid Jun) 3 reps	Early Summer (mid Jun-Jul) 7 reps	Mid Summer (early-mid Aug) 4 reps
Western snowberry Shoot frequency	(%)	58.3a	5.0b	10.3b	33.4ab	21.5b
% change	(%)		-91.4	-82.3	-42.7	-63.1
Western rose Shoot frequency % change	(%) (%)	35.8c	8.0d -77.7	4.3d -87.9	41.7c 16.4	41.3c 15.1
Silverberry Shoot frequency % change	(%) (%)	17.3e	2.0f -88.5	7.3e -57.7	6.6e -62.1	0.5f -97.1

 Table 9. Shoot frequency of shrubs on the seasonal period of every-other-year burn treatments and percent change from nonburned control.

Data from Manske 1992

		No Burns	One Burn	Two Burns	Three Burns	Four Burns
Plant Biotypes		6 reps	4 reps	4 reps	4 reps	3 reps
Western snowberry Fungi infection	(%)	93.8a	84.7a	84.3a	85.2a	85.9a
Smooth bromegrass Fungi infection	(%)	32.3b	55.0b	50.0b	31.4b	40.1b
Western wheatgrass Fungi infection	(%)	66.0c	67.0c	61.3c	76.8c	63.8c
Blue grama Fungi infection	(%)	78.8d	77.1d	84.9d	79.9d	73.5d
Mineral Nitrogen NH <sub>4</sub> -NO <sub>3</sub>	(ppm)	9.56e	9.65e	9.41e	5.54e	8.36e

Table 10. Mycorrhizal fungal infection of plant roots and soil mineral nitrogen (NH<sub>4</sub>-NO<sub>3</sub>) on the number of repeated every-other-year burn treatments.

Means in the same row and followed by the same letter are not significantly different (P<0.05).

Table 11.	Mycorrhizal fung	al infection of pl	ant roots and so	il mineral nitroger	$n (NH_4-NO_3) c$	on the seasonal	period of
	every-other-year l	burn treatments.					

Plant Biotypes		No Burns 6 reps	Early Spring (mid-late Apr) 1 rep	Spring (May-mid Jun) 3 reps	Early Summer (mid Jun-Jul) 7 reps	Mid Summer (early-mid Aug) 4 reps
Western snowberry Fungi infection	(%)	93.8a	92.3a	85.5a	82.7a	86.6a
Smooth bromegrass Fungi infection	(%)	32.3b	33.7b	40.0b	37.2b	65.7b
Western wheatgrass Fungi infection	(%)	66.0c	74.7c	48.0c	73.7c	69.5c
Blue grama Fungi infection	(%)	78.8d	70.7d	79.6d	82.0d	76.2d
Mineral Nitrogen NH <sub>4</sub> -NO <sub>3</sub>	(ppm)	9.56e	3.64	8.47e	9.42e	7.09e

Data from Manske 1992



Fig. 1. Western snowberry colony before (left) and after (right) four every-other-year prescribed burns during mid summer (August). Photographs were taken by Karen Smith.

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#### Grazing and Burning Treatment Effects on Soil Mineral Nitrogen and Rhizosphere Volume

Report DREC 11-1066c

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Insufficient nitrogen availability limits productivity more often than water in temperate grasslands (Tilman 1990). Enhancement of the nitrogen cycle in grassland ecosystems that causes an increase in available inorganic (mineral) nitrogen results in greater productivity of herbage, livestock, and wildlife.

Soil organic matter of mixed grass prairie ecosystems generally contains about three to eight tons of organic nitrogen per acre. Organic nitrogen is a form of nitrogen not directly usable by grass plants. Organic nitrogen must be converted into mineral nitrogen in order to be usable by plants. The biogeochemical processes of the nitrogen cycle in grassland ecosystems that convert nitrogen into the various forms are a function of the interactions among rhizosphere organisms, grass plants, and large grazing herbivores (Manske 1999).

This project uses soil mineral nitrogen and rhizosphere volume data to evaluate the effects that grazing treatments and burning treatments have on rhizosphere organism activity and on enhancement of biogeochemical processes of the nitrogen cycle in grassland ecosystems and the resulting increase in quantity of available mineral nitrogen converted from soil organic nitrogen.

#### **Study Areas**

This project was conducted at two locations. The NDSU Dickinson Research Extension Center ranch is located in Dunn county in western North Dakota, at 47° 14' north latitude, 102° 50' west longitude. Mean annual temperature is 42.5° F (5.8° C). January is the coldest month, with a mean temperature of 14.5° F (-9.7° C). July and August are the warmest months, with mean temperatures of 69.4° F (20.8° C) and 68.8° F (20.4° C), respectively. Long-term (1982-2006) mean annual precipitation is 16.87 inches (428.59 mm). The growing-season precipitation (April through October) is 14.09 inches (357.86 mm) and is 83.50% of the annual precipitation. June has the greatest monthly precipitation, at 3.27 inches (83.09 mm). The precipitation received during the 3-month period of

May, June, and July (8.11 inches, 205.99 mm) accounts for 48.07% of the annual precipitation (Manske 2007a). Soils are primarily Typic Haploborolls developed on sedimentary deposits. The fine loamy soils have 5 to 6 tons of organic nitrogen per acre. Native vegetation is the Wheatgrass-Needlegrass Type (Barker and Whitman 1988) of the mixed grass prairie.

The USDI Lostwood National Wildlife Refuge is located in Burke and Mountrail counties in northwestern North Dakota between 48° 50' and 48° 30' north latitude and  $102^{\circ}$  40' and  $102^{\circ}$  20' west longitude. The region has cold winters and hot summers typical of continental climates. January is the coldest month, and July and August are the warmest months. Long-term (1936-1989) mean annual precipitation is 16.50 inches (419.10 mm). The growing-season precipitation (April through October) is 13.80 inches (350.52 mm) and is 83.64% of the annual precipitation. June has the greatest monthly precipitation, at 3.32 inches (84.33 mm). The precipitation received during the 3-month period of May, June, and July (7.70 inches, 195.58 mm) accounts for 46.67% of the annual precipitation (Manske 2007b). Soils are primarily Typic Haploborolls and Typic Argiborolls developed on glacial terminal moraine. The fine loamy soils have 4 to 6 tons of organic nitrogen per acre. Native vegetation is the Wheatgrass-Needlegrass Type (Barker and Whitman 1988) of the mixed grass prairie.

#### Procedures

The effects from defoliation treatments on enhancement of the nitrogen cycle and rhizosphere organism biomass and activity were evaluated from differences in soil mineral nitrogen content and in rhizosphere volume. The quantitative soil mineral nitrogen study was conducted on grazing treatments and on repeated prescribed burning treatments. The quantitative rhizosphere volume study was conducted on grazing treatments.

The grazing treatments were conducted at the Dickinson Research Extension Center ranch

between 1983 and 2002 on mixed grass prairie that had a history of management with moderate to heavy seasonlong grazing. The treatments were (1) longterm nongrazed control (NG), (2) 6.0-month seasonlong (6.0-m SL), (3) 4.5-month seasonlong (4.5-m SL), and (4) twice-over rotation (TOR). Each of the grazing treatments had two replications. The long-term nongrazed management treatment had not been grazed, mowed, or burned for more than 30 years before the initiation of these research treatments in 1983. Livestock on the 6.0-month seasonlong management treatment grazed one native range pasture for 6.0 months (183 days) from mid May until mid November. Livestock on the 4.5-month seasonlong management treatment grazed one native range pasture for 4.5 months (135 days) from early June until mid October. Livestock on the twice-over rotation management treatment followed a double rotation sequence through three native range pastures for 4.5 months (135 days) from early June until mid October. Each of the three pastures in the rotation were grazed for about 15 days during the first period, the 45-day interval from 1 June to 15 July. During the second period, after mid July and before mid October, each pasture was grazed for double the number of days it was grazed during the first period.

The repeated every-other-year prescribed burning treatments were conducted by refuge manager Karen Smith at the Lostwood National Wildlife Refuge during a thirteen year period between 1978 and 1990 on degraded mix grass prairie that had greater than 50% of the upland occupied with western snowberry colonies (Smith 1988). The number of repeated every-other-year prescribed burns was (1) no burns control, (2) one burn, (3) two burns, (4) three burns, and (5) four burns. The prescribed burns were conducted during four seasons: (6) early spring, (7) spring, (8) early summer, and (9) mid summer. Annual burns were not possible because of insufficient production of plant biomass for fuel (Smith 1985). The refuge land had not been burned for more than 100 years before the initiation of these prescribed burn treatments in 1978. However, between 1940 and 1975, about 26% of the refuge was annually grazed with a deferred-type seasonlong management for 4.5 to 5.0 months at low to moderate stocking rates, primarily during July through November (Smith 1988). Some areas were grazed only one time and other areas were grazed as many as 22 times over the 35-year period (Smith 1988). About 15% of the refuge was never burned, grazed, or mowed (Smith 1997). The no burns control treatment had 6 replications with an average size of 436.8 acres (176.90 ha) and had no grazing, mowing, or burning during the thirteen year study period. The

every-other-year prescribed burning treatments had an average size of 530.5 acres (214.85 ha) and had no grazing or mowing during the thirteen year study period. The one burn, two burns, and three burns treatments had 4 replications each, and the four burns treatment had 3 replications. The early spring burns had 1 replication, the spring burns had 3 replications, the early summer burns had 7 replications, and the mid summer burns had 4 replications.

The quantitative soil mineral nitrogen study was conducted on the nongrazed, 4.5-month seasonlong, and twice-over rotation grazing treatments; on the no burns, one burn, two burns, three burns, and four burns every-other-year prescribed burning treatments; and on the early spring, spring, early summer, and mid summer everyother-year prescribed burning treatments. Field samples of grazing and burning treatment soils were collected from nearly level loam soils. Aboveground vegetation was clipped from soil sample areas and discarded. Each soil field sample consisted of five soil cores that were air dried, ground, and thoroughly mixed. Twenty percent of each field sample was retained for quantitative laboratory analysis.

Field samples were collected during mid June 1989 at the start of the seventh year of the grazing treatment study. Each soil core from the grazing treatments was collected with a bucket auger and was 2 inches (5.08 cm) in diameter and 6 inches (15.24 cm) in depth. Two field samples were collected in each of the two replications of each nongrazed control and grazing treatment, resulting in four replicated field samples per grazing treatment.

Field samples were collected during July and August 1990 after thirteen years of every-other-year prescribed burning treatments. Each soil core from the burn treatments was collected with a soil probe and was 1 inch (2.54 cm) in diameter and 6 inches (15.24 cm) in depth. One field sample was collected during each time period for each replication of the no burns and the seasonal period and number of repeated every-other-year prescribed burning treatments. A mean of the July and August sample periods was determined for each treatment replication.

In the laboratory, subsamples of soil from the field samples were analyzed for total incubated mineralizable nitrogen (N) using procedures outlined by Keeney (1982). Inorganic forms of nitrogen were extracted from the soil subsamples by adding a reagent; alkaline phosphate-borate buffer to the grazing treatment soils, and 2 M KCl to the burning treatment soils. The mixtures of reagent and soil were shaken for one hour. The extract was quantified into parts per million (ppm) of mineral nitrogen, ammonia (NH<sub>3</sub>), ammonium (NH<sub>4</sub>), and nitrate NO<sub>3</sub>), with steam distillation (Keeney and Nelson 1982). These quantified values of mineral nitrogen were converted from parts per million to pounds per acrefoot. Soil nitrite (NO<sub>2</sub>) was not quantified because it is seldom present in detectable amounts.

The rhizosphere volume study was conducted on the nongrazed (NG), 6.0-month seasonlong (6.0-m SL), 4.5-month seasonlong (4.5-m SL), and twice-over rotation (TOR) grazing treatments in 2002 during the twentieth year of the grazing treatment study. Two replications of soil cores containing western wheatgrass roots and rhizospheres were collected monthly during June, July, August, and September from nearly level loam soils in each grazing treatment. Plastic PVC pipe 3 inches (7.62 cm) in diameter and 4 inches (10.16 cm) long was forced into sample site soil. Intact soilplant-rhizosphere cores and pipe were excavated and transported to the laboratory.

In the laboratory, the soil matrix of collected soil cores was carefully removed from between the rhizospheres surrounding the roots of western wheatgrass plants. The roots and rhizospheres of other plant species were separated from the soil cores and discarded. The western wheatgrass 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 with a vernier caliper. During the process of extraction from matrix soil, portions of some rhizospheres were damaged and small segments were detached from the root surface. The length measurements of damaged rhizospheres were the length of the root, including the regions of detached rhizosphere segments. The length and diameter measurements were used to determine the volume of the rhizosphere around each root. The sum of the individual roots' rhizosphere volume was the total rhizosphere volume per replicated soil core. The total rhizosphere volume reported as cubic centimeters per cubic meter of soil (Gorder, Manske, and Stroh 2004) was converted to cubic feet per acre-foot. Differences among treatments in the means of mineral nitrogen content and rhizosphere volume were analyzed by a standard paired-plot t-test (Mosteller and Rourke 1973).

#### Results

Precipitation during the 1989 growing season at the Dickinson Research Extension

Center ranch was low at 10.60 inches (75.24% of LTM). During 1989, plants experienced water stress during July, August, and September. The previous growing season of 1988 had drought conditions receiving only 5.30 inches of precipitation (37.62% of LTM). During 1988, plants experienced water stress during April, June, July, August, September, and October. The drought of 1988 started in August 1987 and near-drought conditions lasted through the growing season of 1992 (Manske 2007a).

Precipitation during the 1990 growing season at the Lostwood Wildlife Refuge was near normal at 14.44 inches (104.64% of LTM). Below normal precipitation occurred during June, September, and October. The previous growing season of 1989 had low precipitation at 10.84 inches (78.55% of LTM). Below normal precipitation occurred during April, May, August, and October. The growing season of 1988 had drought conditions receiving 9.61 inches (69.64% of LTM). Below normal precipitation occurred during April, July, August, and October (Manske 2007b).

Precipitation during the 2002 growing season at the Dickinson Research Extension Center ranch was greater than normal at 18.85 inches (133.79% of LTM). June, July, and August were wet months with precipitation greater than 125% of LTM. September and October had water deficiencies receiving precipitation at less than 75% of LTM and plants experienced water stress during September (Manske 2007a).

Mineral nitrogen in pounds per acre-foot (lbs/ac-ft) is the quantity of mineral nitrogen available in the soil and usable by grassland plants for growth of aboveground herbage and belowground root biomass. Differences in the pounds of available mineral nitrogen on the grazing treatments and burning treatments indicates the differences in the effects the defoliation treatments have on the rhizosphere organism activity and on the enhancement of the nitrogen cycle and the quantity of mineral nitrogen converted from soil organic nitrogen.

The mineral nitrogen available on the one, two, three, and four repeated every-other-year burning treatments and on the early spring, spring, early summer, and mid summer seasonal every-otheryear burning treatments was low and was not significantly different (P<0.05) from that on the no burns control treatment (table 1). The number of repeated every-other-year burning treatments and the seasonal period of prescribed burning treatments did
not affect the quantity of available mineral nitrogen in the soil. The mineral nitrogen available on all of the every-other-year burning treatments and no burns control treatment was significantly less (P<0.05) than the mineral nitrogen available on all of the grazing treatments and nongrazed control treatment (table 1, figure 1). The mineral nitrogen available on the no burns treatment and every-other-year prescribed burning treatments was less than 30% of the mineral nitrogen available on the nongrazed treatment. The antagonistic effects on the rhizosphere organism activity from the previous management with deferred grazing practices caused the available mineral nitrogen to decrease 70.6% on the no burns treatment and to decrease 74.5% on the prescribed repeated burning treatments.

The mineral nitrogen available on the 4.5-m and 6.0-m seasonlong grazing treatments and on the nongrazed control treatment were significantly different (P<0.05) (table 1, figure 1). After six grazing seasons, the seasonlong grazing treatments did not beneficially affect the rhizosphere organisms and the biogeochemical nitrogen cycle processes and did not increase the quantity of available mineral nitrogen. The seasonlong grazing treatments were antagonistic to rhizosphere organism activity causing a reduction in the quantity of soil organic nitrogen converted into mineral nitrogen.

The mineral nitrogen available on the twiceover rotation treatment was more than 100 lbs/ac and was significantly greater (P<0.05) than that on the 4.5-m and 6.0-m seasonlong grazing treatments and on the nongrazed control treatment (table 1, figure 1). The twice-over rotation treatment beneficially affected the rhizosphere organisms and the available mineral nitrogen was increased 67.7% greater than on the nongrazed treatment and 131.9% and 188.7% greater than on the seasonlong grazing treatments.

The twice-over rotation treatment consisted of three native range pastures grazed for about 15 days each during the first period from 1 June to 15 July when grass tillers are between the three and a half new leaf stage and the flowering stage. The available mineral nitrogen was high on each of the three rotation pastures and was not significantly different (P<0.05) among the first, second, and third pasture grazed in the rotation sequence (table 2). Effects from first period grazing treatments on the first, second, and third pastures of the twice-over rotation sequence stimulated the rhizosphere organism activity that increased the mineral nitrogen 87.7%, 54.6%, and 60.8% greater than that on the nongrazed treatment, respectively. Rhizosphere volume in cubic feet per acrefoot (ft<sup>3</sup>/ac-ft) is the quantity of space occupied by active rhizosphere organisms on the grazing treatments. Differences in rhizosphere volume indicates the differences in the effects the grazing treatments have on the rhizosphere organism biomass and activity and subsequently on the proportional differences in the quantity of mineral nitrogen converted from soil organic nitrogen.

Rhizosphere volume on the 6.0-m seasonlong grazing treatment was significantly lower (P < 0.05) than that on the nongrazed control treatment (table 1, figure 2) and was lower than, but not significantly different (P<0.05) from, that on the 4.5m seasonlong grazing treatment (table 1, figure 2). The rhizosphere volume was greatly reduced in size as a result of the antagonistic effects from the 6.0-m seasonlong treatment on rhizosphere organism activity. The rhizosphere volume on the 6.0-m seasonlong treatment was 51.2% smaller than on the nongrazed treatment and 26.4% smaller than on the 4.5-m seasonlong treatment. Rhizosphere volume on the 4.5-m seasonlong grazing treatment was lower than, but not significantly different (P<0.05) from, that on the nongrazed control treatment (table 1, figure 2). After 20 grazing seasons, the 4.5-m seasonlong treatment was slightly antagonistic to the activity of the rhizosphere organisms. The rhizosphere volume was 33.7% smaller on the 4.5-m seasonlong treatment than on the nongrazed treatment. Rhizosphere volume on the twice-over rotation treatment was significantly greater (P<0.05) than the rhizosphere volume on the 6.0-m seasonlong and the 4.5-m seasonlong grazing treatments and on the nongrazed control treatment (table 1, figure 2). The twice-over rotation treatment greatly stimulated the activity of the rhizosphere organisms. The rhizosphere volume on the twice-over rotation treatment was 356.4%, 235.8%, and 122.7% greater than on the 6.0-m seasonlong, 4.5-m seasonlong, and nongrazed treatments, respectively.

Rhizosphere volume changes as a result of increases or decreases in rhizosphere organism activity. The effects on rhizosphere volume during the growing season from the grazing treatments are shown in figures 3 and 4. Rhizosphere volume on the nongrazed control treatment increased slightly during the early summer in July and remained at about the same volume for the duration of the season. The volume at the end of the season on the nongrazed treatment was about 40% greater than the volume at the beginning. Rhizosphere volume on the 6.0-m seasonlong grazing treatment declined gradually during the growing season and the volume at the end

of the season was less than half the volume of the beginning. The 6.0-m seasonlong treatment was antagonistic to the activity of the rhizosphere organisms during the entire grazing period. The rhizosphere volume on the 6.0-m seasonlong treatment was significantly reduced (P<0.05) and was 73% less than that on the nongrazed treatment. Rhizosphere volume on the 4.5-m seasonlong treatment remained about the same during the growing season except for a substantial decrease during July. The volume at the end of the season was the same as the volume at the beginning. The antagonistic effects on rhizosphere organism activity from the 4.5-m seasonlong treatment caused a 35.2% decrease in rhizosphere volume, but not significant (P<0.05) from that on the nongrazed treatment. Rhizosphere volume on the twice-over rotation treatment increased slowly during the early growing season and then, following the first grazing period that occurred when grass tillers were between the three and a half new leaf stage and the flower stage from early June to mid July, the volume increased rapidly and more than doubled in size. At the end of the growing season, the volume on the twice-over rotation treatment was 105% greater than the volume at the beginning. The twice-over rotation treatment greatly stimulated the activity of the rhizosphere organisms. The rhizosphere volume on the twiceover rotation treatment significantly increased (P<0.05) and was 900.6%, 265.3%, and 170.1% greater than the rhizosphere volumes on the 6.0-m seasonlong, 4.5-m seasonlong, and nongrazed treatments, respectively.

The first grazing period on the third pasture in the twice-over rotation sequence was for 15 days during early July. The rhizosphere volume increased 20.3% during the two weeks following the first grazing period and increased 85.7% during the next five weeks (figure 5). The rhizosphere volume increased 123.4% in size because of the enhanced rhizosphere organism activity that resulted from the beneficial effects of partial defoliation during the first grazing period on the twice-over rotation treatment.

#### Discussion

In grassland ecosystems, the conversion of plant usable mineral nitrogen from soil organic nitrogen requires active rhizosphere organisms. Rhizosphere organisms require short chain carbon exudates from roots of grass plants. Rhizosphere organisms trade nitrogen to grass plants for carbon, and grass plants trade carbon to soil microorganisms for nitrogen. This interdependent symbiotic relationship between grass plants and rhizosphere organisms is controlled by partial defoliation of aboveground plant parts. The effects of defoliation can be beneficial or antagonistic depending on the degree of foliage removal and phenological growth stage of the grass tillers. Knowledge of grass developmental morphology and physiological processes that help grass tillers withstand and recover from defoliation is necessary to comprehend the biological requirements of grasses and the effects from defoliation management practices.

## Grazing Defoliation

Grass plants have developed defoliation resistance mechanisms in response to grazing during the period of coevolution with herbivores (McNaughton 1979, 1983; Coleman et al. 1983; Briske and Richards 1995; Manske 1999) that help grass tillers withstand and recover from partial defoliation by grazing.

Traditional grazing management practices are not beneficial for grassland ecosystems. The deferred grazing, 6.0-m seasonlong, and 4.5-m seasonlong management strategies are antagonistic to rhizosphere organism activity and biogeochemical processes of the nitrogen cycle and these traditional grazing practices do not stimulate beneficial grass growth processes. The rhizosphere organism biomass and activity and the quantity of available mineral nitrogen decrease by small amounts annually along with proportional decreases in grass density and herbage biomass production. After two, three, or more decades of management of grassland ecosystems with traditional grazing practices, the losses in productivity are substantial. The deferred grazing strategy caused a 70.6% decrease in available mineral nitrogen after 35 years of treatment. The 6.0m seasonlong grazing strategy caused a 41.9% decrease in mineral nitrogen and a 51.2% decrease in rhizosphere volume after 20 years of seasonlong treatment. The 4.5-m seasonlong grazing strategy caused a 27.7% decrease in mineral nitrogen after 6 years of treatment and caused a 33.7% decrease in rhizosphere volume after 20 years of treatment.

The twice-over rotation grazing management strategy is beneficial for grassland ecosystems. The advantageous effects from partial defoliation managed with a twice-over rotation grazing strategy caused a 67.7% increase in available mineral nitrogen after six years of treatment and caused a 122.7% increase in rhizosphere volume after 20 years of treatment.

## **Burning Defoliation**

Prescribed burning of grasslands severely removes all or nearly all of the aboveground plant material. Complete defoliation by fire of grass tillers does not activate the defoliation resistance mechanisms that help grass tillers withstand and recover from partial defoliation. Replacement of fire removed plant material must develop from crown buds and stored root carbohydrate reserves. Growth of roots and shoots that depends on stored carbohydrates occurs at greatly reduced rates (Coyne et al. 1995) compared to replacement growth after partial defoliation by grazing that triggers the compensatory physiological processes, stimulates vegetative reproduction by tillering, and stimulates rhizosphere organism activity with the resulting increase in available mineral nitrogen (Manske 1999).

The quantity of mineral nitrogen on the prescribed burning treatments and no burns treatment was not different, and was less than 30% of the quantity of mineral nitrogen on the nongrazed treatment, indicating that the previous management practices on the burning and no burns treatments were antagonistic to the rhizosphere organisms and the nitrogen cycle biogeochemical processes, and that none of the prescribed burning treatments stimulated activity of the rhizosphere organisms. The nongrazed treatment had more than 37 years with no defoliation events and had 106 pounds per acre-foot of available mineral nitrogen. The burning and no burns treatments had no grazing defoliation events during the period from 1975 to 1990, however, the entire area, except for about 15% of the land, was managed with periodic deferred-type seasonlong grazing from July through November in combination with full growing season rest for 35 years, between 1940 and 1975. As a result, the burning and no burns treatments had only around 31 pounds per acre-foot of available mineral nitrogen.

The intended purpose for deferment of grazing on a wildlife refuge was the avoidance of disturbance from grazing livestock of ground nesting birds from nest initiation through egg hatch. The intended biological purpose of deferred grazing was to increase grass density by promoting seedling development from increased seed stalk quantities and to use trampling by livestock to scatter and plant the resulting seeds. However, grassland ecosystem processes do not function in accordance with these proposed deferred grazing hypotheses.

The combination of periodic deferred-type seasonlong grazing and full growing season rest used

to manage the Lostwood Wildlife Refuge between 1940 and 1975 caused the mixed grass prairie ecosystem to degrade and permitted the increase of western snowberry. This deterioration did not occur at a uniform rate. In the mid to late 1930's, only about 5% of the land area was occupied with western snowberry. The shrub composition in the plant community did not change much during the next 20 years. A substantial increase in shrub cover occurred between 1953 and 1969, and the western snowberry colonies expanded rapidly and infested extensive areas of degraded grassland between 1969 and 1975; as a result, over 50% of the upland was transformed into a shrubland of western snowberry and associated Kentucky bluegrass by 1979 (Smith 1988).

A prescribed repeated every-other-year burning regime designed with the intent to reduce the invading western snowberry and exotic grasses and to renovate the mixed grass prairie plant community (Smith 1985) was conducted for 13 years between 1978 and 1990. Nevertheless, the nitrogen cycle biogeochemical processes of the degraded grassland were not restored and the available mineral nitrogen was not increased by any of the prescibed burning treatments and, consequently, the quantity of aboveground biomass production on the burning treatments remained low and was not different from that produced on the no burns treatment (Manske 2007b). However, the percent composition of the aboveground biomass did change as a result of the prescribed burn treatments. The composition of introduced grasses, early succession and weedy forbs, and shrubs decreased with the increasing number of repeated prescribed burning treatments. Native grasses, sedges, and perennial forbs benefitted from the reduction in competition for sunlight from the decreased canopy cover of the taller shrubs (Manske 2007b).

Repeated prescribed burning does reduce undesirable opportunistic grasses, forbs, and shrubs. However, prescribed burning does not stimulate vegetative reproduction by tillering, prescribed burning does not stimulate endomycorrhizal fungal colonization of grass roots, prescribed burning does not stimulate rhizosphere organism biomass and activity, and prescribed burning does not stimulate conversion of soil organic nitrogen into mineral nitrogen (Manske 2007b).

Degraded grassland ecosystems are not restored by prescribed burning practices because restoration of native plant composition and biomass production takes place after the improvement of rhizosphere organism activity and the increase of available mineral nitrogen. Prescribed burning does not enhance the nitrogen cycle biogeochemical processes in grassland ecosystems.

#### Conclusion

Productivity on grassland ecosystems depends on the level of rhizosphere organism activity, on the quantity of available mineral nitrogen converted from soil organic nitrogen, and on the effectiveness of beneficial physiological processes within grass tillers.

Traditional grazing practices of deferred grazing, 6.0-m seasonlong, and 4.5-m seasonlong management are antagonistic to grass tiller biological requirements, and to rhizosphere organism biomass and activity. Over numerous growing seasons, the antagonistic effects from traditional grazing management result in greatly reduced biogeochemical processes of the nitrogen cycle and substantially decreased quantities of available mineral nitrogen that cause degradation of ecosystem plant communities and reduction of herbage biomass production. Nondefoliation management is antagonistic to grass tiller biological requirements and to rhizosphere organism biomass and activity causing slow degradation of the grassland ecosystem. The degree of ecosystem deterioration caused by the antagonistic effects from long-term nondefoliation treatments is greater than that from long-term traditional grazing management.

Fire has been an environmental factor on grassland ecosystems historically. Repeated prescribed burning reduces the undesirable opportunistic plants of introduced grasses, early succession and weedy forbs, and shrubs that increase in composition on degraded grassland ecosystems. However, prescribed burning that completely defoliates the plant material on grassland ecosystems does not stimulate beneficial physiological processes, does not stimulate rhizosphere organism biomass and activity, and does not stimulate conversion of mineral nitrogen from soil organic nitrogen. Consequently, the poor native plant species composition and the low herbage biomass production on degraded grassland ecosystems is not improved by burning. Repeated prescribed burning treatments do not restore degraded grassland ecosystems.

Partial defoliation controlled with the twiceover rotation grazing management strategy that removes 25% to 33% of the leaf material from grass tillers at phenological growth between the three and a half new leaf stage and the flower (anthesis) stage (early June to mid July); stimulates compensatory physiological processes resulting in greater replacement leaf, shoot, and root growth; stimulates vegetative reproduction by tillering resulting in greater grass tiller density and herbage biomass production; and stimulates rhizosphere organism biomass and activity resulting in quantities of available mineral nitrogen greater than 100 pounds per acre. Restoration of degraded grassland ecosystems and maintenance of healthy functioning grassland ecosystems requires annual partial defoliation by grazing that meets the biological requirements of grass tillers, that enhances rhizosphere organism activity, and that increases the quantity of available mineral nitrogen.

Grassland ecosystems have been customarily managed from the perspective of the "use", e.g. for wildlife habitat or for livestock forage. The designated use receives priority consideration when management decisions are made. These management practices that focus on the use of a grassland ecosystem are antagonistic to the plants and to the rhizosphere organisms and cause reductions in grassland ecosystem productivity.

Management of partial defoliation by grazing that focuses on meeting the biological requirements of the plants and of the rhizosphere organisms is beneficial, and enhances the health and productivity of grassland ecosystems. Placing the decision priorities with the living components of the ecosystem is biologically effective, and results in greater forage for livestock, better habitat for wildlife, and more aesthetic open spaces for recreation and sightseeing as sustainable products from biologically managed grassland ecosystems.

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	Mineral	Rhizosphere
	(inorganic) Nitrogen	Volume
	lbs/acre-foot	ft <sup>3</sup> /acre-foot
Grazing Treatments		
Nongrazed (NG)	106.05 b	101.93 y
6.0-m Seasonlong (6.0-m SL)	61.61 c	49.75 z
4.5-m Seasonlong (4.5-m SL)	76.70 c	67.61 yz
Twice-over Rotation (TOR)	177.84 a	227.06 x
Burning Treatments		
No Burns	31.20 d	
One Burn	31.49 d	
Two Burns	30.71 d	
Three Burns	18.08 d	
Four Burns	27.28 d	
Early Spring Burns	11.88	
Spring Burns	27.64 d	
Early Summer Burns	30.74 d	
Mid Summer Burns	23.14 d	

 Table 1. Mineral nitrogen (lbs/acre-foot) for grazing treatments and repeated every-other-year burning treatments and rhizosphere volume (ft³/acre-foot) for grazing treatments.

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

	Mineral (inorganic) Nitrogen lbs/acre-foot
Twice-over Rotation (TOR)	
First pasture grazed	199.05 a
Second pasture grazed	163.97 a
Third pasture grazed	170.50 a
Three pasture mean	177.84 a

Table 2. Mineral nitrogen (lbs/acre-foot) for the three pastures of the twice-over rotation treatment.

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



Figure 1. Mineral nitrogen in pounds per acre-foot for grazing treatments and every-other-year burn treatments.



Seasonal Mean

Figure 2. Seasonal mean rhizosphere volume in cubic feet per acre-foot for grazing treatments.



Figure 3. Monthly rhizosphere volume in cubic feet per acre-foot during June and July for grazing treatments.



Figure 4. Monthly rhizosphere volume in cubic feet per acre-foot during August and September for grazing treatments.



Figure 5. Rhizosphere volume change on the twice-over rotation treatment third pasture that was grazed for 15 days during early July.

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# Initial Restoration Changes of Degraded Rangeland with the Twice-over Rotation Grazing Strategy

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Biogeochemical processes are the processes that renew nutrient flow activities in ecosystem soils of renewable natural resources. The biogeochemical processes of rangeland natural resources managed with traditional management practices are impeded from functioning at potential levels. Traditional practices that: 1. start grazing too early, before the 3.5 new leaf stage; 2. remove greater than 25% to 33% of green leaf biomass before the flower stage; 3. start grazing too late, after the flower stage; 4. remove greater than 50% of green leaf biomass after peak aboveground herbage biomass produced in late July; 5. continue grazing carryover tillers, after mid October; or 6. "rest" vegetation with nondefoliation during parts or entire growing seasons to increase seed production; are detrimental to the ecosystem biogeochemical processes and are antagonistic to the native plants and soil organisms causing degradation of rangeland renewable natural resources.

Traditional management practices also diminish the compensatory physiological processes, the vegetative reproduction mechanisms, and the rhizosphere organism population (Manske 2011). Decreases in vegetative reproduction by tillering and reductions of compensatory physiological processes prevent grass tillers from replacing a sufficient proportion of the leaf area removed by partial defoliation. Low leaf surface area reduces the quantity of carbon fixed by photosynthesis. Decreases in rhizosphere organism biomass and activity result in decreased biogeochemical processes of the nitrogen cycle causing reductions in the quantity of organic nitrogen converted into mineral nitrogen (Manske 2007). Decreases in the availability of the quantity of fixed carbon and the quantity of mineral nitrogen in an ecosystem degrade grassland plant communities by causing reductions in grass herbage biomass production and native plant density, creating larger and more numerous open spaces between grass plants. These open spaces in the plant communities provide ideal habitat for growth of opportunistic "weedy" plant species that are not dependent on the nitrogen converted by rhizosphere organisms (Manske 2011). Opportunistic grasses and forbs are not highly competitive and do

not increase into plant communities unless openings are created by management caused reductions in native grass densities. Degraded grassland ecosystems have an increasing composition of opportunistic introduced grasses, early succession and weedy forbs, and shrubs (Manske 2007).

This project will describe and evaluate development of biological restoration of degraded rangeland resources through implementation of biological effective grazing management. Quantitative annual changes of aboveground herbage biomass, plant species basal cover, forb density, rhizosphere biomass, and available soil mineral nitrogen will document progress of rangeland ecosystem restoration.

# Study Area

Restoration research was conducted during 2006 to 2011 on 1,988 acres (804.5 hectares) of degraded native rangeland located about 2 miles (3.22 kilometers) east of the town of Richardton, eastern Stark County in western North Dakota, between  $47^{\circ}$  00' and  $46^{\circ}$  50' north latitude and between  $102^{\circ}$  20' and  $102^{\circ}$  10' west longitude.

The untilled rangelands with native plant ecosystems became severely degraded by traditional management practices that were antagonistic to native plant and soil rhizosphere organism biological requirements and to ecosystem biogeochemical processes causing great decreases in the native plant density and herbage biomass. The degraded rangelands subsequently became invaded with undesirable introduced cool season domesticated grasses, primarily kentucky bluegrass with smaller areas of smooth bromegrass and crested wheatgrass, that caused additional problems of shading and increased mulch biomass.

## **Regional Climatic Factors**

The western North Dakota region near Richardton has cold winters and hot summers typical of continental climates. Mean annual temperature is 43.0° F ( $6.1^{\circ}$  C). January is the coldest month, with a mean temperature of 13.5° F (-10.3° C). July and August are the warmest months, with mean temperatures of 70.0° F ( $21.1^{\circ}$  C) and 68.9° F ( $20.5^{\circ}$  C), respectively. Long-term (1971-2000) mean annual precipitation is 17.78 inches (451.61 mm). The precipitation during the perennial plant growing season (April through October) is 14.79 inches (375.67 mm) and is 83.2% of the annual precipitation. June has the greatest monthly precipitation received during the three month period of May, June, and July is 8.15 inches (207.01 mm) and is 45.8% of the annual precipitation.

Water stress develops in perennial plants during water deficiency periods when the amount of rainfall is less than evapotranspiration demand. Water deficiency months were identified from historical temperature and precipitation data by the ombrothermic diagram technique (Emberger et al. 1963). The long-term (1971-2000) ombrothermic diagram shows near water deficiency conditions during August, September, and October, and favorable water relations during April, May, June, and July. Reoccurrence of water deficiency conditions during April, May, June, and July is 16.9%, 13.6%, 10.2%, and 38.1%, respectively, and during August, September, and October water deficiency reoccurs 52.5%, 50.0%, and 46.6% of the years, respectively. Long-term occurrence of water deficiency conditions is 32.7% of the growing season months, for a mean of 2.0 water deficient months per growing season (Manske et al. 2010).

Mean growing season precipitation during the six years of the study was 12.31 inches (83.23% of LTM). During the first three years, growing season precipitation was 9.42, 10.75, and 11.45 inches, respectively, with a mean of 10.54 inches (71.26% of LTM). During the last three years, growing season precipitation was 11.63, 13.43, and 17.15 inches, respectively, with a mean of 14.07 inches (95.13% of LTM) (table 1).

## **Grazing Management**

Restoration of native mixed grass prairie ecosystem biogeochemical processes and plant species composition on degraded rangeland requires implementation of a biologically effective grazing management strategy that activates the defoliation resistance mechanisms and ecosystem biogeochemical processes (Manske 2011). These mechanisms are: compensatory internal physiological processes, internal vegetative reproduction of secondary tillers from axillary buds, and external symbiotic rhizosphere organism activity (McNaughton 1979, 1983; Coleman et al. 1983; Ingham et al. 1985; Mueller and Richards 1986; Richards et al. 1988; Briske 1991; Murphy and Briske 1992; Briske and Richards 1994, 1995; Manske 1999, 2011). The defoliation resistance mechanisms accelerate growth rates of replacement roots, leaves, and shoots, increase photosynthetic capacity of remaining mature leaves, increase allocation of carbon and nitrogen, increase secondary tiller development from axillary buds, and increase conversion of soil organic nitrogen into plant usable mineral nitrogen (Manske 2011).

The twice-over rotation grazing management strategy was the biologically effective management practice implemented to restore the degraded mixed grass prairie biogeochemical processes and native plant communities. During 2006 to 2011, three grassland pastures were grazed from early June until mid October, with each pasture grazed for two periods. A fourth pasture was not grazed and was used as a control. Each of the three pastures in the rotation was grazed for 14 to 16 days during the first period, the 45 day interval from 1 June to 15 July, during which the defoliation resistance mechanisms can be activated by partial defoliation by grazing (Manske 1999, 2011). The length of the first period on each pasture was the same percentage of 45 days as the percentage of the total season's grazeable forage contributed by each pasture (Manske 2000). 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. The first pasture grazed in the rotation sequence was the last pasture grazed the previous year.

The stocking rate on the study area was assessed using recently updated ecological site maps and determined that a total of 789.90 AUM's of forage was available on the 1,519 acres of the three grazed pastures. With a grazing season of 4.5 months from early June to mid October, 175 AU's with a total herd weight of 175,533 pounds could graze at a stocking rate of 1.92 ac/AUM. The intended stocking rates were to graze at 75%, 85%, and 95% of the assessed stocking rate during years 1, 2, and 3, respectively, and thereafter to graze at slightly less than 100% of the assessed stocking rate for the duration of the study. The actual stocking rates used during the study were a little different than the intended stocking rates. Heavy weight livestock (1450 lbs mean weight) grazed at 72.3%, 82.6%, and 107.5% of the assessed stocking rate during years 1,

2, and 3, respectively. Medium weight livestock (900 lbs mean weight) grazed at 79.8% and 87.4% of the assessed stocking rate during years 4 and 5, respectively. Light weight livestock (650 lbs mean weight) grazed at 37.8% of the assessed stocking rate during year 6.

The twice-over rotation grazing management strategy coordinates defoliation by grazing with grass phenological growth stages which improves plant health and activates biological and ecological processes within grass plants and the ecosystem so that beneficial changes to plant growth, rhizosphere organisms, and biogeochemical cycles in the ecosystem result (Manske 2000). During the first grazing period, grass lead tillers are between the three and a half new leaf stage and the flower (anthesis) stage; these are the vegetative stages of tiller development at which partial defoliation by grazing produces beneficial effects by activating the defoliation resistance mechanisms that increase compensatory growth rates, increase tillering from axillary buds, and enhance activity of rhizosphere organisms. Increased compensatory growth rates replace leaf and stem material at greater quantities than that removed by grazing substantially increasing the amounts of herbage biomass production. Increased vegetative reproduction by tillering contributes to the development of greater plant basal cover and to the production of greater grass herbage weight. Increased activity of the soil organisms in the rhizosphere supplies the plant with greater quantities of essential nutrients, primarily mineral nitrogen, to support additional growth (Manske 2000, 2011). Removal of livestock from native rangeland pastures in mid October, towards the end of the perennial grass growing season, allows grass plants to store carbohydrates and nutrients needed to maintain plant processes over the winter that will retain the fall vegetative tiller growth and the other carryover tillers which become next season's tillers (Manske 2003, 2011). The twice-over grazing strategy ensures healthy plants in the spring and greater herbage production during the next growing season (Manske 2000, 2011).

Renewable natural resources are complex ecosystems with several trophic layers of living organisms that have biological requirements, and with numerous nonliving components that have characteristics that are transformable between organic and inorganic forms. Management of renewable natural resources needs to meet all the requirements of the living and nonliving components of the ecosystem for the purpose of improving the biogeochemical processes and maintaining production at potential sustainable levels.

Continued ecosystem production at potential sustainable levels of rangeland and grassland renewable natural resources requires that management meets the biological requirements of the plants and soil organisms. The ecosystem biogeochemical processes and the organism physiological mechanisms that provide the biological requirements for grassland plants and rhizosphere organisms must be activated annually with partial defoliation by large graminivores during vegetative growth stages. Grazing animals are a necessity for full functionality of grassland plants and rhizosphere organisms.

Performance of the defoliation resistance mechanisms and biogeochemical processes ensure healthy productive native grass plants, active rhizosphere organisms, and fully functioning mixed grass prairie ecosystems. Activation of the mechanisms and processes result in increased herbage biomass production, increased plant density (basal cover), increased available forage nutrients, increased soil aggregation, improved soil quality, increased soil water holding capacity, increased resistance to drought conditions, improved wildlife habitat, improved grassland aesthetics, and improved grassland ecosystem quality (Manske 2011).

## Procedures

The effects from a biologically effective grazing management strategy in the restoration of native mixed grass prairie ecosystem biogeochemical processes and grassland plant species composition on degraded untilled rangeland were evaluated in four pastures on silty ecological sites with permanent sample plots organized in a paired-plot design. A 16' X 32' (4.88 m X 9.75 m) stock panel exclosure prevented livestock access to an ungrazed plot. A grazed plot on an adjacent area of equal size was accessible by livestock. Ungrazed and grazed plots were established at each reference site for nondestructive data collection. An additional area of similar size accessible by livestock was established at each reference site for destructive data collection. Ecosystem changes in aboveground herbage biomass, plant species basal cover, forb density, and rhizosphere biomass were evaluated with data collected from late May through mid October during six growing seasons, 2006 to 2011. The plant basal cover and forb density data were collected along permanent transect lines established at each nondestructive sample site both inside (ungrazed) and outside (grazed) each exclosure. The major transect lines were parallel to each other on opposite sides of the exclosure fence. The minor transect lines were perpendicular to the major transect lines and were parallel to each other.

Aboveground herbage biomass was collected by the standard clipping method (Cook and Stubbendieck 1986) at each pasture rotation date, with seven clip periods per year. The herbage material from five 0.25 m<sup>2</sup> quadrats (frames) at each destructive sample site outside (grazed) each exclosure was hand clipped to ground level and sorted in the field by biotype categories: domesticated grasses, cool season grasses, warm season grasses, upland 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  $140^{\circ}$  F (60° C), and weighed. Herbage biomass in pounds per acre for each category were determined from the clipping data. Mean monthly herbage biomass for each category were determined for each growing season. Relative composition of herbage biomass biotype categories were determined.

Plant species basal cover was determined by the ten-pin point frame method (Cook and Stubbendieck 1986), with 2000 points collected along permanent transect lines during peak growth between mid July and mid August. Basal cover plant species data were sorted into biotype categories: domesticated grasses, cool season grasses, warm season grasses, upland sedges, forbs, and litter. Percent basal cover and relative composition of basal cover for the biotype categories were determined from the ten-pin point frame data.

Density of forbs were determined by counting individual stems of each plant species rooted inside twenty five  $0.1 \text{ m}^2$  quadrats placed along permanent transect lines between mid July and mid August. Forb species were categorized as: late succession, mid succession, and early succession forbs. Density per  $0.1 \text{ m}^2$  and relative composition of forb categories were determined from the forb density data.

Rhizosphere biomass was collected at each destructive sample site outside (grazed) each exclosure by three replicated soil cores 3 inches (7.6 cm) in diameter and 4 inches (10.2 cm) in depth 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.

Soil mineral nitrogen, nitrate and ammonium, was sampled towards the end of the sixth grazing season from both inside (ungrazed) and outside (grazed) each exclosure by three replicated soil cores with 6 inch (15.2 cm) increments to a 12 inch (30.5 cm) depth collected using a Veihmeyer soil tube with 1 inch (2.5 cm) diameter. Soil cores were placed on ice immediately and were frozen within 2 to 3 hours of collection. Analysis of soil core samples for available mineral nitrogen (NO<sub>3</sub> and NH<sub>4</sub>) was conducted by the North Dakota State University Soil Testing Laboratory. Total available mineral nitrogen at a one foot depth was determined from the soil core data and reported as pounds per acre.

Interpretation of treatment effects on plant community characteristics assumes only minor differences in the vegetation of the grazed area and ungrazed area at the time of exclosure construction on each reference site. A standard t-test was used to analyze differences among means (Mosteller and Rourke 1973). Nomenclature of plant species follows Flora of the Great Plains (1986).

#### Results

## **Effects of Previous Management**

The mixed grass prairie ecosystems on the study area degraded because the previous management was designed for the intended "use" and did not meet the biological requirements of the perennial native grass plants and the rhizosphere organisms and was detrimental to the biogeochemical processes. The use of rangeland natural resources should not be the objective of management. The management should be the means to accomplish the uses.

Ecosystem processes functioned at some degree less than potential level each growing season that the rangeland was managed with traditional concepts based on a use. Soon after the first ecosystem process failed to function properly, the other belowground processes and mechanisms began to deteriorate. The native grass live root biomass decreased (Whitman 1974), the defoliation resistance mechanisms within grass plants diminished, the ecosystem biogeochemical processes declined, and the competitiveness of grass plant resource uptake deteriorated (Manske 2011).

The reduction of live root surface area caused a decrease in active root length for interaction with symbiotic rhizosphere organisms and caused a decrease in absorption of water and nutrients from the soil. Reduction of active root biomass and diminishment of grass plant health vigor resulted in a loss of resource uptake efficiency and a suppression of the competitiveness of grass plants to take up mineral nitrogen, essential elements, and soil water (Li and Wilson 1998, Kochy 1999, Kochy and Wilson 2000, Peltzer and Kochy 2001). The loss of active root length was a contributing factor in the reduction of rhizosphere biomass. The primary cause for the reduction in rhizosphere biomass was, however, the great reduction in the quantity of carbohydrates exuded from the grass roots into the rhizosphere zone. The antagonistic traditional practices greatly reduced the quantity of short carbon chain energy exuded from the grass roots into the rhizosphere; this low amount of simple carbon compounds was not enough to sustain an adequate rhizosphere biomass. The small biomass of rhizosphere organisms mineralized small quantities of nitrogen and other essential elements (Coleman et al. 1983, Klein et al. 1988).

The decreased amounts of available mineral nitrogen below 100 lbs/ac in the ecosystem caused reductions in native grass herbage biomass production (Wight and Black 1972, 1979) and caused decreases in native grass density (basal cover). As degradation continued, numerous bare spaces between native grass plants were created in the plant communities. The open spaces were ideal habitat for growth of opportunistic domesticated grass species. The composition of grass species changed with decreases in the desirable native species and increases in the less desirable domesticated species.

Standing dead leaves accumulated (Brand and Goetz 1986) as ecosystem deterioration progressed. The accumulation of live and standing dead leaves of domesticated grasses reduced light penetration greatly. Reduced sunlight to native grasses caused reduced rates of photosynthesis, decreased rates of herbage production, and increased rates of leaf senescence (Langer 1972, Briske and Richards 1995) decreasing native grass composition further. Great quantities of standing dead material did not make contact with soil preventing decomposition through microbial activity and causing litter to build up into a thick mulch layer. The thick mulch modified soil temperatures, inhibited water infiltration, and tied up carbon and nitrogen (Wright and Bailey 1982; Manske 2000, 2011). Native grasses were further inhibited by deficiencies of soil water, cool soil temperatures during spring, and reduced ecosystem nutrients caused by thick mulch.

The change in plant composition from desirable native grasses to less desirable domesticated grasses was the visible symptom of ecosystem degradation; the fundamental degradation of the ecosystem was the reduction of rhizosphere biomass, the reduction of biogeochemical processes, the reduction of available mineral nitrogen below 100 lbs/ac, and the reduction of all the other essential elements. The degree of the aboveground plant species deterioration lagged behind the degree of degradation of the belowground ecosystem processes and mechanisms (Manske 2011).

There is a major fundamental problem with traditional concepts that manage renewable natural resources from the perspective of their use or for the product removed. Management of renewable resources for a use narrowly considers only a few factors directly related to that use or product, and neglects to address the needs of all the other components required for the ecosystems to function at potential levels. The renewable natural resources (rangelands, grasslands, croplands, forestlands, and fisheries) have all been managed traditionally for their use. The ecosystem processes that renew the renewable natural resources are functioning at subpotential levels. The declining production from the worlds renewable resources is a symptom of degraded ecosystem processes that have resulted from management for a use.

# **Control Pasture NG**

The mixed grass prairie study area in the ungrazed control pasture NG was a degraded silty ecological site dominated by kentucky bluegrass. Control pasture NG was not grazed during the six years of study. At the start of the study (May year 1), 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, 2.5% native grasses (2.0% cool season grasses, 0.4% upland sedges, and less than 0.1% warm season grasses), and 2.4% forbs (tables 2 and 3).

The domesticated grass herbage biomass changed little numerically throughout the study. From a starting biomass of 1684.81 lbs/ac (95.2% composition), the weight decreased (12.8%) to 1468.69 lbs/ac (72.7% composition) during years 1 to 5, and increased (38.5%) to 2333.98 lbs/ac (85.1% composition) during year 6 (tables 2 and 3). The domesticated grass basal cover changed little during the first 4 years, greatly increased during the fifth year, and greatly decreased during the sixth year. The dominant domesticated grass was kentucky bluegrass. From a starting basal cover of 10.55% (71.3%) composition), the basal cover increased (123.7%) to 23.60% (83.1% composition) during years 1 to 5, and then decreased to 15.35% (86.0% composition) during year 6 (tables 4 and 5).

The quantity of native cool season grasses was low during the study. The cool season grass herbage biomass increased slightly during the first four years and then decreased during the last two years. From a low starting biomass of 35.68 lbs/ac (2.0% composition), the weight increased (597.1%)to 248.74 lbs/ac (11.1% composition) during years 1 to 4, and then decreased (76.9%) to 57.49 lbs/ac (2.1% composition) during years 5 to 6 (tables 2 and 3). The native cool season grass basal cover increased slightly during the first three years and then decreased during the last three years. The cool season grasses were western wheatgrass, needle and thread, and green needlegrass. From a starting basal cover of 1.20% (8.1% composition), the basal cover increased (195.8%) to 3.55% (20.7% composition) during years 1 to 3, and then decreased (90.1%) to 0.35% (2.0% composition) during years 4 to 6 (tables 4 and 5).

The quantity of native warm season grasses was low during the study. The warm season grass herbage biomass changed little during the study. From an extremely low starting biomass of 0.71 lbs/ac (0.04% composition), the weight increased to 153.73 lbs/ac (5.6% composition) during years 1 to 6 (tables 2 and 3). The native warm season grass basal cover increased slightly during the study. The warm season grass remaining on the ungrazed site was a small remnant colony of prairie sandreed. From an extremely low starting basal cover of 0.05% (0.34% composition), basal cover increased a little to 0.85% (4.8% composition) during years 1 to 6 (tables 4 and 5).

The native cool and warm season grasses changed little during the study. From a low starting biomass of 36.39 lbs/ac (2.1% composition), the weight increased to 304.91 lbs/ac (13.6% composition) during years 1 to 4 and then decreased to 211.22 lbs/ac (7.7% composition) during years 5 to 6 (tables 2 and 3). The total cool and warm season grass basal cover increased slightly during the first three years, and then decreased slightly during the last three years. From a low starting basal cover of 1.25% (0.34% composition), the basal cover increased (216.09%) to 3.95% (23.0% composition) during years 1 to 3, and the decreased (69.6%) to 1.20% (6.7% composition) during years 4 to 6 (tables 4 and 5).

The quantity of upland sedges was low during the study. The upland sedge herbage biomass increased slightly during the first four years and then decreased during the last two years. From a low starting biomass of 7.14 lbs/ac (0.4% composition), the weight increased to 49.44 lbs/ac (2.2% composition) during years 1 to 4, and then decreased to 11.82 lbs/ac (0.4% composition) during years 5 to 6 (tables 2 and 3). The upland sedge basal cover decreased during the study. The upland sedge was primarily threadleaf sedge. From a starting basal cover of 2.85% (19.3% composition), the basal cover decreased (63.2%) to 1.05% (5.9% composition) during years 1 to 6 (tables 4 and 5).

The quantity of forbs was low during the study. The forb herbage biomass decreased during the first three years, increased during the fourth and fifth years, and then decreased during the sixth year. From a low starting biomass of 42.10 lbs/ac (2.4% composition), the weight decreased (35.8%) to 27.02 lbs/ac (1.8% composition) during years 1 to 3, the weight increased (783.2%) to 238.65 lbs/ac (11.8% composition) during years 4 to 5, and the weight decreased (22.3%) to 185.43 lbs/ac (6.8% composition) during year 6 (tables 2 and 3). Forb density decreased during the first three years and increased during the last three years. From a starting density of 5.44 forbs/0.10 m<sup>2</sup>, the density decreased (89.7%) to 0.56 forbs/0.10 m<sup>2</sup> during years 1 to 3, and the density increased (535.7%) to 3.56 forbs/0.10  $m^2$  during years 4 to 6.

Standing dead biomass was 1824.68 lbs/ac (28.6% composition) at the start of the study, decreasing (76.3%) to 432.54 lbs/ac (11.0% composition) during years 1 to 5, and then increasing (184.1%) to 1229.02 lbs/ac (17.2% composition) during year 6. Litter biomass was 2785.89 lbs/ac (43.7% composition) at the start of the study, decreased (47.0%) to 1476.03 lbs/ac (37.6% composition) during years 1 to 5, and then increased (115.4%) to 3178.78 lbs/ac (44.5% composition) during year 6. The litter layer was very thick during

each year on the ungrazed control pasture. The biomass of the litter was greater during the sixth year than during the first year. The mean annual litter biomass was 2356.04 lbs/ac (43.4% composition). Total dead biomass (standing dead and litter) was 4610.57 lbs/ac (72.3% composition) at the start of the study, the biomass decreased (58.6%) to 1908.57 lbs/ac (48.6% composition) during years 1 to 5, and then increased (130.9%) to 4407.80 lbs/ac (61.7% composition) during year 6. The mean annual total dead biomass was 3356.59 lbs/ac (61.8% composition).

After 6 growing seasons on the ungrazed control pasture NG, 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, 8.1% native grasses (5.6% warm season grasses, 2.1% cool season grasses, and 0.4% upland sedges), and 6.8% forbs (table 3).

The vegetation on control pasture NG changed slightly during the 6 years of nongrazing management. Domesticated grass herbage biomass increased 38.5% and basal cover increased 45.5%. Cool season grass herbage biomass increased 61.1% and basal cover decreased 70.8%. Warm season grass herbage biomass increased 2155.1% and basal cover increased 1600.0%. Upland sedge herbage biomass increased 65.5% and basal cover decreased 63.2%. Forb herbage biomass increased 340.5% and density decreased 34.6%. Total live herbage biomass increased 54.9% and total live basal cover increased 20.6%. Standing dead herbage biomass decreased 32.6% and litter biomass increased 14.1%.

The total available soil mineral nitrogen was 81.36 lbs/ac. The quantity of nitrate was 10.50 lbs/ac and the quantity of ammonium was 70.86 lbs/ac. The amount of available mineral nitrogen was high, indicating that the level of plant growth was reduced and the rate of available mineral nitrogen use was also reduced. The high quantities of nitrate appear to be related to the greater quantities of easily decomposed labile roots of domesticated grasses. The ungrazed control pasture had high domesticated grass basal cover. The high quantities of ammonium are usually related to greater quantities of native grass roots and greater rhizosphere biomass. However, the ungrazed control pasture had extremely low basal cover of native grasses. The ungrazed control pasture did have high kentucky bluegrass basal cover. It appears that the high quantity of ammonium on the ungrazed control pasture could be related to the easily decomposed labile roots of kentucky bluegrass with low rhizosphere biomass.

The domesticated grasses and native grasses on the ungrazed control pasture responded to water stress at different levels. The amount of precipitation during the first three years was a little less than 75% of the long-term mean resulting in early stages of water stress. The domesticated grasses were under water stress and the herbage biomass production decreased (21.6%) to 1320.16 lbs/ac. The native grasses, however, were not under as much water stress and the reduction of domesticated grass herbage biomass permitted the native grasses to increase production (273.9%) to 162.75 lbs/ac during the first three years with low precipitation.

The amount of precipitation during the last three years was greater at about 95% of the long-term mean. The domesticated grasses were no longer under water stress and the herbage biomass production increased (76.8%) to 2333.98 lbs/ac. The increased domesticated grass herbage biomass increased the shading effect on most of the smaller native grasses and caused a reduction in herbage biomass. However, a small remnant colony of prairie sandreed, a tall native grass, was able to grow above the shading by the domesticated grass leaves and was able to produce some herbage biomass at 223.05 lbs/ac which was an increase of 37.1% greater than the native grass herbage biomass production during the three years with low precipitation.

Nondefoliated live and standing dead leaves of grasses accumulated on the ungrazed control pasture NG and greatly reduced light penetration. Grass plants produce double the quantity of leaf biomass than needed for normal plant growth and maintenance as an evolutionary survival mechanism in response to partial defoliation and the loss of leaf area as forage to grazing graminivores (Crider 1955, Covne et al. 1995). This mechanism does not stop on ungrazed pastures. Without grazing graminivores to remove half of the annual herbage production, the surplus leaf material accumulated rapidly and changed from an asset to a detriment. The accumulation of overstory vegetation reduced light penetration below native grass light saturation points (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 in low light conditions.

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

The standing dead biomass on ungrazed pastures rapidly accumulated and the resulting tanglement of leaves and stems could not make contact with the soil surface and decompose quickly through microbial activity. The standing dead biomass decreased slowly by leaching and weathering and built up into a thick mulch layer. Thick mulch effectively blocked sunlight from reaching understory young grass leaves. Thick mulch insulates the soil from warm spring air temperatures preventing heating of cold soil that caused delays in plant and soil organism activity. Thick mulch ties up and holds organic nutrients above the soil surface preventing accession to the soil organic matter which limits nutrient cycling through biogeochemical processes increasing the deficiencies of essential elements. Thick mulch absorbs and holds precipitation for later evaporation preventing the water from infiltrating into the soil diminishing soil water to deficiency quantities (Wright and Bailey 1982, Manske 2000, 2011). These undesirable modifications to the ecosystem cause decreases in soil microorganism biomass and activity resulting in further reductions in the rates of organic material decomposition (Anderson et al. 1981, Curl and Truelove 1986, Whipps 1990).

The decreased supply of soil water, mineral nitrogen, and fixed carbon resulted in a major reduction in assimilation of plant tissue, reducing growth of leaves and roots, and reducing the development of vegetative secondary tillers (Langer 1972, Briske and Richards 1995). Native grass tiller mortality increased and native plant density decreased (Grant et al. 1983), creating large open spaces available for invasion by the less desirable domesticated cool season grasses, kentucky bluegrass and smooth bromegrass. The increasing live herbage biomass and increasing standing dead biomass of the invading domesticated grasses caused additional shading that resulted in accelerated reductions of the native grasses.

## **Grazed Pastures TOR**

The mixed grass prairie study areas in grazed pastures TOR were degraded silty ecological sites dominated by kentucky bluegrass. At the start of the study (May year 1), 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, 26.2% native grasses (22.9% upland sedges, 2.7% cool season grasses, and 0.6% warm season grasses), and 9.0% forbs (table 7).

The domesticated grass herbage biomass decreased during the first and third years, increased slightly during the second, fourth, and fifth years, and increased greatly during the sixth year. From a pregrazing biomass of 1066.48 lbs/ac (64.8% composition), the weight decreased (35.7%) to 685.72 lbs/ac (39.8% composition) during years 1 to 5, and increased (83.9%) to 1261.24 lbs/ac (58.0% composition) during year 6 (tables 6 and 7). The domesticated grass basal cover increased during the first 5 years. The domesticated grasses were primarily kentucky bluegrass with small quantities of smooth bromegrass and crested wheatgrass. From a pregrazing basal cover of 3.45% (24.8% composition), the basal cover increased (99.4%) to 6.88% (22.7% composition) during years 1 to 5 and 6 (tables 8 and 9). Kentucky bluegrass had the greatest basal cover increase.

The native cool season grass herbage biomass increased during the first six years. From a pregrazing biomass of 43.53 lbs/ac (2.7% composition), the weight increased (1090.4%) to 518.18 lbs/ac (23.8% composition) during years 1 to 6 (tables 6 and 7). The native cool season grass basal cover increased during the first five years and then decreased during the sixth year. The primary cool season grasses were western wheatgrass, needle and thread, and prairie junegrass. From a pregrazing basal cover of 1.85% (13.3% composition), the basal cover increased (228.6%) to 6.08% (20.1% composition) during years 1 to 5, and then decreased to 3.93% (16.8% composition) during year 6 (tables 8 and 9).

The native warm season grass herbage biomass increased during the first six years. From a pregrazing biomass of 10.00 lbs/ac (0.6% composition), the weight increased (388.9%) to 48.89 lbs/ac (2.3% composition) during years 1 to 6 (tables 6 and 7). The warm season grass basal cover increased during the first six years. The primary warm season grasses were blue grama, prairie sandreed, and plains muhly. From a pregrazing basal cover of 0.43% (3.1% composition), the basal cover increased (488.4%) to 2.53% (10.8% composition) during years 1 to 6 (tables 8 and 9).

The total native cool and warm season grass herbage biomass increased during the first six years. From a pregrazing biomass of 53.53 lbs/ac (3.3% composition), the weight increased (959.3%) to 567.07 lbs/ac (26.1% composition) during years 1 to 6 (tables 6 and 7). The total native cool and warm season grass basal cover increased during the first five years, and then decreased slightly during the sixth year. From a pregrazing basal cover of 2.28% (16.4% composition), the basal cover increased (236.0%) to 7.66% (25.3% composition) during years 1 to 5, and then decreased (15.7%) to 6.46% (27.6% composition) during year 6 (tables 8 and 9).

The native upland sedge herbage biomass decreased during the first six years. From a pregrazing biomass of 377.14 lbs/ac (22.9% composition), the weight decreased (35.0%) to 245.17 lbs/ac (11.3% composition) during years 1 to 6 (tables 6 and 7). The upland sedge basal cover increased during the first five years, and then decreased during the sixth year. The upland sedge was threadleaf sedge. From a pregrazing basal cover of 7.63% (54.8% composition), the basal cover increased (66.4%) to 12.70% (41.9% composition) during years 1 to 5, and then decreased (24.8%) to 9.55% (40.9% composition) during year 6 (tables 8 and 9).

The forb species composition was dynamic on the grazed pastures. The forb herbage biomass decreased during the first three years, increased greatly during the fourth and fifth years, and then decreased during the sixth year. From a pregrazing biomass of 147.72 lbs/ac (9.0% composition), the weight decreased (76.2%) to 35.17 lbs/ac (4.3%) composition) during years 1 to 3, the forb biomass increased (917.5%) to 357.87 lbs/ac (20.8% composition) during years 4 to 5, and then decreased (72.0%) to 100.11 lbs/ac (4.6% composition) during year 6 (tables 6 and 7). Forb density decreased during the first three years and then the densities of blue wild lettuce and yellow sweetclover increased greatly during the last three years. From a pregrazing density of 5.52 forbs/0.10 m<sup>2</sup>, the density of the late, mid, and early succession forbs all decreased (76.1%) to a total density of 1.32 forbs/0.10 m<sup>2</sup> during years 1 to 3. The early succession forbs increased greatly by 4666.7% and 4655.6% during years 4 and 6, respectively, resulting in total forb density increases (790.9%) to 11.76 forbs/0.10 m<sup>2</sup> and increases

(753.0%) to 11.26 forbs/0.10  $m^2$  during years 4 and 6, respectively.

Standing dead biomass was 1213.84 lbs/ac (26.9% composition) at the start of the study, the biomass decreased (91.2%) to 107.40 lbs/ac (4.9% composition) during years 1 to 4, and then increased (374.6%) to 509.77 lbs/ac (14.2% composition) during years 5 to 6. Litter biomass was 1661.26 lbs/ac (36.8% composition) at the start of the study, the biomass decreased (71.5%) to 473.94 lbs/ac (18.5% composition) during years 1 to 5, and then increased (89.5%) to 898.17 lbs/ac (25.1% composition) during year 6. The litter layer was not thick after the first year and the litter biomass averaged 825.60 lbs/ac during years 2 to 6. The total dead biomass (standing dead and litter) was 2875.10 lbs/ac (63.6% composition) at the start of the study, the total biomass decreased (75.0%) to 718.19 lbs/ac (32.8% composition) during years 1 to 4, and then increased (96.0%) to 1407.93 lbs/ac (39.3% composition) during years 5 to 6 (tables 6 and 7).

After 6 growing seasons managed with the twice-over rotation system, the aboveground vegetation biomass on grazed pastures TOR consisted of 39.3% standing dead and litter and 60.7% live herbage. The live herbage was 58.0% domesticated grasses, 37.4% native grasses (23.8% cool season grasses, 11.3% upland sedges, and 2.3% warm season grasses), and 4.6% forbs (table 7).

The vegetation on grazed pastures TOR changed for the better during the 6 years of twiceover rotation grazing management. Domesticated grass herbage biomass increased 18.3% and basal cover increased 99.4%. Cool season grass herbage biomass increased 109.0% and basal cover increased 112.4%. Warm season grass herbage biomass increased 388.9% and basal cover increased 488.4%. Upland sedge herbage biomass decreased 35.0% and basal cover increased 25.2%. Forb herbage biomass decreased 32.2% and forb density increased 104.0%. Total live herbage biomass increased 67.8%. Standing dead herbage biomass decreased 58.0% and litter biomass decreased 45.9%.

The total available soil mineral nitrogen of nitrate and ammonium was 53.78 lbs/ac on the exclosure and 56.74 lbs/ac on the grazed area, with an increase of 5.5% on the grazed area. The quantity of mineral nitrogen was greater on the grazed area than on the ungrazed exclosure. The quantity of nitrate was 6.84 lbs/ac on the exclosure and 5.67 lbs/ac on the grazed area, with a decrease of 17.1% on the

grazed area. The quantity of ammonium was 46.95 lbs/ac on the exclosure and 51.08 lbs/ac on the grazed area, with an increase of 8.8% on the grazed area. The exclosure had greater nitrate and lower ammonium and the grazed area had lower nitrate and greater ammonium. The quantity of nitrate was reduced on both the exclosure and grazed area. Reduced quantities of nitrate appear to be related to reduced quantities of easily decomposed labile roots of domesticated grasses. Both the exclosure and grazed area had reduced domesticated grass basal cover. The grazed area had greater ammonium. The greater quantities of animonium appear to be related to the greater quantities of native grass roots and greater rhizosphere biomass.

The degraded mixed grass prairie silty ecological sites on grazed pastures TOR were dominated by kentucky bluegrass with 64.8% composition. The native cool and warm season grasses had been reduced to 3.3% composition. These degraded prairie sites were managed with the twice-over rotation system for 6 years. During the first three years, heavy weight livestock with mean weight of 1450 lbs were stocked at 72.3%, 82.6%, and 107.5% of the assessed stocking rate, respectively, and the livestock consumed 394 lbs, 450 lbs, and 585 lbs of herbage weight per acre, respectively. During the first three years, standing dead biomass decreased (65.3%) to 420.37 lbs/ac, litter biomass decreased (32.9%) to 1114.80 lbs/ac, domesticated grass herbage biomass decreased (70.9%) to 310.77 lbs/ac, and total live herbage biomass decreased (49.9%) to 824.67 lbs/ac. The native cool and warm season grasses increased (295.6%) to 211.74 lbs/ac.

The near full stocking rates at a mean of 87.5% of the assessed stocking rate of the heavy weight livestock that consumed a mean of 476 lbs of herbage biomass per acre removed sufficient quantities of domesticated grass and standing dead herbage biomass to substantially reduce the shading problems, permitting sunlight to reach the shorter native grasses. Partial defoliation of the native grasses by grazing livestock during vegetative phenological stages stimulated the defoliation resistance mechanisms, activated the rhizosphere organisms, and enhanced the ecosystem biogeochemical processes resulting in increased herbage biomass production and greater vegetative tiller growth.

The stocking rate at greater than 100% of the assessed value during year 3 caused both beneficial effects and negative effects. The third growing

season was the only time that grazing livestock were at great enough quantities to remove 916.93 lbs/ac of domesticated grass and standing dead biomass and open the grass leaf canopy sufficiently. Carryover effects lasted into the following grazing season during year 4. Standing dead biomass decreased (74.5%) to 107.40 lbs/ac and litter biomass decreased (45.2%) to 610.79 lbs/ac. Native cool and warm season grass herbage biomass increased (109.8%) to 444.27 lbs/ac. However, the negative factors were also great, total forb biomass increased (168.7%) to 94.50 lbs/ac and weedy forb density increased 4666.7% during the next growing season as a result of opening the grass canopy excessively and exposing sunlight on large areas of soil causing weed seeds to germinate.

During the fourth and fifth years, medium weight livestock with mean weight of 900 lbs were stocked at 79.8% and 87.4% of the assessed stocking rate, respectively, and the livestock consumed 304 lbs and 333 lbs of herbage weight per acre, respectively. The quantity of herbage biomass consumed per acre was insufficient to maintain ecosystem improvement. Standing dead biomass increased (238.6%) to 363.68 lbs/ac, domesticated grass herbage biomass increased (120.7%) to 685.72 lbs/ac, and forb herbage biomass increased (278.7%) to 357.87 lbs/ac. Native cool and warm season grass herbage biomass decreased (21.9%) to 346.76 lbs/ac.

During the sixth year, light weight livestock with mean weight of 650 lbs were stocked at 37.8% of the assessed stocking rate and the livestock consumed 113 lbs of herbage weight per acre. At this low stocking rate, the quantity of herbage biomass consumed per acre was so inadequate that ecosystem restoration stopped, causing the return of advancements of degradation. Standing dead biomass increased (40.2%) to 509.77 lbs/ac, litter biomass increased (89.5%) to 898.17 lbs/ac, domesticated grass herbage biomass increased (83.9%) to 1261.24 lbs/ac, and early succession forb density increased 256.7%. Native cool and warm season grass herbage biomass increased (63.5%) to 567.07 lbs/ac as a result of continued effects from previous beneficial partial defoliation by grazing at heavier stocking rates.

#### **Differences in Restoration Changes**

The restoration changes of the ecosystems on the ungrazed control pasture NG and on the grazed pastures TOR were different. At the start of the study, a similarity index of 52.6% indicated that the degraded plant communities on the ungrazed control pasture NG and on the grazed pastures TOR were more similar than dissimilar.

During the study, the grazing managed with twice-over rotation strategy on pastures TOR reduced the standing dead biomass 58.5% and reduced the litter biomass 71.7% below that on the ungrazed control pasture NG. This reduction in standing dead decreased the problems caused by shading that reduced the rates of photosynthesis and increased the rates of leaf senescence in native grasses. This reduction in litter decreased the problems caused by thick mulch that modified soil temperatures, inhibited water infiltration, and tied up carbon and nitrogen.

During the study, domesticated grass herbage biomass increased 649.17 lbs/ac (38.5%) and basal cover increased 4.80 percentage points (45.5%) on the ungrazed pasture NG. Domesticated grass herbage biomass increased 194.76 lbs/ac (18.3%) and basal cover increased 3.43 percentage points (99.4%) on the grazed pastures TOR. At the end of the study, domesticated grass herbage biomass was 46.0% less and basal cover was 55.2% less on the grazed pastures TOR than on the ungrazed control pasture NG. The rate of increase of domesticated grass herbage biomass and basal cover was restricted by the grazing treatment.

The combined aboveground vegetation biomass of domesticated grass and standing dead caused the shading problem and prevented most of the sunlight from reaching the smaller native grass plants growing in the understory. During the study, the annual mean combined vegetation biomass was 2617.80 lbs/ac on the ungrazed pasture NG and was significantly less at 1277.06 lbs/ac on the grazed pastures TOR. The grazing treatment reduced the shading problem 51.2% which was sufficient to permit greater amounts of sunlight to reach native grass leaves, increasing the photosynthetic rates that increased the quantities of fixed carbon available to native grasses resulting in increased growth rates. The annual mean native grass herbage biomass on the ungrazed pasture NG was only 245.28 lbs/ac and was significantly greater at 650.01 lbs/ac on the grazed pastures TOR. The grazing treatment increased the native grass production 165.0% as a result of reducing the shading problem.

At the end of the study, cool and warm season grass herbage biomass was 168.5% greater and basal cover was 438.3% greater on the grazed pastures than on the ungrazed pasture. Native upland sedge herbage biomass was 1974.2% greater and basal cover was 809.5% greater on the grazed pastures than on the ungrazed pasture. Forb herbage biomass was 46.0% less and forb density was 216.3% greater on the grazed pastures than on the ungrazed pasture.

The improvements of the native grasses in the degraded native mixed grass prairie communities on the grazed pastures TOR indicated that the defoliation resistance mechanisms were activated to some degree by the twice-over rotation grazing management strategy. Increased activity of the compensatory internal physiological growth processes resulted in increased production of native grass herbage biomass. Increased activity of the internal vegetative reproductive processes resulted in increased native grass basal cover. At the end of the study, a similarity index of 37.3% indicated that the improved plant communities on grazed pastures TOR and the degraded plant communities on ungrazed control pasture NG were more dissimilar than similar.

The previous management degraded the plant communities and caused reductions in the quantity of plant carbon exudates released into the rhizosphere that caused a decrease in rhizosphere biomass. At the start of the study, the rhizosphere biomass on ungrazed control pasture NG had decreased to 52.23 kg/m<sup>3</sup>. The rhizosphere biomass on grazed pastures TOR had decreased to 77.99 kg/m<sup>3</sup>. These low quantities of rhizosphere biomass are less than 20% of the potential rhizosphere biomass of 406.44 kg/m<sup>3</sup> recorded on silty ecological sites managed long-term with a twice-over rotation grazing strategy.

Rhizosphere organism biomass and activity are limited by access to simple carbon chains because the microflora trophic levels lack chlorophyll and have low carbon (energy) content. Partial defoliation of grass plants at vegetative phenological growth stages by large grazing graminivores causes greater quantities of exudates containing simple carbon compounds to be released through the plant roots into the rhizosphere. With the increase in availability of carbon compounds in the rhizosphere, activity of the microorganisms increases. The increase in rhizosphere organism activity causes an increase in rhizosphere volume and biomass. Without partial defoliation by grazing at specific grass growth stages, only small quantities of plant material leak from the grass roots into the rhizosphere; this low amount of carbon compounds is barely enough to sustain a small rhizosphere biomass at less than 20% of potential biomass (Manske 1999, 2011).

During the first two years of the study, the rhizosphere weights on the grazed pastures were numerically greater than the rhizosphere weights on the ungrazed control pasture, but were not significantly different. However, the rhizosphere weights on the grazed pastures became significantly greater than those on the ungrazed pasture during years 3 to 6 (table 10 and figure 1).

The rhizosphere weights on the ungrazed control pasture NG changed little during years 1 to 5. The changes in rhizosphere weight did appear to be responses to changes in growing season precipitation. The rhizosphere weights on the ungrazed pasture were not significantly different during years 1 to 3, when the mean growing season precipitation was 10.54 inches (71.26% of LTM). The rhizosphere weights during years 4 and 5 increased slightly when the mean growing season precipitation increased slightly to 12.53 inches (84.72% of LTM). The rhizosphere weights were not significantly different from each other during years 4 and 5 but were significantly greater than the rhizosphere weights during years 1 to 3. The rhizosphere weights increased greatly when the growing season precipitation increased greatly to 17.15 inches (115.96% of LTM) during year 6. The rhizosphere weights on ungrazed pasture NG were significantly greater during year 6 than those during years 1 to 5 (table 10 and figure 1). Changes in rhizosphere weights on the ungrazed pasture appeared to be related to changes in growing season precipitation, or more specifically to changes in the availability of hydrogen, which in turn effected the quantity of atmospheric carbon fixed during photosynthesis increasing the quantity of carbohydrates. With greater quantities of plant carbohydrates, the quantity of simple carbon compounds that leaked from grass roots into the rhizosphere increased proportionally, increasing the rhizosphere biomass. During this study, the rhizosphere biomass on the ungrazed pasture changed from 12.9% to 32.1% of the potential rhizosphere biomass on long-term twiceover rotation management strategies.

The rhizosphere weights on the grazed pastures TOR did not change significantly during years 1 and 2, and the rhizosphere weights on the grazed pastures were not significantly different than those on the ungrazed pasture during the first 2 years. The rhizosphere weights increased 33% between the second and third years on the grazed pastures and continued to increase weight at a mean rate of 30.5 kg/m<sup>3</sup> per year from the second year to the sixth year which was 131.5% greater than the change in rhizosphere weights on the ungrazed control pasture. The rhizosphere weights on the grazed pastures were significantly greater than those on the ungrazed control pasture each year from year three to year six (table 10 and figure 1). The rhizosphere biomass increases during years three to six on the grazed pastures appeared to be related to increases in carbon exudates that resulted from partial defoliation by grazing of grass lead tillers during vegetative growth stages. During this study, the rhizosphere biomass on the grazed pastures changed from 19.2% to 52.7% of the potential rhizosphere biomass on long-term twice-over rotation management strategies.

Six years of twice-over rotation grazing management increased the rhizosphere biomass 175% and improved the plant community composition of native grasses 43%, however, neither the rhizosphere or the plant community had been fully restored and require continuation of the grazing treatment.

# Discussion

Traditionally, rangeland ecosystems have been managed from the perspective of the "use" of the grassland. Livestock grazing along with watershed, wildlife, and recreation were considered to be the major uses. Management of rangelands from the perspective of a single use or for multiple uses narrowly considers only a few ecosystem components directly related with these primary uses or products removed. Management for a use does not consider rangelands as complex ecosystems and neglects to address the needs of all other ecosystem components. Management of rangelands for a use results in degradation of the ecosystems.

Rangelands are complex ecosystems consisting of numerous interactive biotic (living) and abiotic (nonliving) components. The biotic components are the plants, soil organisms, and large grazing graminivores that have biological and physiological requirements. The abiotic components require the presence of sunlight and include the essential major elements of carbon, hydrogen, nitrogen, and oxygen that have transformable characteristics between organic and inorganic forms through biogeochemical processes. Rangeland ecosystems are functioning units of coacting biotic organisms interacting with the abiotic components and the environment. The complex of mechanisms and processes connected with these extensive interactions have been collectively identified as defoliation resistance mechanisms and biogeochemical processes. If any of the numerous processes are not functioning at potential level, the ecosystem does not produce at potential levels.

Management of rangeland ecosystems needs to meet the biological and physiological requirements of the biotic components and activate the biogeochemical processes that cycle the abiotic components. Mixed grass prairie communities require biologically effective partial defoliation by annually managed grazing animals in order to persist as healthy and productive ecosystems. Thus, providing the means to accomplish the uses of watershed, wildlife habitat, recreation, and livestock forage at the same time on fully functional rangeland ecosystems.

Implementation of the biologically effective twice-over rotation grazing management strategy activates the defoliation resistance mechanisms meeting the biological and physiological requirements of the biotic components and activates the biogeochemical processes that cycle the abiotic components (Manske 2011). The three main defoliation resistance mechanisms are : compensatory internal physiological processes, internal vegetative reproduction of secondary tillers from axillary buds, and external symbiotic rhizosphere organism activity (McNaughton 1979, 1983; Coleman et al. 1983; Ingham et al. 1985; Mueller and Richards 1986; Richards et al. 1988; Briske 1991; Murphy and Briske 1992; Briske and Richards 1994, 1995; Manske 2009, 2011).

The defoliation resistance mechanisms developed early during the coevolution of grass plants and grazing graminivores (McNaughton 1979, 1983; Coleman et al. 1983; Briske 1991; Briske and Richards 1995; Manske 1999, 2011) and are a complex assemblage of biogeochemical and physiological processes that involve intricate interactions among rhizosphere microorganisms, grass plants, and large grazing graminivores. Activation of these mechanisms provides important biological and physiological processes permitting native grasses to produce greater herbage biomass and to increase basal cover; these mechanisms also enable grass plants to replace lost leaf material, to restore disrupted physiological processes, and to vegetatively reproduce secondary tillers from axillary buds after partial defoliation by grazing. The defoliation resistance mechanisms function at variable levels of activation depending on the quantity of available mineral nitrogen in the ecosystem soil and on the quantity of available recently fixed carbon (Richards and Caldwell 1985). When mineral nitrogen is available at 100 lbs/ac or greater, the defoliation resistance mechanisms function at full activation. When mineral nitrogen is available at less than 100 lbs/ac, the defoliation resistance mechanisms function at levels less than full activation (Manske 2009). In addition, the water (precipitation) use efficiency processes decrease in grass plants growing in ecosystems with less than 100 lbs/ac available mineral nitrogen causing herbage biomass production to be reduced by 49.6% (Wight and Black 1972, 1979).

The quantity of available mineral nitrogen in grassland ecosystem soils is dependent on the rate of mineralization of soil organic nitrogen by rhizosphere organisms. The larger the rhizosphere volume and microorganism biomass, the greater the quantity of soil mineral nitrogen converted. Rhizosphere volume and microorganism biomass are limited by access to simple carbohydrates (Curl and Truelove 1986). Healthy grass plants capture and fix carbon during photosynthesis and produce carbohydrates in quantities greater than the amount needed for tiller growth and development (Coyne et al. 1995). Partial defoliation of grass lead tillers at vegetative physiological growth stages by large grazing graminivores causes greater quantities of exudates containing simple carbohydrates to be released from the grass tillers through the roots into the rhizosphere (Hamilton and Frank 2001). With the increase in availability of 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 converting 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). Mineral nitrogen available in quantities of 100 lbs/ac or greater allows partially defoliated grass tillers full activation of the defoliation resistance mechanisms (Manske 2009). Full activation of the compensatory internal physiological processes within grass plants accelerates growth rates of replacement roots, leaves, and shoots, increases photosynthetic capacity of remaining mature leaves, increases allocation of carbon and nitrogen, 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 internal processes of vegetative reproduction increases secondary tiller development from axillary buds and increases initiated tiller density during the grazing season. Full activation of the external symbiotic rhizosphere organism activity increases mineralization of inorganic nitrogen, increases ecosystem biogeochemical cycling of essential elements, and improves belowground resource uptake

competitiveness (Wight and Black 1972, 1979; McNaughton 1979, 1983; Coleman et al. 1983; Ingham et al. 1985; Mueller and Richards 1986; Richards et al. 1988; Briske 1991; Murphy and Briske 1992; Briske and Richards 1994, 1995; Manske 1999, 2011; Kochy and Wilson 2000).

Restoration of degraded mixed grass prairie ecosystems depends on the successfulness of recovery of the biogeochemical processes and revival of the indispensable component resources. The rhizosphere organism biomass must be improved initially to increase mineralization of nitrogen and other essential elements. Rhizosphere organisms increase when greater quantities of short carbon chain energy are exudated by partial defoliation by grazing. Two growing seasons are required before substantial increases in rhizosphere biomass occur. The rhizosphere biomass continues to increase as the quantity of short carbon chain energy exudated from native grasses increases.

Removal of substantial quantities of the overstory vegetation comprised of domesticated grass live and standing dead herbage biomass increases sunlight intensities reaching the understory native grass leaves sufficiently to enhance the photosynthetic rates and to increase the quantity of available fixed carbon. Reduction of adequate quantities of overstory vegetation requires grazing graminivores at relatively high stocking rates, around 85% to 100% of the assessed level. Stocking rates greater than 100% of assessed level can remove great quantities of overstory vegetation in a short time period. However, because native grass plants cannot fill the plant community open spaces rapidly, exposure of sunlight to large proportions of bare ground causes negative effects that initiate extreme increases in weedy forbs that remain problems for several years. Stocking rates less than 85% of assessed level remove insufficient quantities of overstory vegetation to effectively reduce the shading problem. Native grasses in the understory receiving insufficient sunlight continue to decline. Stocking rates between 85% and 100% of the assessed level remove enough overstory vegetation to permit an increased intensity of sunlight to reach the leaves of native grasses that increases the photosynthetic rates and fixes carbon at greater quantities sufficient for increased grass growth to progress at reasonable restoration rates.

Dead plant material does not decompose through microbial activity unless it makes contact with soil. Livestock do not preferentially consume old dead litter. However, the presence of the livestock caused greater proportions of the thick mulch to make soil contact and reduce the litter mulch biomass by greater than 70% of the litter biomass on areas without livestock.

Restoration of mixed grass prairie ecosystems degraded by traditional management concepts requires implementation of a biologically effective grazing management strategy that activates the ecosystem biogeochemical processes and the defoliation resistance mechanisms. The activation trigger for these processes and mechanisms is partial defoliation by large grazing graminivores that removes 25% to 33% of the aboveground leaf biomass from grass lead tillers at vegetative growth stages between the 3.5 new leaf stage and the flower (anthesis) stage (Manske 1999, 2011). In addition, reduction of the shading problem and of the thick mulch problem will assist the restoration process. Following recovery of some of the biogeochemical processes, the previously deficient quantities of available mineral nitrogen, essential elements, fixed carbon, and soil water start increasing upwards towards functional quantities. With the increase of component resources, native grass plants are able to synthesize increasing quantities of carbohydrates, proteins, and nucleic acids. Activation of the defoliation resistance mechanisms provides important biological and physiological processes permitting native grasses to use the vital organic compounds in increasing quantities for the production of herbage biomass and basal cover. Within a few years, the composition of native grasses increases in the plant community and becomes the dominant vegetation. The native grasses improve in competitiveness for the belowground resources of soil water, mineral nitrogen, and other essential elements which eventually reduces the remaining introduced domesticated grasses to minor composition in the plant community. The length of time required to reach recovery is related exponentially to the severity of degradation. The belowground ecosystem processes and mechanisms must be restored before the aboveground plant community can be restored.

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

Pasture NG	Pregrazing	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6
Domesticated	1684.81	1833.85	1791.31	1320.16	1779.62	1468.69	2333.98
Cool Season	35.68	73.30	110.22	97.30	248.74	180.95	57.49
Warm Season	0.71	59.02	126.59	45.16	56.17	102.76	153.73
Native Grass	36.39	132.32	236.81	142.46	304.91	283.71	211.22
Sedges	7.14	25.91	22.27	20.29	49.44	30.48	11.82
Forbs	42.10	128.75	75.97	27.02	116.81	238.65	185.43
Total Live	1770.44	2120.83	2126.36	1509.93	2250.78	2021.53	2742.47
Standing Dead	1824.68	1381.12	708.48	928.70	499.32	432.54	1229.02
Litter	2785.89	2452.03	2131.29	2521.86	1946.39	1476.03	3178.78
Total Dead	4610.57	3833.15	2839.77	3450.56	2445.71	1908.57	4407.80
Total Biomass	6381.01	5953.98	4966.13	4960.49	4696.49	3930.10	7150.27

 Table 2. Mean herbage biomass (lbs/ac) for nongrazed silty native rangeland sites dominated by kentucky bluegrass.

 Table 3. Percent composition of herbage biomass for nongrazed silty native rangeland sites dominated by kentucky bluegrass.

Pasture NG	Pregrazing	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6
Domesticated	95.16	86.47	84.24	87.43	79.07	72.65	85.11
Cool Season	2.02	3.46	5.18	6.44	11.05	8.95	2.10
Warm Season	0.04	2.78	5.95	2.99	2.50	5.08	5.61
Native Grass	2.06	6.24	11.13	9.43	13.55	14.03	7.71
Sedges	0.40	1.22	1.05	1.34	2.20	1.51	0.43
Forbs	2.38	6.07	3.57	1.79	5.19	11.81	6.76
Total Live	27.75	35.62	42.82	30.44	47.92	51.44	38.35
Standing Dead	28.60	23.20	14.27	18.72	10.63	11.01	17.19
Litter	43.66	41.18	42.92	50.84	41.44	37.56	44.46
Total Dead	72.25	64.38	57.18	69.56	52.08	48.56	61.65

Pasture NG	Pregrazing	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6
Domesticated	10.55	12.35	19.95	11.20	15.30	23.60	15.35
Cool Season	1.20	0.40	2.30	3.55	0.70	1.25	0.35
Warm Season	0.05	0.50	0.80	0.40	0.55	0.65	0.85
Native Grass	1.25	0.90	3.10	3.95	1.25	1.90	1.20
Sedges	2.85	2.00	2.20	1.90	2.35	1.75	1.05
Forbs	0.05	0.80	0.30	0.10	0.10	1.15	0.20
Total Live	14.80	16.05	25.55	17.15	19.00	28.40	17.85
Litter	85.20	83.95	74.45	82.85	81.00	71.60	82.15
Total	100.00	100.00	100.00	100.00	100.00	100.00	100.00

Table 4. Basal cover (%) for nongrazed silty native rangeland sites dominated by kentucky bluegrass.

 Table 5. Percentage composition (%) of basal cover for nongrazed silty native rangeland sites dominated by kentucky bluegrass.

Pasture NG	Pregrazing	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6
Domesticated	71.28	76.95	78.08	65.31	80.53	83.10	85.99
Cool Season	8.11	2.49	9.00	20.70	3.68	4.40	1.96
Warm Season	0.34	3.12	3.13	2.33	2.89	2.29	4.76
Native Grass	8.45	5.61	12.13	23.03	6.57	6.69	6.72
Sedges	19.26	12.46	8.61	11.08	12.37	6.16	5.88
Forbs	0.34	4.98	1.17	0.58	0.53	4.05	1.12
Total Live	14.80	16.05	25.55	17.15	19.00	28.40	17.85
Litter	85.20	83.95	74.45	82.85	81.00	71.60	82.15

Pastures TOR	Pregrazing	Year1	Year 2	Year 3	Year 4	Year 5	Year 6
Domesticated	1066.48	954.70	1156.08	310.77	547.59	685.72	1261.24
Cool Season	43.53	177.90	300.19	188.95	395.69	307.77	518.18
Warm Season	10.00	28.19	42.21	22.79	48.58	38.99	48.89
Native Grass	53.53	206.09	342.40	211.74	444.27	346.76	567.07
Sedges	377.14	287.58	264.50	266.99	382.80	334.68	245.17
Forbs	147.72	122.28	77.06	35.17	94.50	357.87	100.11
Total Live	1644.85	1570.64	1840.03	824.67	1469.15	1725.02	2173.57
Standing Dead	1231.84	853.06	491.99	420.37	107.40	363.68	509.77
Litter	1661.26	1479.24	1030.31	1114.80	610.79	473.94	898.17
Total Dead	2875.10	2332.30	1522.30	1535.16	718.19	837.62	1407.93
Total Biomass	4520.05	3902.94	3362.33	2359.83	2187.34	2562.64	3581.51

Table 6. Mean herbage biomass (lbs/ac) for grazed silty native rangeland sites dominated by kentucky bluegrass.

 Table 7. Percent composition of herbage biomass for grazed silty native rangeland sites dominated by kentucky bluegrass.

Pastures TOR	Pregrazing	Year1	Year 2	Year 3	Year 4	Year 5	Year 6
Domesticated	64.84	60.78	62.83	37.68	37.27	39.75	58.03
Cool Season	2.65	11.33	16.31	22.91	26.93	17.84	23.84
Warm Season	0.61	1.79	2.29	2.76	3.31	2.26	2.25
Native Grass	3.25	13.12	18.61	25.68	30.24	20.10	26.09
Sedges	22.93	18.31	14.37	32.38	26.06	19.40	11.28
Forbs	8.98	7.79	4.19	4.26	6.43	20.75	4.61
Total Live	36.39	40.24	54.72	34.95	67.17	67.31	60.69
Standing Dead	26.85	21.86	14.63	17.81	4.91	14.19	14.23
Litter	36.76	37.90	30.64	47.24	27.92	18.49	25.08
Total Dead	63.61	59.76	45.28	65.05	32.83	32.69	39.31

Pastures TOR	Pregrazing	Year1	Year 2	Year 3	Year 4	Year 5	Year 6
Domesticated	3.45	4.80	5.35	4.08	6.20	6.88	6.88
Cool Season	1.85	1.85	6.83	4.08	4.98	6.08	3.93
Warm Season	0.43	0.70	1.83	2.73	2.63	1.58	2.53
Native Grass	2.28	2.55	8.66	6.81	7.61	7.66	6.46
Sedges	7.63	7.75	10.83	10.75	11.05	12.70	9.55
Forbs	0.58	0.45	0.43	0.20	0.45	3.05	0.50
Total Live	13.93	15.55	25.25	21.83	25.30	30.28	23.38
Litter	85.90	84.43	74.75	78.18	74.70	69.73	76.63
Total	99.83	99.98	100.00	100.00	100.00	100.00	100.00

Table 8. Basal cover (%) for grazed silty native rangeland sites dominated by kentucky bluegrass.

 Table 9. Percentage composition (%) of basal cover for grazed silty native rangeland sites dominated by kentucky bluegrass.

Pastures TOR	Pregrazing	Year1	Year 2	Year 3	Year 4	Year 5	Year 6
Domesticated	24.77	30.87	21.19	18.69	24.51	22.72	29.43
Cool Season	13.28	11.90	27.05	18.69	19.68	20.08	16.81
Warm Season	3.09	4.50	7.25	12.51	10.40	5.22	10.82
Native Grass	16.37	16.40	34.30	31.20	30.08	25.30	27.63
Sedges	54.77	49.84	42.89	49.24	43.68	41.94	40.85
Forbs	4.16	2.89	1.70	0.92	1.78	10.07	2.14
Total Live	13.93	15.55	25.25	21.83	25.30	30.28	23.38
Litter	85.90	84.43	74.75	78.18	74.70	69.73	76.63

	Control Pasture kg/m <sup>3</sup>	Grazed Pastures kg/m <sup>3</sup>	% Difference
Pregrazing	52.23	77.99	49.32
Year 1	64.24x	83.28x	29.64
Year 2	77.82x	92.22x	18.50
Year 3	70.67y	122.61x	73.50
Year 4	82.88y	140.32x	69.31
Year 5	86.85y	183.00x	110.71
Year 6	130.56y	214.34x	64.17

Table 10. Rhizosphere weight (kg/m<sup>3</sup>) for the control pasture and grazed pastures during six years of twice-over rotation management.

Means in the same row and followed by the same letter (x, y) 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.

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# Degradation and Biological Restoration of Mixed Grass Prairie Ecosystems

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Strategies used to manage mixed grass prairie ecosystems that are antagonistic to grass plant and soil organism biological requirements cause diminished activation of the defoliation resistance mechanisms within grass plants, reduced productiveness of ecosystem biogeochemical processes, and decreased competitiveness of grass plant resource uptake processes resulting in ecosystem degradation. Symptoms of degraded ecosystem processes are visually expressed as changes in the vegetation composition. The severity of ecosystem degradation corresponds to the severity and duration of the management strategy antagonism. The functionality of ecosystem processes descend on a regressive continuum of deterioration from an extremely healthy ecosystem downwards to a disfunctional ecosystem with great reductions of native grasses, considerable decreases of desirable forbs, enormous increases of introduced domesticated grasses, remarkable increases of woody shrubs and trees, and excessive quantities of standing dead and litter.

Degraded mixed grass prairie ecosystem processes show symptoms of three major stages of vegetation deterioration: A) mixed grass prairie ecosystems with highly degraded processes at increasing intensities have plant communities dominated by native grasses at decreasing condition, B) mixed grass prairie ecosystems with severely degraded processes have plant communities dominated by invasive domesticated grasses, and C) mixed grass prairie ecosystems with extremely degraded processes have plant communities invaded by or dominated by woody shrubs. Biological restoration of the processes within mixed grass prairie ecosystems showing symptoms of degradation at the three major stages of vegetation deterioration is possible with implementation of biologically effective management plus the addition of specific management practices needed to correct the additional specific problems created by each of the increasing stages of deterioration.

Mixed grass prairie communities are complex ecosystems consisting of numerous

interactive biotic (living) and abiotic (nonliving) components. The biotic components are the plants, soil organisms, and large grazing graminivores that have biological and physiological requirements. The abiotic components include the essential major elements of carbon, hydrogen, and nitrogen in the presence of sunlight that have transformable characteristics through biogeochemical processes. Prairie ecosystems are functioning units of coacting biotic organisms interacting with the abiotic components and the environment. In order for all the ecosystem components to interact and function at potential levels, the defoliation resistance mechanisms within the grass plants, the ecosystem biogeochemical processes, and the grass plant resource uptake processes require activation through partial defoliation of 25% to 33% of the leaf material of grass lead tillers at phenological growth stages between the 3.5 new leaf stage and the flower stage by large grazing graminivores (Manske 2010a. 2010b). Partial defoliation by grazing graminivores managed by the biologically effective twice-over rotation strategy coordinates defoliation events with grass phenological growth stage, meets the nutritional requirements of the grazing animals, meets the biological requirements of the grass plants and the rhizosphere organisms, and activates the ecosystem processes to function at potential levels. Grazing animals are essential annual growing season components of mixed grass prairie ecosystems. If any of the numerous processes are not functioning at potential level, the ecosystem does not function at potential level (Manske and Schneider 2012a). When the ecosystem is not functioning at the potential level, the ecosystem is functioning at a degraded level.

#### **Degradation from Traditional Grazing Practices**

Ecosystem processes function at some degree less than potential level each growing season that grazing defoliation is managed with traditional practices that are based on grassland use as forage for livestock. Soon after the ecosystem processes fail to function properly, the belowground components begin to deteriorate. The native grass live root biomass decreases (Whitman 1974), the defoliation resistance mechanisms within grass plants diminish, the ecosystem biogeochemical processes decline, and the competitiveness of grass plant resource uptake deteriorates (Manske 2011b).

The reduction of live root surface area causes a decrease in active root length for interaction with symbiotic rhizosphere organisms and causes a decrease in absorption of water and nutrients from the soil. Reduction of active root biomass and diminishment of grass plant health vigor result in a loss of resource uptake efficiency and a suppression of the competitiveness of grass plants to take up mineral nitrogen, essential elements, and soil water (Kochy 1999, Kochy and Wilson 2000). The loss of active root length is a contributing factor in the reduction of rhizosphere biomass. The primary cause for the reduction in rhizosphere biomass is, however, the great reduction in the quantity of carbohydrates exuded from the grass roots into the rhizosphere zone. The antagonistic defoliation by grazing managed by traditional practices greatly reduces the quantity of short carbon chain energy exuded from the grass roots into the rhizosphere; this low amount of simple carbon compounds is not enough to sustain an adequate rhizosphere biomass. A small biomass of rhizosphere organisms mineralize small quantities of nitrogen and other essential elements (Coleman et al. 1983, Klein et al. 1988).

Rhizosphere organism biomass and activity are limited by access to simple carbon chain energy (Curl and Truelove 1986) because the microflora trophic levels lack chlorphyll and have low carbon (energy) content. Partial defoliation by large grazing graminivores of grass lead tillers at vegetative phenological growth stages is required to cause greater quantities of exudates containing simple carbon compounds to be released through the grass roots into the rhizosphere. Biomass and activity of the microorganisms increase with the increase in availability of energy from simple carbon compounds in the rhizosphere. The increase in rhizosphere organism biomass and activity causes an increase in mineralization of nitrogen and other essential elements. The severity of antagonism to ecosystem processes and the degree of resulting deficiencies in mineral nitrogen and other essential elements are not the same for all traditional grazing practices.

The traditional grazing management practice of 4.5 month seasonlong is grazed at moderate stocking rates on native grasslands between early June and mid October and is moderately antagonistic to mixed grass prairie ecosystems. After 20 years, the plant communities had been degraded 23% and were dominated by native grasses. The mean monthly grazing season live herbage biomass production was 1280 lbs/ac with 55.1% native grasses, 30.6% upland sedges, and 14.5% forbs. The rhizosphere volume was reduced to 67.6 ft  $^{3}$ /ac (29.8% of potential volume) and the available mineral nitrogen was reduced to 76.7 lbs/ac (Manske 2011a) (table 1).

The traditional grazing management practice of 6.0 month seasonlong is grazed at slightly heavier than moderate stocking rates on native grasslands between mid May and mid November and is highly antagonistic to mixed grass prairie ecosystems. After 20 years, the plant communities had been degraded 38% and were dominated by native grasses. The mean monthly grazing season live herbage biomass production was 906 lbs/ac with 53.8% native grasses, 27.1% upland sedges, and 19.2% forbs. The rhizosphere volume was reduced to 49.8 ft <sup>3</sup>/ac (21.9% of potential volume) and the available mineral nitrogen was reduced to 61.6 lbs/ac (Manske 2011a) (table 1).

The traditional grazing management practice of long duration 7.0 to 8.0 month seasonlong is grazed at moderate stocking rates on native grasslands starting in early May and continuing until late December or until inclement weather prevents late season grazing and is severely antagonistic to mixed grass prairie ecosystems. After 75 years, the plant communities had been degraded 57% and were dominated by native grasses; woody plant cover had invaded only 0.03% of the grazed area. The aboveground vegetation consisted of 23.9% standing dead and litter and 76.1% live herbage biomass. The peak growing season live herbage biomass was 1719 lbs/ac with 10.3% invasive domesticated grasses, 64.0% native grasses, 15.7% upland sedges, and 10.1% forbs. The rhizosphere biomass was reduced to 154.7 kg/m<sup>3</sup> (38.1% of potential weight) and the available mineral nitrogen was reduced to 42.4 lbs/ac (Manske and Schneider 2012a) (table 1).

The traditional grazing management practice of 4.0 month deferred, delays grazing until grass tillers are mature, is grazed at moderate to heavy stocking rates on native grasslands between mid July and mid November and is extremely antagonistic to mixed grass prairie ecosystems. After 50 years, the plant communities had been severely degraded 69%. The great loss of native grasses created large open spaces and the deferred grazed areas became dominated by Kentucky bluegrass, smooth bromegrass, and western snowberry. Woody shrub cover had invaded greater than 50% of the area. The aboveground vegetation consisted of 32.5% standing dead, 32.1% current year shrub growth, and 35.4% live herbage biomass. The peak growing season live herbage biomass was 891 lbs/ac with 20.7% invasive domesticated grasses, 25.5% native grasses, 25.2% upland sedges, and 28.6% forbs. The rhizosphere biomass was greatly reduced and the available mineral nitrogen was reduced to 31.2 lbs/ac (Manske 2007a, 2011a) (table 1).

The traditional grazing management practice of long duration repeat seasonal is grazed at heavier than moderate stocking rates on native grasslands starting as early in May as possible, before the grass lead tillers reach the 3.5 new leaf stage, continuing until inclement weather prevents late season grazing and is extremely antagonistic to mixed grass prairie ecosystems. After more than 30 years, the plant communities had been severely degraded. The great loss of native grasses created large open spaces and the heavily grazed long duration pastures became dominated by invasive Kentucky bluegrass and western snowberry. The mean herbage biomass was 802 lbs/ac. The rhizosphere biomass was greatly reduced and the available mineral nitrogen was low.

#### **Degradation from Nondefoliation**

Nondefoliation management by complete rest of mixed grass prairie ecosystems is not a revitalizing inactivity. Removing graminivores from grassland ecosystems to provide rest from grazing is a devitalizing activity that results in decreased rhizosphere organism biomass causing deficiencies in mineral nitrogen and other essential elements, and that results in decreased sunlight intensity and soil water causing deficiencies in fixed carbon and vital organic compounds.

An evolutionary survival mechanism of grass plants in response to partial defoliation and the loss of leaf area as forage to grazing graminivores is the production of double the quantity of leaf biomass than needed for normal plant growth and maintenance (Crider 1955, Coyne et al. 1995). This survival mechanism does not stop upon removal of grazing graminivores. Without grazing graminivores to remove half of the annual herbage production, the surplus leaf material accumulates rapidly and changes from an asset to a detriment. The accumulation of nondefoliated live and standing dead leaves of grasses reduce light penetration below native grass light saturation points (Peltzer and Kochy 2001). Native grasses have high light saturation points and require near full sunlight. Warm season grasses have higher light saturation points then 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 in low light conditions.

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

The rapidly accumulating quantities of standing dead biomass cannot make contact with the soil surface and decompose quickly through microbial activity. The standing dead biomass decreases slowly by leaching and weathering and builds up into a thick mulch layer. Thick mulch effectively blocks sunlight from reaching understory young grass leaves. Thick mulch insulates the soil from warm spring air temperatures preventing heating of cold soil that causes delays in plant and soil organism activity. Thick mulch ties up and holds organic nutrients above the soil surface preventing accession to the soil organic matter which limits nutrient cycling through biogeochemical processes increasing the deficiencies of essential elements. Thick mulch absorbs and holds precipitation for later evaporation preventing the water from infiltrating into the soil diminishing soil water to deficiency quantities (Wright and Bailey 1982, Manske 2000, 2011a). These undesirable modifications to the ecosystem cause decreases in soil microorganism biomass and activity resulting in further reductions in the rates of organic material decomposition (Anderson et al. 1981, Curl and Truelove 1986, Whipp 1990).

Grass plants developed several physiological, biological, and biogeochemical processes early during coevolution with graminivores in response to partial defoliation and the removal of leaf area (McNaughton 1979, 1983; Coleman et al. 1983; Briske 1991; Briske and Richards 1995; Manske 1999). This set of processes are collectively recognized as defoliation resistance mechanisms (Briske 1991, Briske and Richards 1995). Annual partial defoliation by grazing graminivores of grass lead tillers at growth stages between the 3.5 new leaf stage and the flower stage is required to activate the defoliation resistance mechanisms.

Native grass plants need the essential major elements of carbon, hydrogen, and nitrogen in the presence of sunlight for physiological growth processes to produce leaves, stems, roots, and secondary tillers (Manske 2011b). Removal of grazing graminivores from grasslands and the failure to activate the defoliation resistance mechanisms restricts grass plants use of important mechanisms and processes necessary for normal grass growth and development.

The carbon allocated for grass growth after the 3.5 new leaf stage does not come from stored material in the roots but is carbon recently fixed (Richards and Caldwell 1985, Coyne et al. 1995, Briske and Richards 1995). The carbon comes from atmospheric carbon dioxide which composes about 0.03% of the gasses in the atmosphere and exists at concentrations of around 370 to 385 mg/kg. Atmospheric carbon dioxide is not limiting on rangelands. The carbon dioxide is fixed with hydrogen from soil water during the process of photosynthesis which converts energy from sunlight into chemical energy and assimilates simple carbohydrates. However, when nondefoliated live and standing dead leaves of grasses reduce sunlight reaching understory grass leaves, photosynthetic rates are greatly reduced and available fixed carbon becomes deficient.

The hydrogen allocated for grass growth comes from soil water absorbed through the roots. Soil water is infiltrated precipitation. In western North Dakota, the perennial plant growing season months have a long-term periodicity rate of water deficiency conditions at 32.7%, for a mean of 2.0 months with water deficiency per growing season (Manske et al. 2010). The thick mulch that builds up on nondefoliation managed grasslands causes additional soil water problems. Thick mulch intercepts a portion of the precipitation inhibiting infiltration. The thicker the mulch, the greater the quantity of the precipitation absorbed. Absorption of the precipitation by the mulch causes a deficiency in soil water further inhibiting carbon assimilation.

The nitrogen allocated for grass growth can be mobilized from shoot and root tissue (Briske and Richards 1995) when the preferential source of mineral nitrogen recently converted from soil organic nitrogen by active rhizosphere organisms is low. 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). However, mixed grass prairie soils are not deficient of nitrogen. Most of the nitrogen is immobilized in the soil as organic nitrogen. Untilled grassland soils contain about 3 to 8 tons of organic nitrogen per acre. Soil organic nitrogen must be mineralized by rhizosphere organisms to become plant usable mineral nitrogen. The quantity of rhizosphere organisms is the limiting factor in grassland ecosystems low in mineral nitrogen. Biomass and activity of organisms in the rhizosphere are limited by access to energy from simple carbohydrates which can be exudated from grass lead tillers with partial defoliation by grazing graminivores when grass tillers are at vegetative growth stages after the 3.5 new leaf stage. Available mineral nitrogen becomes extremely deficient when grazing graminivores are removed from a grassland.

Light is radiant energy from the sun and is necessary for photosynthesis. Intensity of sunlight can be greatly reduced by shading from other plants. Nondefoliated live and standing dead leaves of grasses reduce light penetration to a similar degree as shrubs, even though shrub leaves are flat and wide and grass leaves are erect and linear (Kochy 1999). The light levels penetrating the leaf canopy can be about 20% of the light levels above the canopy (Peltzer and Kochy 2001).

Nondefoliation of native grass plants reduces exudation of short carbon chain energy into the rhizosphere to a low level of incidental leakage, reducing biomass and activity of soil organisms, decreasing mineralization processes that cause deficiencies in mineral nitrogen and other essential elements. Failure to remove the double produced leaf biomass annually causes shading that greatly reduces the sunlight intensity reaching native grass leaves, reducing photosynthetic rates that cause deficiencies in available fixed carbon. Accumulation of undecomposed grass leaf material modifies soil temperatures, causes deficiencies in soil water, and causes additional deficiencies in essential elements. The deficiencies of indispensable component resources of carbon, hydrogen, nitrogen, and other essential elements prevent grass plants from synthesizing sufficient quantities of vital carbohydrates, proteins, and nucleic acids. Without sufficient quantities of vital organic compounds native grass plants cannot maintain production of herbage biomass and tiller numbers (Langer 1972. Briske and Richards 1995).

Grasses with long shoots and stalk leaves provide low basal cover and low soil cover. Plant communities with increasing composition of grasses with long shoots and stalk leaves have low ground cover that results in great amounts of soil exposed to erosion and exposed to high levels of solar radiation that causes high rates of soil water transpiration. Grasses with short shoots and basal leaves provide high basal cover and high soil cover. Plant communities with good composition of grasses with short shoots and basal leaves have high ground cover near the soil surface which forms a closed community that protects the soil from erosion and solar radiation. Increases of canopy cover shading by taller live and standing dead leaves reduces the intensity of sunlight reaching the understory leaves of grasses with short shoots and basal leaves causing a disproportional mortality that creates large open spaces in the plant community.

Advanced degradation by antagonistic nondefoliation management of mixed grass prairie communities results in creation of numerous large bare spaces between native grass plants in the plant community. These open spaces, that lack competition from native grasses, are ideal habitat for growth of introduced cool season domesticated grasses like Kentucky bluegrass, and smooth bromegrass, and for growth of other opportunistic "weedy" plant species. These introduced plants have labile roots that break down easily making the nutrients contained in dead roots readily available to support continued growth and expansion of these nonnative plants without assistance from symbiotic rhizosphere organisms. The composition of plant species changes with decreases in the desirable species and increases in less desirable species, and later with increases in undesirable species. The change in plant composition from desirable to undesirable species is actually the symptom of ecosystem degradation; the fundamental degradation is the diminishment of defoliation resistance mechanisms within grass plants, the decrease of ecosystem biogeochemical processes, and the reduction of available mineral nitrogen below 100 lbs/ac. The degree of plant species change lags behind the degree of ecosystem biogeochemical degradation. Nondefoliation management can degrade mixed grass prairie more drastically than poor grazing management.

The nondefoliation management practice of complete rest from grazing removes large graminivores for longer than a decade and is extremely antagonistic to mesic mixed grass prairie ecosystems. After 13 years of nondefoliation, the plant communities at site #1 had been greatly degraded 87%. The 82.0% loss of native grasses with short shoots and basal leaves created large open spaces and the nongrazed areas at site #1 became dominated by invasive smooth bromegrass. The aboveground vegetation biomass consisted of 68.0% standing dead and litter and 32.0% live herbage. The mean nongrazed live herbage biomass was 1630 lbs/ac with 93.9% invasive domesticated grasses, 2.2% native grasses, 2.2% upland sedges, and 1.8% forbs. The rhizosphere biomass was reduced to 54.1 kg/m<sup>3</sup> (13.3% of potential weight) and the available mineral nitrogen was greatly reduced (Manske and Schneider 2012b) (table 1).

The nondefoliation management practice of complete rest from grazing removes large graminivores for longer than a decade and is extremely antagonistic to mesic mixed grass prairie ecosystems. After 13 years of nondefoliation, the plant communities at site #2 had been greatly degraded 81%. The 39.9% loss of native grasses with short shoots and basal leaves created large open spaces and the nongrazed areas at site #2 became dominated by invasive Kentucky bluegrass. The aboveground vegetation biomass consisted of 63.6% standing dead and litter and 36.4% live herbage. The mean nongrazed live herbage biomass was 1645 lbs/ac with 64.8% invasive domesticated grasses, 3.3% native grasses, 22.9% upland sedges, and 9.0% forbs. The rhizosphere biomass was reduced to 78.0  $kg/m^3$  (19.2% of potential weight) and the available mineral nitrogen was greatly reduced (Manske and Schneider 2012b) (table 1).

The long-term nondefoliation management practice of complete rest from grazing removes large graminivores for longer than multiple decades and is extremely antagonistic to xeric mixed grass prairie ecosystems. After 75 years of nondefoliation, the plant communities at long-term nongrazed areas had been degraded 61%. The 74.4% loss of native grasses with short shoots and basal leaves created large open spaces and the long-term nongrazed areas became dominated by invasive Kentucky bluegrass; woody plant cover had invaded 54.4% of the nongrazed area. The aboveground vegetation biomass consisted of 52.4% standing dead and litter and 47.6% live herbage. The mean nongrazed live herbage biomass was 1713 lbs/ac with 59.8% invasive domesticated grasses, 22.7% native grasses, 7.0% upland sedges, and 10.6% forbs. The rhizosphere biomass was reduced to 127.2 kg/m<sup>3</sup> (31.3% of potential weight) and the available mineral nitrogen was reduced to 39.5 lbs/ac (Manske and Schneider 2012a) (table 1).

Removal of cattle grazing from mixed grass prairie plant communities causes discontinuation of defoliation resistance mechanisms, degeneration of ecosystem biogeochemical processes, depletion of plant species composition with severe reductions of native grasses, excessive increases of standing dead and litter, extreme increases of introduced domesticated grasses, and remarkable increases of woody shrub and tree species.

A remarkably high quantity of woody shrub and tree species and an exceptionally great number of individual woody plants were able to develop on nongrazed mixed grass prairie because the competitive advantage of grasses for belowground resources was diminished in conjunction with the degradation of the native grass plant communities that resulted from removal of grazing defoliation by large graminivores. The existence of a shrub component in a grassland plant community is not an ecological beneficial relationship. Shrubs and grasses are adversarial inhibitive competitors. Grasses and shrubs compete for sunlight, mineral nitrogen, and soil water.

The degree of difference in competitive abilities between prairie grasses and shrubs on the mixed grass prairie was investigated during 1994 to 1998 at the University of Regina, Saskatchewan, with direction from Dr. S.D. Wilson. The grass growth form has competitive advantages over the shrub growth form. The per gram of biomass effects on resource use efficiency are smaller for shrub growth forms than for grass growth forms (Kochy and Wilson 2000). Shrubs must use a portion of the photosynthates produced in the leaves to build and maintain their unproductive woody stems; the result is a great reduction in resource uptake efficiency. Shrubs require 6 times greater aboveground biomass than the grasses before grass production is decreased as a result of competition from the shrubs (Kochy 1999). Grass aboveground biomass is primarily productive photosynthetic leaves; the result is a high resource uptake efficiency. Grasses have a 1.4 times greater per gram of biomass resource efficiency effect than shrubs (Kochy 1999). Grasses on prairie habitat attenuated more light, took up more mineral nitrogen, and took up more soil water per gram of biomass than did shrubs on brush habitat (Kochy 1999, Kochy and Wilson 2000). Because grasses have high root : shoot ratios and no woody stems to maintain; grasses are good competitors for belowground resources and superior competitors for mineral nitrogen. Shrub's taller growth form makes the plants superior competitors for aboveground resources (Kochy and Wilson 2000).

Competition between shrubs and grasses during early stages of shrub expansion into prairie habitat of healthy grasses is primarily for belowground resources of nutrients and soil water; under these initial conditions, grasses have the advantage and the shrubs are suppressed (Kochy and Wilson 2000, Peltzer and Kochy 2001). Competition from healthy grasses also reduces the growth rates of shrub rhizomes and causes high mortality rates of young suckers, preventing expansion into healthy grass communities (Li and Wilson 1998). Seedlings of grasses and shrubs are also unable to compete effectively for resources in healthy established grass plant communities and are suppressed (Peltzer and Kochy 2001). Successful competition of grasses for belowground resources prevents shrubs from becoming established in healthy grass communities. Shrubs can compete for some of the belowground resources only after the grass plants have been degraded by antagonistic management practices.

Following the reduction in grass plant resource competitiveness, establishment of shrubs can occur in degraded grasslands, which frequently starts as clumps (Li and Wilson 1998). The belowground resources previously used by the healthy robust grasses, but no longer consumed by the smaller, less vigorous degraded grasses, are taken up by the shrub plants resulting in proportional increases of biomass production (Kochy and Wilson 2000). After grass competition for belowground resources is reduced, shrub rhizome suckers can regain the faster growth rates and higher survival rates (Li and Wilson 1998) resulting in greater shrub stem densities. As shrub stem density increases, the competition shifts to primarily the aboveground resources of light; under these different degraded conditions, shrubs have the advantage and the grasses are strongly suppressed (Kochy and Wilson 2000).

The increase of woody shrubs and trees in grass communities would have traditionally been explained as a result of fire suppression (Humphrey 1962, Stoddart, Smith, and Box 1975, Wright and Bailey 1982). The greater increase of woody plants into the grass communities of the ungrazed prairie has not been the result of fire suppression. The increased woody plant infestation of the ungrazed prairie has been caused by the greatly reduced competitiveness of the ungrazed grasses for belowground resources of mineral nitrogen and soil water in conjunction with diminution of the defoliation resistance mechanisms and the degradation of the biogeochemical processes in the grass plant communities that resulted from the removal of partial defoliation by large grazing graminivores. The grazed grasses have maintained a

greater degree of competitiveness than the ungrazed grasses.

Many ecologists have observed that the repeated occurrence of fire was the force that prevented intrusion of shrubs and trees into grasslands (Weaver 1954, Humphrey 1962, Daubenmire 1974, Stoddart, Smith, and Box 1975, Wright and Bailey 1982), and, the obvious corollary, that suppression of fire has facilitated the invasion of shrubs and trees into grasslands (Humphrey 1962, Stoddart, Smith, and Box 1975, Wright and Bailey 1982, Manske et al. 2006a). However, the presence of fire does not prove that grasslands need or are caused by fire (Heady 1975). In the northern mixed grass prairie, fire cannot prevent the invasion of or cause the removal of shrubs and trees that reproduce by vegetative secondary suckers growing from crown (stem base) or rhizome buds (Wright and Bailey 1982, Manske et al. 2006a, Manske 2006b). Almost all deciduous woody plants growing in the Northern Plains grassland reproduce vegetatively.

The increase of undesirable introduced herbaceous grasses and forbs in grass communities has also often times been explained as a result of fire suppression (Wright and Bailey 1982). Kirsch and Kruse (1972) concluded that the cool season exotic grasses, Kentucky bluegrass and smooth bromegrass, have invaded much of the northern mixed grass prairie in the absence of fire. Seedlings of trees. shrubs, weedy forbs, and introduced grasses cannot become established in healthy functioning grassland ecosystems with grasses that have retained full resource uptake competitiveness (Peltzer and Kochy 2001). Establishment of intrusive seedlings can occur only after the grass communities have been degraded by antagonistic management practices. Furthermore, fire cannot prevent establishment and expansion of undesirable seedlings into degraded grass communities unless the fire frequency suppresses seed production of all invasive species in a region.

Repeated prescibed fire can change the percent composition of the aboveground vegetation biomass in degraded mixed grass prairie invaded by shrubs. The composition of introduced cool season grasses, early succession and weedy forbs, and shrub aerial stems decrease temporarily from four repeated every-other-year prescribed fires (Manske 2007a, 2011a). However, the fundamental problems of low native grass resource uptake competitiveness, diminished defoliation resistance mechanisms, and degraded biogeochemical processes remain in the grassland ecosystems following repeated fire events. None of the physiological and asexual processes of the defoliation resistance mechanisms within grass plants and none of the biogeochemical processes performed by symbiotic rhizosphere organisms within grassland ecosystems are activated by fire. Fire does not stimulate vegetative reproduction by tillering. Fire does not stimulate endomycorrhizal fungal colonization of perennial grass roots. Fire does not stimulate rhizosphere organism biomass and activity. Fire does not stimulate mineralization of soil organic nitrogen into mineral nitrogen (Manske 2007a, 2011a). Fire does not replace partial defoliation by grazing for management of healthy and productive rangeland ecosystems.

The invasion of shrubs, trees, and undesirable forbs and grasses into degraded grassland communities is not the result of fire suppression. Shrubs, trees, and introduced forbs and grasses can effectively compete for a portion of the belowground resources of nutrients and soil water after the competitiveness of the native grasses has been reduced as a result of degradation of the grass community caused by antagonistic management practices.

## **Restoration from Biologically Effective Defoliation**

Implementation of a biologically effective grazing management strategy that stimulates the defoliation resistance mechanisms will meet the biological and physiological requirements of the biotic components and will stimulate the biogeochemical processes that cycle the abiotic components (Manske 2011b). The three main defoliation resistance mechanisms are: compensatory internal physiological processes, internal vegetative reproduction of secondary tillers from axillary buds, and external symbiotic rhizosphere organism activity (McNaughton 1979, 1983; Coleman et al. 1983; Ingham et al. 1985; Mueller and Richards 1986; Richards et al. 1988; Briske 1991; Murphy and Briske 1992; Briske and Richards 1994, 1995; Manske 1999).

The defoliation resistance mechanisms developed early during the coevolution of grass plants and grazing graminivores (McNaughton 1979, 1983; Coleman et al. 1983; Briske 1991; Briske and Richards 1995; Manske 1999) and are a complex assemblage of biogeochemical and physiological processes that involve intricate interactions among rhizosphere microorganisms, grass plants, and large grazing graminivores. Activation of these mechanisms provides important biological and physiological processes permitting native grasses to produce greater herbage biomass and to increase basal cover; these mechanisms also enable grass plants to replace lost leaf material, to restore disrupted physiological processes, and to vegetatively reproduce secondary tillers from axillary buds after partial defoliation by grazing. The defoliation resistance mechanisms function at variable levels of activation depending on the quantity of available mineral nitrogen in grassland ecosystem soil. When mineral nitrogen is available at 100 lbs/ac or greater, the defoliation resistance mechanisms function at full activation. When mineral nitrogen is available at less than 100 lbs/ac, the defoliation resistance mechanisms function at levels less than full activation (Manske 2009). In addition, the water (precipitation) use efficiency processes decrease in grass plants growing in ecosystems with less than 100 lbs/ac available mineral nitrogen causing herbage biomass production to be reduced by 49.6% (Wight and Black 1972, 1979).

The quantity of available mineral nitrogen in grassland ecosystem soils is dependent on the rate of mineralization of soil organic nitrogen by rhizosphere organisms. The larger the rhizosphere volume and microorganism biomass, the greater the quantity of soil mineral nitrogen converted. Rhizosphere volume and microorganism biomass are limited by access to simple carbohydrates (Curl and Truelove 1986). Healthy grass plants capture and fix carbon during photosynthesis and produce carbohydrates in quantities greater than the amount needed for tiller growth and development (Coyne et al. 1995). Partial defoliation of grass tillers at vegetative phenological growth stages by large grazing graminivores causes greater quantities of exudates containing simple carbohydrates to be released from the grass tillers through the roots into the rhizosphere (Hamilton and Frank 2001). With the increase in availability of 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 converting 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). Mineral nitrogen available in quantities of 100 lbs/ac or greater allows defoliated grass tillers full activation of the defoliation resistance mechanisms (Manske 2009). Full activation of the compensatory internal physiological processes within grass plants accelerates growth rates

of replacement leaves and shoots, increases photosynthetic capacity of remaining mature leaves, increases allocation of carbon and nitrogen, 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 internal processes of vegetative reproduction increases secondary tiller development from axillary buds and increases initiated tiller density during the grazing season. Full activation of the external symbiotic rhizosphere organism activity increases mineralization of mineral nitrogen, increases ecosystem biogeochemical cycling of essential elements, and improves belowground resource uptake competitiveness (Wight and Black 1972, 1979; McNaughton 1979, 1983; Coleman et al. 1983; Ingham et al. 1985: Mueller and Richards 1986: Richards et al. 1988; Briske 1991; Murphy and Briske 1992; Briske and Richards 1994, 1995; Manske 1999, 2011b; Kochy and Wilson 2000). Restoration of plant community vegetation composition lags behind restoration of ecosystem processes.

Restoration of degraded mixed grass prairie ecosystems dominated by native grasses can be accomplished by implementation of the twice-over rotation system which is a biologically effective grazing management strategy that coordinates defoliation events with grass phenological growth stages, meets the biological requirements of the plants and soil organisms, and meets the nutritional requirements of the grazing animals. After 20 years, the plant communities had been restored 100% with fully functioning defoliation resistance mechanisms, ecosystem biogeochemical processes, and resource uptake competitiveness. The mean monthly grazing season live herbage biomass production was 1794 lbs/ac with 74.7% native grasses, 14.2% upland sedges, and 11.1% forbs. The rhizosphere volume was increased to 227.1 ft <sup>3</sup>/ac and the biomass increased to 406.4 kg/m<sup>3</sup> (100.0% of potential volume and biomass) and the available mineral nitrogen was increased to 177.8 lbs/ac (Manske 2007b, 2011a) (table 1).

Restoration of degraded mesic mixed grass prairie ecosystems at site #1 dominated by smooth bromegrass was attempted by implementation of the twice-over rotation system. Before the grazing treatment started, the aboveground vegetation biomass consisted of 68.0% standing dead and litter and 32.0% live herbage. The mean live herbage biomass was 1630 lbs/ac with 93.9% invasive domesticated grasses, 2.2% upland sedges, and 1.8% forbs. The rhizosphere biomass was reduced to 54.1 kg/m<sup>3</sup> (13.3% of potential weight) and the available mineral nitrogen was greatly reduced. After 6 years of twice-over rotation grazing management, the plant communities at site #1 had been restored about 41%. The aboveground vegetation biomass consisted of 35.6% standing dead and litter and 64.4% live herbage. The mean live herbage biomass was 2447 lbs/ac with 89.6% domesticated grasses, 3.5% native grasses, 2.0% upland sedges, and 4.9% forbs. The rhizosphere biomass had increased to 167.1 kg/m<sup>3</sup> (41.1% of potential weight) and the available mineral nitrogen had increased to 59.9 lbs/ac (Manske and Schneider 2012b) (table 1).

Restoration of degraded mesic mixed grass prairie ecosystems at site #2 dominated by Kentucky bluegrass was attempted by implementation of the twice-over rotation system. Before the grazing treatment started, the aboveground vegetation biomass consisted of 63.6% standing dead and litter and 36.4% live herbage. The mean live herbage biomass was 1645 lbs/ac with 64.8% invasive domesticated grasses, 3.3% native grasses, 22.9% upland sedges, and 9.0% forbs. The rhizosphere biomass had been reduced to 78.0 kg/m<sup>3</sup> (19.2% of potential weight) and the available mineral nitrogen was greatly reduced. After 6 years of twice-over rotation grazing management, the plant communities at site #2 had been restored about 53%. The aboveground vegetation biomass consisted of 39.3% standing dead and litter and 60.7% live herbage. The mean live herbage biomass was 2174 lbs/ac with 58.0% domesticated grasses, 26.1% native grasses, 11.3% upland sedges, and 4.6% forbs. The rhizosphere biomass had increased to 214.3 kg/m<sup>3</sup> (52.7% of potential weight) and the available mineral nitrogen had increased to 56.7 lbs/ac (Manske and Schneider 2012b) (table 1).

Degraded mixed grass prairie ecosystems that were subsequently invaded by introduced domesticated grasses have two additional major problems, increased shading and increased mulch biomass, that must be overcome by management in order to successfully restore the plant communities. Increased shading reduces rates of photosynthesis, reducing fixed carbohydrates supply, causing reduced growth rates of leaves and roots in understory native grasses. Increased mulch biomass reduces precipitation infiltration and early season soil temperatures causing reduced soil organism biomass and activity, slowed nutrient cycles, and decreased biogeochemical processes resulting in greatly reduced native plant density and in creating large open spaces for invasion of undesirable introduced domesticated grasses (Manske and Schneider 2012b).

The degraded mesic mixed grass prairie communities on sites #1 and #2 that were subsequently invaded by smooth bromegrass and Kentucky bluegrass, respectively, were not fully restored during the 6 years of management with the twice-over rotation system because the defoliation resistance mechanisms were not sufficiently activated as a result of insufficient quantities of fixed carbon and mineral nitrogen available to the understory native grass plants. The native grasses on sites #1 and #2 were inhibited by the severe deficiency of sunlight reaching the understory leaves caused by shading from accumulated domesticated grass live and standing dead leaves. The stocking rates were high enough only during 2 of the 6 years to remove sufficient quantities of domesticated grass live and standing dead leaves and reduce the detrimental shading effect. The low sunlight intensity caused reduced photosynthetic rates that resulted in a deficiency of available fixed carbon. A deficiency of available soil mineral nitrogen was caused by low rhizosphere biomass because of low carbon exudates released through the grass roots. The deficiency of sunlight attenuated by native grass leaves caused by shading must be corrected through grazing defoliation with relatively high stocking rates, at 85% to 100% of assessed levels, managed by the twice-over rotation strategy before degraded mixed grass prairie communities that were subsequently invaded by domesticated grasses can be fully restored back to functional native grass ecosystem (Manske and Schneider 2012b).

The amount of woody shrubs and trees growing on the northern mixed grass prairie prior to European settlement is not known quantitatively but appears to have been low. From the early aerial photographic record for parts of the Northern Plains, Smith (1988) reported that during the mid to late 1930's only about 5% of the upland landscape area was occupied by shrubs. Traditional management practices that have deferred the start of grazing until after mid July, grazed during early or late season (before June or after mid October), or grazed at heavy grass utilization stocking rates caused mixed grass prairie ecosystem degradation severe enough to end suppression of shrub growth and provided enough access to belowground resources to facilitate shrub invasion and expansion (Manske 2007a, 2011a). Long-term nongrazing has also caused severe degradation of mixed grass prairie ecosystems that has resulted in shrub invasion and expansion (Manske

and Schneider 2012a). Antagonistic management practices have caused the abundant shrub cover greater than 5% on the regions upland landscapes.

The key to invasive woody shrub and tree and herbaceous plant control in grassland communities is to regain the competitive advantage of the grasses by restoration of the mechanisms and processes in the mixed grass prairie ecosystem that results from biologically effective partial defoliation by large grazing graminivores. Biological management of invaded shrubs with the twice-over rotation strategy improves the health and competitive abilities of the native grass plants and enhances the biogeochemical processes within the ecosystem (Manske 2011a, 2011b). With increasing competitiveness of native grasses for belowground resources of soil water and nutrients, the shrubs are detrimentally effected; the quantity of vegetative buds with activated meristematic tissue decreases, the growth rate of developing rhizome suckers decreases, and the mortality rate of young sucker stems increases. The effect of resource competition from healthy grasses causes the shrubs to decrease in density of rhizome and crown suckers, young stems, and total aerial stems, and causes a decrease in the canopy cover of shrub colonies. Unfortunately, this biological shrub control process is relatively slow and requires two to three decades to cause substantial shrub reductions.

Most deciduous shrubs and trees that grow in the Northern Plains, with the exception of big sagebrush, have the ability to reproduce vegetatively. Coniferous shrubs and trees do not reproduce vegetatively. Aerial stems of deciduous and coniferous shrubs are relatively easy to kill to the ground with a single treatment of prescribed burning, mechanical mowing, or chemical herbicide application. Successful deciduous shrub control management requires termination of the vegetative regenerative capabilities of the rhizomes and the stem bases on the crown.

Fire has been an environmental factor on mixed grass prairie as lightning-set fires and Indianset fires. Spring burns result in great quantities of shrub sucker stems, and lost carbohydrate stores can be completely replenished by the new plant material in one growing season. August fires remove all or most of the top growth of shrubs and result in fewer sucker shoots the following year than spring burns. When the soil is not dry, prescribed burns during August cause the least damage to native cool season and warm season grasses and perennial forbs. The belowground rhizomes and crown stem bases are usually not damaged by fire because of the protection provided by soil. Some of the buds on the belowground parts develop into new aerial sucker stems. Prescribed burning alone will not remove deciduous shrubs that can reproduce vegetatively from the northern mixed grass prairie (Manske et al. 2006a, Manske 2007a, 2011a).

Mechanical mowing treatments can effectively reduce shrub stem densities if the stored nonstructural carbohydrate energy is depleted. Single annual mowing treatments do remove existing aerial stems, temporarily eliminating the competition for sunlight to the understory plants, but do not reduce shrub stem numbers because the regrowth of sucker stems can replenish the carbohydrate reserves during one growing season. Double mowing per growing season can reduce shrub stem numbers when the two mowing periods match shrub growth stages and cause carbohydrate depletion. The first mowing period should be conducted when the shrubs are at seasonal low carbohydrate reserve, which occurs between the end of rapid spring growth and the start of flowering; for many shrubs this first mow period occurs during the last week in May through the third week in June. The resulting new sucker shoot growth should continue to deplete carbohydrate reserves for nearly six weeks. At that time, these new suckers require mowing to reduce the leaf area that would replenish the carbohydrate stores. The second mowing period should be conducted when the new suckers have completed the rapid growth stage; for many shrubs this second mow period occurs during late July through August. This second mowing also causes a substantial amount of winter injury to the resulting late-season lateral bud sprouts on the stem bases. The mowing height in grazed pastures during both mowing periods should be about 8 to 9 inches above the ground. Stems cut short are rigid and can puncture the sole of the hoofs of cattle walking through mowed areas causing an injury open to infection. Double mowing treatments will need to be repeated two or more seasons until the carbohydrate reserves are fully depleted (Manske et al. 2006a).

Chemical management control of shrubs requires termination of the regenerative capabilities of the rhizomes and the crown stem bases by disabling the meristematic tissue. Foliage-active herbicides must enter the leaf tissue through the stomata openings or penetrate the outer cuticle layer, be absorbed through leaf tissue by diffusion, moved to the vascular system within the leaf, and be translocated from the leaves downward through the phloem vascular system to the metabolically active sites of the crowns and rhizomes. During rapid stem elongation, nonstructural carbohydrates move from the storage sites in the rhizomes and the crowns upward through the phloem vascular system to the active growing points of the stem preventing downward movement of herbicides. As leaf area increases and carbohydrate production by photosynthesis exceeds the demands for growth, the surplus carbohydrates are moved downward through the phloem for storage in the rhizomes and crowns (Coyne et al. 1995); downward translocation of herbicides could occur at that time if the herbicides were in the leaf tissue. However, as leaves mature, they develop a thick cuticle layer and dense cell walls reducing herbicide penetration and absorption. Most shrubs have a brief vulnerable stage when the herbicide plus surfactant penetration into leaf tissue is decreasing and herbicide translocation downward is increasing. For many shrubs this short period occurs from about 10 June until 20 June.

Soil-active herbicides, like tebuthiuron (Spike 20P), have a relatively wide window of opportunity for treatment and require only that application be ahead of a rainy period. The herbicide moves into the roots anytime the roots are absorbing water. Movement upward in the xylem vascular system is not as complex as movement within the phloem system. Plants have few resistance mechanisms to restrict activity of soil-applied herbicides. Usually low rates are quite effective (Manske et al. 2006a, Manske 2006b).

Restoration of degraded mixed grass prairie ecosystems that have been subsequently invaded by woody shrubs requires regaining the competitive advantage of fully activated physiological and biogeochemical processes of healthy native grass plants through biologically effective management of grazing graminivores and requires termination of the vegetative regenerative capabilities of the shrubs rhizomes and stem bases through combinations of biological, prescribed burning, mechanical mowing, and chemical herbicide management practices.

#### Recapitulation

Mixed grass prairie ecosystems are complex and consist of numerous biotic and abiotic components that function together through three primary processes; the defoliation resistance mechanisms, the ecosystem biogeochemical processes; and the plant resource uptake processes. When these three primary processes do not function at potential level, the ecosystem does not function at potential level. Partial defoliation by grazing graminivores is required to trigger activation of the primary processes and the quantity of available soil mineral nitrogen and the quantity of available fixed carbon affect the level at which the primary processes function. Fully functioning processes require mineral nitrogen to be available at 100 lbs/ac or greater and require the quantity of carbon fixed through photosynthesis from 75% to 67% of the leaf area of grass lead tillers prior to peak live herbage biomass and from 50% of the leaf area post peak biomass.

The primary processes function at less than potential levels when soil mineral nitrogen and fixed carbon are available at quantities less than the threshold amounts and the processes are not activated when grazing graminivores do not remove 25% to 33% of lead tiller leaf area during vegetative growth stages between the 3.5 new leaf stage and the flower stage. When the defoliation resistance mechanisms, the biogeochemical processes, and the resource uptake processes function at degraded levels, the mixed grass prairie ecosystems are degraded to the levels at which the primary processes are functioning.

A reduction in the condition of the aboveground vegetation is a visable symptom that the primary processes are functioning at degraded levels. The symptom of descending quality of the vegetation from desirable native grasses, to less desirable domesticated grasses, to undesirable shrubs indicates the increasing severity in the deterioration of the primary ecosystem processes. The symptoms of descending changes in the vegetation lag behind the descending changes in the functionality of the primary processes.

Restoration of degraded mixed grass prairie ecosystems requires returning the three primary ecosystem processes back to functioning at potential levels. The rhizosphere organism biomass must be improved initially to increase mineralization of nitrogen and other essential elements. Rhizosphere organisms are limited by access to energy in the form of short carbon chains. Exudation of short carbon chain energy can be released from the grass lead tillers through the roots into the rhizosphere by removal of 25% to 33% of the aboveground leaf biomass by large grazing graminivores when the lead tillers are between the 3.5 new leaf stage and the flower stage. As a result of this successful initial recovery of the biogeochemical processes, the amounts of available mineral nitrogen, essential elements, fixed carbon, and soil water start increasing from deficient quantities towards functional quantities. With the increasing component resources, native grass plants are able to synthesize increasing

quantities of carbohydrates, proteins, and nucleic acids. Activation of the defoliation resistance mechanisms provides important biological and physiological processes permitting native grasses to use the vital organic compounds in increasing quantities for the production of herbage biomass and basal cover. With implementation of the belowground resource uptake of soil water, mineral nitrogen, and other essential elements, the native grasses become more competitive. The symptoms of improving changes in the vegetation lag behind the improving changes in the functionality of the primary processes.

Restoration of degraded mixed grass prairie ecosystems that were subsequently invaded by introduced domesticated grasses requires reduction of two additional problems, the increased shading from domesticated grass live and standing dead leaves and the increased mulch biomass. Both problems are corrected by stocking rates at 85% to 100% of assessed levels. The high stocking rates removes sufficient quantities of domesticated grass leaf material to reduce the shading effect of low sunlight intensity and the weight of the greater number of animals pushes the mulch biomass into contact with the soil where microbial decomposition can take place.

Restoration of degraded mixed grass prairie ecosystems that were subsequently invaded by woody shrubs requires termination of the vegetative regenerative capabilities of the shrubs rhizomes and stem bases through combinations of biological, prescribed burning, mechanical mowing, and chemical herbicide management practices.

Restoration of degraded mixed grass prairie ecosystems requires annual partial defoliation by grazing graminivores managed by the biologically effective twice-over rotation strategy that coordinates defoliation events with grass phenological growth stages. The twice-over strategy activates the defoliation resistance mechanisms, the biogeochemical processes, and the resource uptake processes, and then maintains their functionality at potential levels. The twice-over strategy also meets the biological requirements of the grass plants and the rhizosphere organisms, and meets the nutritional requirements of the grazing animals. Biologically effective management of restored mixed grass prairie ecosystems sustains healthy renewable natural resources that simultaneously provide greater forage for livestock, better habitat for wildlife, more aesthetic prairie ecosystems for recreation and sightseeing, and increased food and fiber for people.

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Management Treatment		Mineral Nitrogen lbs/ac	Rhizosphere Volume ft <sup>3</sup> /ac	Rhizosphere Biomass kg/m <sup>3</sup>	
Degradation from Traditional Grazing Practices					
4.5 m Seasonlong	20 yr	76.7	67.6		
6.0 m Seasonlong	20 yr	61.6	49.8		
7.0 m Seasonlong	75 yr	42.4		154.7	
Deferred Grazed	50 yr	31.2		-	
Degradation from Nondefoliation					
Nongrazed Site #1	13 yr	-		54.1	
Nongrazed Site #2	13 yr	-		78.0	
Nongrazed Long-term	75 yr	39.5		127.2	
Restoration from Biologically Effective Defoliation					
Restored Ecosystem	20 yr	177.8	227.1	406.4	
Restored Site #1	6 yr	59.9		167.1	
Restored Site #2	6 yr	56.7		214.3	

Table 1.	Ineral nitrogen and rhizosphere volume or biomass for degraded and restored mixed grass prairie
	cosystems.

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