THE INTERACTIONS OF EARLY-FALL PRESCRIBED BURNING, DIFFERENT CUTTING TECHNIQUES AND WHITE-TAILED DEER BROWSING ON BUR OAK REGENERATION IN EASTERN NORTH DAKOTA: PHASE II

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Title

The Interactions of Early-Fall Prescribed Burning, Different Cutting

Techniques and White-tailed Deer Browsing on Bur Oak Regeneration in

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The Supervisory Committee certifies that this ***disquisition*** complies with North Dakota State University's regulations and meets the accepted standards for the degree of

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ABSTRACT

Murdoff, Marc Thomas, M.S., Program of Range Science, College of Agriculture, Natural Resources, and Food Systems, North Dakota State University, April 2010. The Interactions of Early-Fall Prescribed Burning, Different Cutting Techniques and White-tailed Deer Browsing on Bur Oak Regeneration in Eastern North Dakota: Phase II. Major Professor: Dr. Kevin K. Sedivec.

This study was developed to determine the interaction of white-tailed deer browsing with effects of clear and selective cutting, and prescribed burning on bur oak (*Quercus macrocarpa*) regeneration in northeastern North Dakota. The study was conducted on Camp Grafton North (CGN) near Devils Lake, North Dakota, using four treatments: 1) dormant season clear-cut of all trees and shrubs (CC), 2) growing season selective cut of all trees and shrubs except bur oak (SC), 3) early fall prescribed burn (PB) and 4) non-manipulated control (CO). The study consisted of four blocks (replicates) using a randomized complete block design. The interaction of white-tailed deer (*Odocoileus virginianus*) browsing was determined using a split-plot design, creating browsed and non-browsed plots. Bur oak seedling, sprout, sapling, and mature tree production was measured pre-treatment in 2006 and post-treatment in 2007, 2008 and 2009 on two 25 m transects per plot. Herbaceous vegetation was measured using these two transects. An aerial survey conducted 12 March 2007 reported 45 deer per km^2 on CGN. Bur oak seedling density increased ($P \leq 0.05$) 36 Months After Treatment (MAFT). Bur oak saplings decreased ($P \leq 0.05$) at 36 MAFT. Bur oak sprouts were greatest ($P \leq 0.05$) on the CC treatment, while the CO, PB, and SC did not differ ($P > 0.05$). Deer browsing reduced bur oak sprout height, irrelevant of treatment. Clear-cutting increased bur oak sprouts 36 MAFT. Selective cutting to retain bur oak trees did not enhance seedling or sprout development compared to the control.

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INTRODUCTION

When people think of North Dakota their first impression is open plains, farming, ranching and hunting. They do not envision a state with trees, let alone forests. Based on the latest gap analysis (geographic analysis of land use and wildlife habitat), woodland plant communities encompass 4284km^2 (2.3 %) of the total land ground cover in North Dakota (Strong 2005). In North Dakota, the Bur Oak Cover Type accounts for 19 percent of forested land (Curtis 1959, Deitschmann 1965, Kline and Cottman 1979, Grimm 1984). These woodlands can provide shade, escape from wind and insects, and increased quality and quantity of forage to livestock and wildlife (Girard and Bjugstad 1984, Clark 2006).

Gilbert C. Grafton North Dakota Army National Guard Training Base - North Unit (CGN) is located in the Transitional Grasslands and is one of the few areas considered forested in North Dakota. Many of these communities are becoming decadent, with trees dying and regeneration slow or minimal (Clark 2006). A pre-study survey of CGN revealed most of the bur oak (*Quercus macrocarpa*) trees were mature (tree height greater than 300cm), with only a small number (2.5 trees/ha) of bur oak trees between 30-150cm in height. On average, fewer than 15 seedlings (< 30cm in height) per hectare were found on CGN in the pre-study survey. Brudvig and Asbjornsen (2008) reported that oaks in the sapling stage are rarely found in mature bur oak communities.

Camp Grafton North's forested area is classified as a bur oak/green ash (*Fraxinus pennsylvanica*) forest community intermixed with boxelder (*Acer negundo*), hackberry (*Celtis occidentalis*), American elm (*Ulmus americana*), cottonwood (*Populus deltoides*), and quaking aspen (*P. tremuloides*) (Barker et al. 2000). The understory shrub community consists of chokecherry (*Prunus virginiana*), juneberry (*Amelanchier alnifolia*) and western snowberry (*Symphoricarpos occidentalis*) (Barker et al. 2000). This bur oak/green ash plant community was classified as mature with minimal regeneration.

Bur oak communities provide excellent wintering habitat for deer (Severson and Kranz 1978). These ungulates have been shown to limit oak regeneration due to highly selective browsing of seedlings and saplings (Rooney and Waller 2003). Due to the high percentage of bur oak trees on CGN, it is an attractive habitat for white-tailed deer (*Odocoileus virginianus*). White-tailed deer have become very abundant in this area and are the primary browsers on bur oak seedlings and saplings. The first objective of this project was to study the impacts of prescribed burning, clear-cutting, and selective cutting on a bur oak community. The second objective of this study was to determine the impacts of white-tailed deer browsing on seedling recruitment and sapling development on each management treatment. This is the second phase of a four-year study that originated with Hanson (2009).

LITERATURE REVIEW

Bur Oak Characteristics

Bur oak, known as mossycup oak, mossy-overcup oak, and scrub oak, has the largest acorns of all native oaks and is very drought resistant (Johnson 1990). Bur oak is a member of the white oaks (Lepidobalanus) (Maze 1968). This subgenus hybridizes easily with numerous species including white oak (*Quercus alba*), swamp white oak (*Q. bicolor*), overcup oak (*Q. lyrata*), swamp chestnut oak (*Q. michauxii*), chinkapin oak (*Q. muehlenbergii*), post oak (*Q. stellata*), live oak (*Q. virginiana*), and Gambel oak (*Q. gambelii*). The Gambel oak (western species) hybrid is unusual since the two species ranges do not overlap anymore (Tester 1965, Maze 1968, Hardin 1975, Little 1979). Bur oaks common name comes from the cap-covered acorn (USDA-NRCS 2009a).

Distribution

Bur oak is widely distributed throughout much of northcentral United States and eastern Great Plains (Wasser 1982) (Figure 1). It occurs from southern New Brunswick and New England westward to the Dakotas, southeastern Montana, south to Tennessee, Arkansas, central prairies of Texas – with rare outliers in Louisiana, Mississippi, and Alabama (USDA-NRCS 2009a). A northern form of bur oak (*Quercus macrocarpa* var. *olivaeformis*) has been recognized and largely restricted to Iowa, Minnesota, South Dakota and North Dakota (Deitschmann 1965, Great Plains Flora Association 1986).

Bur oak is one of the most drought resistant of the North American oaks (Deitschmann 1965). In the northwestern part of its range, the average annual precipitation is 380mm. Average minimum temperatures is 4° C and a growing season lasting 100 days. In the south region, average annual precipitation exceeds 1270mm, with minimum temperatures -7° C and a growing season of 260 days. The best development of bur oak occurs in southern Illinois and Indiana, where average annual precipitation is 1140mm, minimum temperature -29° C and growing season of 190 days (Deitschmann 1965).

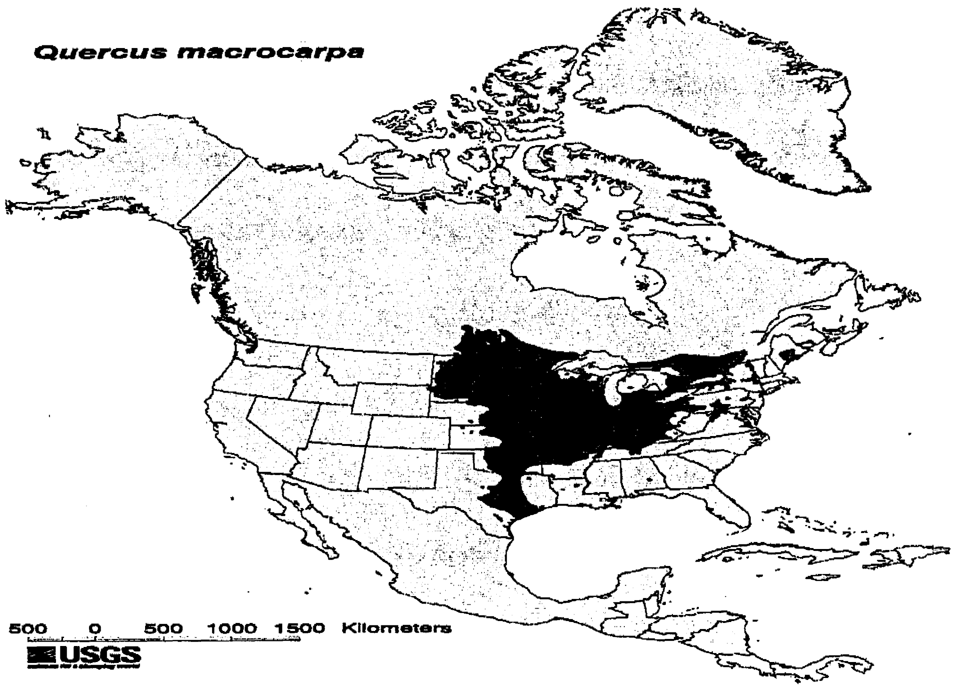

Figure 1. Bur oak range in North America (Image from USGS website, 2011).

Soil

Bur oak grows on a wide range of soil types. It is commonly found on medium to somewhat coarsely-textured soils and rarely on clays (Wasser 1982), and an optimum soil depth greater than 51cm (Dittberner and Olson 1983). Bur oak is well adapted to acidic soils (>4.0 pH) (Wasser 1982). Bur oak is often found on calcareous soils when growing in upland communities (Deitschmann 1965).

Bur oak is found on droughty sandy plains, black prairie loams, and loamy slopes of south and west exposure in the prairie region of the Midwest. At the western edge of its range in eastern Kansas, it is more abundant on the moist, north-facing slopes (Birdsell and Hamrick 1978). Predominant soil orders for bur oak are alfisols in the central and southern region of the United States, and mollisols and spodosols in the western and northern regions (Johnson 1990).

Bur oak is a major component of the stream corridor of the Missouri River in North Dakota (Johnson et al. 1976). It may dominate old stands on high terraces near the edge of the flood plain. Along adjacent draws and upper slopes, it becomes the first tree established along prairie edges (Johnson 1990).

Tree Characteristics

Bur oak is a spreading, deciduous, large shrub to large tree (Maze 1968, Wasser 1982). It typically becomes a large tree reaching 31m in height or greater and 0.9 to 1.2m in diameter (Wasser 1982). On poorly developed soils it often forms dense thickets of low, scrubby shrubs reaching heights of 5 to 10dm. Twigs are thick and corky (Wasser 1982, Welsh et al. 1987). Branches are low and stout, crown generally open and deeply divided into nine rounded lobes (Great Plains Flora Association 1986). Roots have great tensile strength and are well-branched and widely spreading (Weaver and Kramer 1932).

Bur oak is slow growing (Deitschmann 1965) and long lived, sometimes reaching 200 to 300 years old, with rare individuals reaching ages of 1000 years (Weaver and Kramer 1932, Stout 1944, USDA-NRCS 2009a).

Average annual height growth ranged from 0.09 to 0.52m and diameter growth from less than 2.5 to 6.4mm in 12 to 16 year-old plantations on Iowa upland sites. In shelterbelts of the northern Great Plains, annual height growth of about 0.3m was reported for trees kept under clean cultivation (Johnson 1990). Bur oak is relatively shade intolerant

(Deitschmann 1965). Seedlings can tolerate some shade, but eventually need full sunlight to grow (Gildar 2002).

Oaks have the capacity to die back and resprout. The aboveground part of the plant will typically die back to the root collar every 3–10 years (Merz and Boyce 1956, Tryon and Powell 1984, North Central Forest Experiment Station, Columbia, MO, unpublished data), depending on environmental conditions. The root collar has a large number of dormant buds, some of which activate upon loss of the old stem. This cycle, which can be repeated several times during the lifespan of a seedling, leads to the development of a large root system (Merz and Boyce 1956, Tryon and Powell 1984, North Central Forest Experiment Station, Columbia, MO, unpublished data).

Plant Community Association

Bur oak is an important dominant tree in many plant communities (Tirmenstein 1988) and can be associated with many other trees due to its tolerance to a wide range of soil and moisture conditions (Eyre 1980). Bur oak is found in prairies, valley floors and upland forests (USDA-NRCS 2009a). Bur oak grows quickly on moist, rich bottomlands but is relatively intolerant of flooding during the growing season (Daubenmire 1936, Severson and Boldt 1977) and is considered a pioneer or early seral species at prairie edges. However, bur oak savannas have declined due to grazing and fire suppression (Daubenmire 1936, Severson and Boldt 1977, USDA-NRCS 2009a).

In pure or nearly pure stands, it forms the forest cover type Bur Oak (Society of American Foresters Type 42, eastern forests; Type 236, western forests) (Eyre 1980). Bur oak is also an important associate in six other forest cover types: Northern Pin Oak (Type 14), Aspen (Type 16), Black Ash-American Elm-Red Maple (Type 39), White Oak (Type

53), Pin Oak-Sweetgum (Type 65), and Hawthorn (Type 109). Species commonly associated with bur oak include boxelder, black ash (*Fraxinus nigra*), white ash (*F. americana*), red maple (*Acer rubrum*), shellbark and bitternut hickories (*Carya laciniata*, *C. cordiformis*), American elm, hackberries (*Celtis* spp.), eastern cottonwood, black locust (*Robinia pseudoacacia*), basswood (*Tilia americana*), northern red oak, northern pin oak, white oak and swamp white oak (Stout 1944, Severson and Boldt 1977, Wasser 1982, Rothenberger 1985). On drier sites in the northwestern part of its range, it grows in mixed stands of American elm, green ash, bitternut hickory, white oak, and sometimes as nearly pure bur oak stands. Bur oak is the major tree of oak savannas ("oak openings") in the prairie-forest transition zones in Wisconsin, Minnesota, Iowa, and Illinois.

<u>Bur Oak Reproduction</u>

Bur oak reproduces both sexually and vegetatively (Reichman 1987, Gildar 2002). Trees are wind pollinated (Reichman 1987). Bur oak is monoecious with male and female flowers in unconnected catkins borne on the existing year's branchlets. It flowers shortly after the leaves appear, early April in the southern region to mid June in the northern regions of its range (Deitschmann 1965). In North Dakota, trees flower from May to June (Fowells 1965, Gildar 2002). Acorns mature August to November (Wasser 1982, Gildar 2002). A majority of seed dispersal occurs from August to September (Williams and Hanks 1976, Gildar 2002).

Germination is hypogeal (Olson 1974). Germination happens after seed fall, but acorns of northern trees may remain inactive throughout the winter and germinate the following spring (Deitschmann 1965). The northern form of bur oak acorns usually germinate in the spring (Fowells 1965, Olson 1974). Bur oak trees up to 400 years old can continue to produce seeds, longer than any other American oak. Minimum seed-bearing age is about 35 years while optimum is 75 to 150 years (Deitschmann 1965, Fowells 1965, Olson 1974, Gildar 2002, USDA-NRCS 2009a). Bur oak germination is best when herbaceous litter is removed (Fowells 1965). Mineral soils warm more quickly in the spring than soils covered by litter, which improves bur oak germination on open, mineral soils (Sieg and Wright 1996). Under heavy litter cover, acorns were most susceptible to pilferage by rodents and newly developed seedlings are more prone to fungus and insect attack (Fowells 1965, Tinus 1980, USDA-NRCS 2009a). Acorns are extremely vulnerable to predation by insects, small birds and mammals; and unless germination is rapid, few seeds survive (Reichman 1987). Under controlled conditions, seedlings grew best at a day temperature of 31°C and a night temperature of 19°C (Tinus 1980).

The shelterwood regeneration system is an even-aged system where mature trees are removed in two or more consecutive cuttings, with mature trees temporarily serving as a seed and protection source (Nyland 1996). The shelter trees stay on site until a new community of adequate density forms and no longer needs their protection. Shelterwoods are implemented where there is an inadequate seed bank or a change in environmental conditions. "The shelterwood method allows the forester to temper visual characteristics within regenerating stands and maintain essential habitat conditions for selected animals and non-tree vegetation" (Nyland 1996).

To aid in oak regeneration, the shelterwood system is often used to decrease dense shade (Sander et al. 1983; Hannah 1987) allowing oaks already present to develop size and mass (Loftis 1990; Deen et al. 1993); a prerequisite for rapid height growth upon release (Sander 1971, 1972). The first cutting usually initiates regeneration of fast-growing intolerant species (Loftis 1983, Schuler and Miller 1995). The ability to control competition and improve stem form and promote height growth of oak regeneration is necessary in maintaining oak stands on productive sites (Carvell and Tryon 1961, Van Lear and Waldrop 1989; Abrams 1992).

Seed Crop Production

High-quality seed crops happen every two to three years and limited or no seeds are produced in intervening years (Johnson 1990, USDA-NRCS 2009a). Periodic seed producers are dependent upon advantageous disturbance or accumulation of advance reproduction. In order for seedlings to be successful they must grow in height faster than their competition (Clatterbuck and Hodges 1988). A majority of acorns land under or close to the canopy of the parent tree, although several are deposited past the canopy by seed-eating animals. There are no extensive seed banks for oaks since acorns do not carry on from year to year. Instead of a seed bank, many oaks renew from a store of determined seedlings under the canopy known as advance regeneration (Swiecki and Bernhardt 1992).

Bur Oak Seedling Development

The environment under the canopy of oaks should be positive for seedling development. Oak litter guards acorns from aridness and secures a good seedbed for germination. Litter is one of the key traits of the microhabitat for tree seeds in old fields and for old field plants (Monk and Gabrielson 1985) and is potentially complex and contrasting (Facelli and Pickett 1989). Litter can lower seed survival by enhancing fungal development (Shaw 1968) or by escalating seed predation (Abbott and Quink 1970, Price and Jenkins 1986, Schupp 1988b). Alternatively, litter may boost seed success by facilitating dispersal (Jarvis 1964, Shaw 1968, Barnett 1977) and escalating germination of bigger seeds (Barrett 1931, Schopmeyer 1974, Sork 1983). Survival rate of sprouting seedlings to maturity decreases greatly after removal of the oak overstory (McGee and Hooper 1970).

Bur oak seedlings profit from moderate shade and increased soil nutrient levels under the canopy. Seedlings that look dead in August may be back in spring. Energy reserves in the taproot make it possible for seedlings to resprout and develop efficiently from root crowns following shoot loss (Merz and Boyce 1956, Bey 1964, Severson and Boldt 1977, Swiecki and Bernhardt 1992). Bur oak seedlings are known to be resourceful users of water (Wuenscher and Kozlowski 1971). Seedling persistence is subject to site characteristics, such as microclimate and damaging agents that destroy the shoot (Swiecki and Bernhardt 1992). Soil disturbance increases germination of acorns in connection with bare mineral soil (Sieg and Wright 1996).

Bur oaks grow aggressively after fire or additional disturbance (Severson and Boldt 1977). Stump sprouting has been extensively recorded (Stallard 1929, Wasser 1982). As a bottomland species, bur oak seedlings do not like flooding; and mesic, fertile surroundings needed to create seedling establishment (Loucks and Keen 1973, Johnson et al. 1976). In open bottomlands, reproduction can be fast, but yearling death rate is as high as 40 to 50 percent when a seedling is under water for two weeks or longer during the growing season (Tang and Kozlowski 1982). New oak seedlings usually grow more slowly in height compared to older seedlings or sprouts of the same species. Although oaks are relatively intolerant to shade and grow slowly under heavy shade, survival rates are relatively high under moderate shading (Beck 1970, Loftis 1988, Johnson 1993). Seedlings in their first year are poor competitors in terms of height growth (Larsen and Johnson 1998).

Most small oak seedlings present in the understory are poor competitors when either a small canopy gap or larger disturbance occurs (Beck 1970, Ehrenfeld 1980, McGee 1984, Beck and Hooper 1986). At two sites in southwest Wisconsin, understory stems taller than 1.5 m and scattered small canopy trees were removed on half of the main plots, with the other plots retained as controls (Lorimer et al. 1994). Five years later, 74 percent of seedlings planted in the undisturbed mesic forest died compared to only six percent on main plots where tall understory vegetation was removed (Lorimer et al. 1994). Planted seedlings on the understory removal plots at the mesic site displayed a typical height increase of 32cm during the five year period, while seedlings in the undisturbed sections had a net height decrease of 0.05cm. A foliar application of the herbicide Tordon spray onto the understory vegetation had no visual result on survival of planted seedlings on either site. Five years later, seedlings averaged 68cm in height on removal treatments at the mesic site, 52cm on matching treatments at the dry-mesic site, and <35cm on control plots. Site quality had differing effects on the development of planted versus natural oak seedlings. Planted seedlings developed at a reduced rate and had fewer leaves on the dry-mesic site than mesic site (Lorimer et al. 1994).

A harsh drought in 1988 (annual precipitation only 72 percent of 1951-80 mean values) also had no effect on seedling mortality, particularly on the understory removal plots (Lorimer et al. 1994). Mortality in any one year from 1988-1990 was never more than 4.5 percent on the understory removal plots on either site. The drought seemed to have no effect on height growth. "At the end of the experiment the understory removal/foliar spray plots on the dry-mesic site had over 10 times as many oak seedlings as undisturbed plots, while on the mesic site the treated plots had 140 times as many seedlings as the control plots" (Lorimer et al. 1994).

Bur Oak Taproot Development

The taproot of bur oak develops rapidly, reaching approximately 23cm before leaves open (Weaver and Kramer 1932). Root development of immature bur oaks is rapid. At the end of the first year growing period, bur oak roots have been discovered at depths of 1.4m, with lateral spreads of 76cm. Strong initial root growth and elevated water-use efficiency can explain the pioneer status of bur oaks on droughty areas and their ability to successfully compete with prairie shrubs and graminoids (Deitschmann 1965, USDA-NRCS 2009a). A deep tap root system penetrates to lowered water tables during dry periods (USDA-NRCS 2009a). In prairie areas, bur oak and hackberry roots have been found at depths of 3 to 6m and a 43yr old bur oak tree had a lateral spread of 12.5m although the tree that was only 6m tall (Johnson 1990).

Expansive root systems enable oaks to mature quickly in height once suitable conditions occur (Sander 1971). Big roots also aid in the survival after aboveground parts are destroyed or damaged (Larsen and Johnson 1998).

Bur Oak Sapling Development

The sapling phase is the midway point between seedling and overstory trees (Swiecki and Bernhardt 1992). This halfway area includes a size range from a basal diameter of 2.5cm or greater to a DBH of 7.6cm. Oak saplings are persistent rather than fast growing. The speed at which saplings grow is heavily dependent upon soil moisture and browsing.

Bur Oak Stump Sprout Development

Plant regeneration by re-sprouting is the most important mechanism by natural or human disturbance (Retana et al. 1992). Sprouts that branch less with more aggressive height growth are usually from a sprout starting close to the root collar. These trees are able to grow without restriction because of the recently cleared area and usually dominate the new area (Larsen and Johnson 1998). Sprouts usually mature quicker than seedlings (McQuilkin 1975). All oaks can resprout after removal of the stem (Johnson 1975). This ability decreases with larger stem diameter (Johnson 1977). "The negative association of oak sprouting with increasing size may be related to the thicker bark of older trees and/or low carbohydrate reserves associated with senescence in trees" (Johnson 1977).

Bur oaks have the ability to renew themselves from stump sprouts, mainly if the cut trees are youthful and energetic. If a young stand is cut or top-killed, an immense resprout stand of third growth could happen (Swiecki and Bernhardt 1992). Sprout-origin trees can be multi-trunked, or a single trunk with a sweep at the base by the scar of the old stump (Swiecki and Bernhardt 1992). The greatest amount of stems has been found to decrease exponentially with increasing age in many forest stands where differentiation and mortality occur (McFadden and Oliver 1988).

Bur Oak Pests

Few insects or diseases cause serious damage to bur oak (USDA-NRCS 2009a). Bur oaks can be assaulted from many different defoliating insects like redhumped oakworm *(Symmerista canicosta)* in the Northeast, *S. albifrons* in the South, and oak webworm *(Archips fervidana),* oak skeletonizer *(Bucculatrix recognita),* a leaf miner *(Profenusa lucifex),* variable oakleaf caterpillar *(Heterocampa manteo),* June beetles

(Phyllophaga spp.), and oak lacebug *(Corythucha arcuata)* throughout most of its range (Baker 1972, Deitschmann 1965, USDA-NRCS 2009a). Dry weather can be a key factor that causes the intense defoliation of bur oaks in shelterbelts. Bur oak kermes *(Kermes pubescens)* assaults could disfigure leaves and kill twigs of bur oak (Deitschmann 1965, Baker 1972, USDA-NRCSa 2009). In southern Ohio, nut weevils are the key factor disturbing an acorn crop (Barrett 1931).

Bur oak is vulnerable to invasion by cotton root rot *(Phymatotrichum omnivorum)* and Strumella canker *(Strumella coryneoidea)* (Johnson 1990, USDA-NRCS 2009a). Additional fungi that have been seperated from infected sections of bur oak include Dothiorella canker and dieback *(Dothiorella quercina),* Phoma canker *(Phoma aposphaerioides),* Coniothyrium dieback *(Coniothyrium truncisedum),* and shoestring root rot *(Armillaria mellea).*

Succession of Oak Communities

Oak is one of the leading species groups in the eastern deciduous forests of North America. In specific areas of the east, oak dominance mirrors the significance of this genus in pre-settlement forests (Spurr 1951, Whitney and Davis 1986, Abrams and Downs 1990). In the eastern section of North America the Oak-Hickory forest is the peak and most important forest type over the past 6000-9000 years of the postglacial period (Weaver and Clements 1938, Braun 1950, Craig 1969, Watts 1979, Delcourt and Delcourt 1985).

The dominance of oak in pre-settlement woodlands is of interest because nearly all oaks are seen as early to mid-successional species (Abrams 1992). Pre-settlement stands were most likely uneven aged with sprinkled mortality and recruitment in canopy gaps. While present even-aged stands grow they are open to extensive declines, affecting large groups of trees that need renewal of the entire stand over a small time frame (Swiecki and Bernhardt 1992). Oaks have historically been linked to repeated fires (Watts 1979). "Fire, whether it has occurred low, moderate, or high heat degree intensity, seems to be the common denominator for the development of oak forests on upland sites and their past and present ecological status" (Abrams 1990).

Several current central hardwood forests have canopies dominated by expansive oaks and understory controlled by shade-tolerant genera such as maple (*Acer*) and beech (*Fagus*) (Lorimer1993). Three factors associated with European settlement during the 1800s explain the 1926 bell-shaped size distributions (within even aged populations): timber harvest, release of suppressed vegetation and removal of under growth through fire and grazing (Parker 1997, Hicks 1998). Timber mill records (Steer 1948) document oaks comprised 80 percent of Indiana timber production from 1869 to 1899, suggesting that large oaks were present when European settlers arrived (Parker 1997).

The variation found in hardwood forest composition in the central Virginia Coastal Plain showed no correlation with edaphic and topographic variables (DeWitt and Ware 1979). Structural differences among the forest stands not explainable by edaphic and topographic variation might be accounted for, in part, by a time variable. The general pattern of early forest development in this area is one of successive replacement of one leading dominant, and more sharing of dominance by several species (Monette and Ware 1983). White oak remained an important canopy tree in the oldest stands, and insignificant or absent in the understory stratum of the five oldest stands (Monette and Ware 1983). The poor representation of white oak in the small tree category in the older stands may indicate a continuing decrease in the importance of this species in the future. But the highest white oak seedling density occurs in these same stands, so there is potential for continued moderate importance of white oak (Monette and Ware 1983).

Bur oak appears as an understory species in ponderosa pine stands in the Black Hills of South Dakota. The tree is a short-statured, shrubby plant in ravines and along watercourses emptying into the foothills. Poor tree reproduction is frequent in timbered draws and stringers in the Northern Great Plains in bur oak or eastern deciduous forests (Severson and Boldt 1978, Sieg 1991). Western snowberry (*Symphoricarpos occidentalis*) is the primary shrub found in bur oak communities and Kentucky bluegrass (*Poa pratensis*) displaces native sedges (*Carex* spp.) (Hodorff et al. 1989).

Impact of Fire on Bur Oak Communities

Bur oak bark is thick and fire resistant (Daubenmire, 1936, Wasser 1982, Lorimer 1985, USDA-NRCS 2009a). Bur oaks respond well to fire (Daubenmire, 1936, Hoffman and Alexander 1987) with large bur oaks usually living through fires (Stallard 1929, Stout 1944, Wasser 1982, USDA-NRCS 2009a) "Mature oaks are usually killed only by 'severe' fire "(Daubenmire 1936). Bur oaks sprout aggressively at the stump or root crown after fire (Stallard 1929, Severson and Boldt 1977, Lorimer 1985). Oaks are impervious to decay after scarring and seem to flourish where fire has created seedbeds for acorn germination (Lorimer 1985). Vigorous sprout development occurs after burning or cutting of pole-size or smaller bur oaks (Fowells 1965, Hoffman and Alexander 1987, Reichman 1987) while the growth of bigger trees is usually less vigorous (Fowells 1965). With recurrent fire, oak can dominate a site; without fire oak will stay a part of a successional forest (Crow et al. 1994).

Bur oak communities respond well to fire and other disturbances (Hoffman and Alexander 1987, Reichman 1987) and grow rapidly after old growth maple-basswood (*Acer* spp.-*Tilia* spp.) forests were removed by fire (Stallard 1929, Daubenmire 1936). Where fire control is common, bur oak communities can be replaced by more shade-tolerant maple-basswood (*Acer* spp.-*Tilia* spp.) forests (Stallard 1929, Daubenmire 1936, USDA-NRCS 2009a). Recurrent light fires help oak recruitment by creating opportunities by limiting overstory density where non-oak, shade tolerant, fire-intolerant species are hampered and oak seedling sprouts having the advantage (Maslen 1989, Larsen and Johnson1998).

Early European colonists were commonly ignorant of wildland fire and ways to contain them, just as European foresters in the nineteenth century were unaware of prescribed burning as a silvicultural technique (Pyne 1983). Human behavior over the past 150 years has changed the occurrence and amount of fires in oak forests (Swiecki and Bernhardt 1992).

"Tallgrass prairie dominated by big bluestem (*Andropogon gerardii)*, Indiangrass (*Sorghastrum nutans*) and switchgrass (*Panicum virgatum*) once occupied a 575,000 km^2, roughly a triangular area bounded by central Texas to eastern North Dakota and western Indiana" (Küchler 1964). Oak savannas were regular communities on the tallgrass prairie and during the settlement era they covered 11-13 million ha of the Midwest (Nuzzo 1986). Savanna oak species commonly found include bur oak, black oak (*Q. velutina*), white oak, northern pin oak, post oak, and blackjack oak (*Q. marilandica*). These savannas had a prairie-like understory and were thought to have developed from pre-existing forests exposed to frequent burning (Curtis 1959). Pre-settlement prairie border forests were also dominated by oak and maintained by recurring fire (Gleason 1913, Rodgers and Anderson 1979, Pallardy et al. 1988). Pre-settlement, tallgrass prairie developed and flourished in an environment that included recurring fire at one to 10 year intervals (Axelrod 1985).

Recurrent fire in the Central Plains occurred from lightning strikes and Indian activities including cooking, heat, ceramic making, manufacture, communication, field preparation for cultivation, combating insects, hunting, and killing woody vegetation (Day 1953). Once the Central Plains were settled by Europeans, fire occurrence and concentration declined because of road construction, expansion of towns, increased agriculture, intensive cattle grazing, wildfire suppression, and recommendations against burning prairie in the mid-1900s (Abrams 1986). During this period, forests quickly grew at the cost of prairie vegetation and the dominant species in these newly formed forests were generally oak. Instances of development comprise the formation of blackjack and post oak forests in central Oklahoma; white oak, shingle oak (*Q. imbricaria*), bur oak, and American elm forests in central Missouri; and bur oak and chinquapin oak (*Q. muhlenbergii*) forests in eastern Kansas (Howell and Kucera 1956, Rice and Penfound 1959, Abrams 1986).

The decrease in fire occurrence, combined with the drought tolerance of oak, facilitated oak invasion and survival after European settlement. There exist similarities between tallgrass prairie and oak savannas in their alteration to closed woodlands with reduced fire (Rice and Penfound 1959, Abrams 1986).

Prescribed Burning

Prescribed burning is a tool that "thins stands from below". Fire top-kills smaller oak trees and leaves larger trees as acorn producers (White 1983, 1986). Severe prescribed burning can decrease the canopy cover of the overstory, increasing light levels which are paramount in oak reproduction (Johnson 1993). Deciduous trees and shrubs can survive or expand when top-killed by fire due to the ability to sprout from rhizomes, root collars, or stems (White 1983, 1986). After burning, western snowberry and chokecherry sprout in oak stands (Wright and Bailey 1982).

Basal sprouting improved as scorch height increased after a prescribed burn with sprouting only stimulated when the tree was damaged (Sieg and Wright 1996). Soil moisture is a key element that controls the reaction of plant populations to fire, mainly in arid regions (Wright and Bailey 1982).

Sieg and Wright (1996) developed 192 6-by-6 m plots that were fenced to exclude livestock in 1986. The prescription to burn included plots in the fall after a severe frost or a spring burn before trees leafed out and cool season grasses reached adequate height. More sprouts were present after the burn on trees in ravines compared to slopes or floodplains. As burn height increased basal sprouting of oaks increased, indicating that damaging a tree was the key to encourage sprouting. After burning in other regions, chokecherry sprouted vigorously along with many other woody species (e.g. Pelton 1953, Miller 1963, Wright and Bailey 1982). Germination speed of oak acorns or survival of seedlings was not improved by burning (Sieg and Wright 1996).

Controlled burning was initially established in a section of the Cedar Creek Natural History Area (CCNHA) in eastcentral Minnesota in 1964 (White 1983). The rationale for prescribed burning was to return the site to its pre-settlement oak savanna formation and arrangement. Stems $\geq$ 5 cm diameter at breast height (DBH) within a 100-m^2 circle were recorded by species, DBH, and age; then counted and measured in 1 plot/ha with a total of

21 nested plots (White 1983). The overstory was dominated by northern pin oak, having 98 percent of the stems in the unburned and 92 percent of the stems in the burned site. Northern pin oak basal area did not differ between the unburned areas compared to burned areas (Wright 1983). Roughly 63 percent of the entire change in density (stems/ha) was due to greater densities in the unburned area.

Prescribed burning effectively eliminates smaller diameter northern pin oak (White 1983). Density of northern pin oak was greater than the density of early savannas reported by Bray (1955), which ranged from 13 to 36 stems/ha. Bray (1955) showed that switching from oak savanna to oak woods would take >13 yr using annual spring burns (White 1983). Larger (DBH >25 cm) northern pin oak were not killed due to insufficient fire intensity using a spring prescribed burning program.

Oak trees could be selected in a prescribed burn in shelterwood because oak emphasizes root growth over shoot development. Species that compete with oak do not grow well after burning situations (Kelty 1989; Kolb et al. 1990). Prior to burning, 89 percent of all oak (*Quercus* ssp.) reproduction had crooked or flat-topped stems. All treatments (control, summer, winter and spring burns) grew oak stems in the initial growing season after the prescribed burn. Among fire treatments (low, low medium, medium high and high), straight stems occurred in 84 percent of the time (Brose and Van Lear 1998). Oak reproduction that sprouts after fire exhibits improved stem form and height growth relative to no burn. Besides contributing to oak's superior fire resistance, large roots probably account for the improvement in oak stem form after fire treatments (Brose and Van Lear 1998). Proper stem structure is paramount because new sprouts are secured, free of disease and more likely to develop into solid timber trees (Roth and Hepting 1943, Teuke and Van Lear 1982).

Impact of Browsing and Other Animal Disturbances on Bur Oak Regeneration

"Bur oak provides food and cover for wildlife species" (Wasser 1982); however protein and energy values are low (Dittberner and Olson 1983). Seedlings can be broken and consumed by livestock (Stout 1944, Gildar 2002). Several wildlife species and domestic livestock consider bur oak very edible (Wasser 1982, Uresk and Lowery 1984). Acorns are consumed by several birds and mammals such as squirrels, deer, wood ducks, blue jays, cows, mice, thirteen-lined ground squirrels, New England cottontails and other rodents (Deitschmann 1965, Welsh 1986, Reichman 1987, Uresk and Lowery 1984, USDA-NRCS 2009a). Seed dispersal can be assisted by rodents and blue jays that regularly hoard acorns for future use (Tirmenstein 1988). The value of bur oak cover is important for providing escape cover and nesting habitat for many birds and mammals, including red-tailed hawks, screech owls, fox and flying squirrels (Stout 1944, USDA-NRCS 2009a).

Juvenile trees are constantly threatened by animal depredation. Browsing has been documented as a serious problem in artificial and natural regeneration of forests in a number of countries (Hibbs and Yoder 1993, Gerber and Schmidt 1996, Van Hees et al. 1996, Fuchs et al. 2000, Harmer and Gill 2000, Gill and Beardall 2001, Sipe and Bazzaz 2001, Vila et al. 2001, Sawadogo et al. 2002). Oak regeneration has been hampered by browsing damage from wildlife in northeast France (Chaar et al. 1997, Chaar and Colin 1999). Plants may undergo major morphological changes after browsing such as reduced height and side shoots, and reduced foliage density. Browsing can result in severe growth loss (Eiberle 1978, Gill and Beardall 2001).

Continuous browsing harms the plant more than any one episode does, with decreasing growth and stem development creating an imbalance in the shoot-root ratio (Eiberle 1980, Gill 1992b, Gill and Beardall 2001). Intense browsing can eliminate seedlings (Eiberle and Nigg 1987, Gill 1992b, Gerber and Schmidt 1996). Oak trees can survive continuous elimination of above ground stump parts by re-sprouting and remain in a suppressed state for long periods (Anderson 1991, Braithwaite and Mayhead 1996, Gerber and Schmidt 1996, Harmer 2001). Tap roots from white oak saplings can be older than stump portions (Hibbs and Yoder 1993). "A hypothesis for why this occurs may be that plants under severe browsing stress allocated a major part of their resources into the root system, as was reported for grazed shrubs of scarlet oak (*Q. cocciferea*)" (Papatheodorou et al. 1998).

Continuous shoot elimination can place additional food stores to the roots. Expansive and mature root structures are key resources of carbohydrates for regrowth or re-sprouting (Crow 1988). Browsing extensively influenced the growth of immature regenerated durmast oak (*Quercus petraea*) using a modest (i.e. 10 roe deer per 100 ha) browsing intensity. Comparable results were documented for four to 15 year old northern red oak saplings from a Dutch forest area (Van Hees et al. 1996). Other than leaf biomass, the effect of browsing on seedlings biomass allocation was small but evident. Browsed plants were smaller and 30 percent were of shrubby form. Stem biomass did not differ between browsed and unbrowsed stems; however, there were additional stems and leaf biomass on the browsed (Drexhage and Colin 2003).

Past studies have shown tree species have varying levels of seed predation (Mittelbach and Gross 1984) which declines further from seed source (Janzen 1970, Clark and Clark 1984, Howe et al. 1985, Coates-Estrada and Estrada 1988, Schupp 1988a, Sork et al. 1988) and seed success increases with improved formation and greater diversity of vegetation (Thompson 1982, Casper 1987). Seed predators prefer certain micro-habitats; therefore, seed predation is often spatially heterogeneous (Thompson 1982, Sork 1983, Mittelbach and Gross 1984, Louda and Zedler 1985, Webb and Willson 1985, Casper 1987, Schupp 1988a, 1988b). Predation may be seed density independent (Mittelbach and Gross 1984, Webb and Willson 1985) or dependent (Casper 1987, Sork et al. 1988), depending on the foraging habits of seed predators (Reichman and Oberstein 1977). Litter type and depth differences affect seed predation (Price and Jenkins 1986, Schupp 1988b). Litter influences seed detectability because rodents rely on olfaction to locate seeds. Litter can influence odor diffusion (Price and Jenkins 1986). Squirrels mainly stay inside, or at the forest edge, to forage for nuts and rarely venture outside of the woods to cache nuts.

The co-evolutionary ties between scatter-hoarding squirrels and mast-producing trees have been well documented. Gray squirrels are partially responsible for the continued success of many deciduous tree groups (Steele et al. 1993, Vander Wall 1990, 2001). Red and gray squirrels have different styles of storing mast items. Red squirrels are considered larder-hoarders and store their food (88.9 percent) in trees; whereas, gray squirrels are considered scatter-hoarders and store their food (96.9 percent) in the ground (Goheen and Swihart 2003).

Gray squirrels do not protect their caches from nonspecific or other possible seed predators. Burial of nuts by scatter-hoarding animals is highly advantageous, if not essential for regeneration of mast-producing trees (Vander Wall 2001). Scatter hoarding concurrently decreases the likelihood of seed predation, preserves the viability of seeds, and helps germination and establishment of seedlings (Vander Wall 1990). Changes in scatter hoarding could have an effect on regeneration success of affected tree species (Goheen and Swihart 2003).

White-tailed Deer Browsing Impacts

Deer can strongly impact the absolute and relative abundance of woody species (Leopold et al. 1947, Webb et al. 1956). Deer browsing impacts on oak regeneration are consistently cited in forestry textbooks (Allen and Sharpe 1990). Decrease in height is controlled by intensity and frequency of browsing, fluctuating among tree species and selection by deer, or the tree's ability to recover from damage (Eiblerle 1978, Roth 1996, Gill et al. 2000). Browsing has caused tree seedlings' heights to be limited to <50 cm after 25 years (Shaw 1974) giving some species with a low growing appearance with thicker foliage on the flank (side) branches (Forde 1989, Holisova et al. 1992). Plants that grow above browsing height may have all foliage and side branches removed or depleted, producing plants that are tall with reduced foliage and flowering potential (Pollard and Cooke 1994, Martin and Daufresne 1999).

Ecological populations and composition are deeply altered by ungulates. The plant and animal populations are directly and indirectly altered by deer browsing. Deer alter the plants they interrelate with and disturb their dispersal or population. Deer can become a keystone species by reorganizing whole ecological communities (McShea and Rappole 1992, Waller and Alverson 1997, Paine 2000, Rooney 2001).

Since white-tailed deer are a keystone species, current and future restructuring of forest populations is anticipated (Rooney and Waller 2003). While some tree species thrive on elevated deer populations, many species decrease in forest populations based on empirical investigations of forest-herb communities in Wisconsin (Rooney and Waller 2003). Ungulate browsing can decrease local regeneration of preferred tree species for years (Beals et al. 1960).

Deer browsing results were studied on two deciduous tree species, red oak and yellow birch (*Betula alleghaniensis*), in mixed conifer hardwood forests of northern Wisconsin (Rooney and Waller 2003). These species were browsed in the spring and summer with red oak seedling density varying due to local deer browse. Seedling concentration fell suddenly as browsing pressure increased from low to intermediate; signifying red oak regeneration is strongly controlled by deer. However, seedling concentrations were greatest at intermediate deer densities, and lowest in very low and very high deer densities.

Deer browsing decreases vegetative cover in the woodland understory, changing forest floor microclimate. Temperature and light are elevated while humidity and soil moisture decrease (Suominen 1999). Suppression or elimination of palatable seedlings and saplings results in a slow but steady conversion of the stand to less palatable species (Waller and Alverson 1997). "Deer are one of the more important determinants of forest structure in the Allegheny Plateau over the past 50 years" (Whitney 1984). With elevated deer populations, seedlings and saplings of entire tree species are eliminated and forests become park-like with an understory of grass and fern (Waller and Alverson 1997). In central Illinois, deer browsed a large quantity of forage from uncommon species such as white oak and shagbark hickory (*Carya ovata*) (Strole and Anderson 1992). Similarly, in a 10-year study of upland beech-maple (*Fagus-Acer*), lowland ash-elm (*Fraxinus-Ulmus*), and young pin oak forests in Ohio, Boerner and Brinkman (1996) concluded that deer browsing was more important than environmental gradients or climate factors in determining seedling longevity and mortality. Deer also altered normal patterns of succession by reducing the advanced regeneration of hemlock seedlings (Hough 1965).

Old field succession to woodlands was started by massive farmland desertion in the eastern and central United States. Along the edge of deserted fields several tree species have established themselves (Inouye et al. 1994). Before colonization in the 1850s, various sites that are now old fields and upland woods were fire-preserved oak savannah controlled by bur oak and herbaceous vegetation typical of tallgrass prairie. The borders along woodlands fashioned by farm desertion are prime habitat for white-tailed deer. White-tailed deer have increased populations in specific locations and the species has increased its range northward (Tierson et al.1966, Ross et al. 1970, Anderson and Loucks 1979, Alverson et al. 1988). Many trees found in fields next to the field-forest edge show signs of continuous browsing by ungulates, implying deer play an important role in reducing the invasion of old fields by trees (Inouye et al. 1994). Concentration of trees and average tree height was greatest close to the forest and declined with distance from the woods for all tree species (Inouye et al. 1994). The rate of tree establishment can be influenced by deer, altering relative quantity of forest species and diminishing the development of trees after they are established (Alverson et al 1988).

In northern hardwood forests, selective browsing by white-tailed deer can suppress regeneration of palatable sugar maple (*Acer saccharum*) and pin cherry (*Prunus* *pensylvanica*) in forest openings, while less edible American beech, birch, and striped maple (*A. pensylvanicum*) increase in relative abundance (Tierson et al. 1966, Marquis 1981). Selective browsing on oak seedlings can hinder the regeneration of this species relative to other deciduous trees (Strole and Anderson 1992, Healy 1997). However, deer browsing has the ability to remove all tree regeneration, depending on deer density, over all seedling abundance (Jordan 1967, Marquis 1974), and growth conditions (Tilghman 1989, Healy 1997).

Bur oak foliage is a food for white-tailed deer in the Black Hills (Uresk and Lowery 1984). Deer can adversely alter seedling and sapling recruitment and growth, but they are normally less damaging than livestock (Swiecki and Bernhardt 1992). Northern red oak, white oak, and red maple stump sprouts in the oak stands of Lower Michigan were exploited by deer and may have decreased the probability of direct-seeded seedlings being browsed in the oak stands (Buckley et al. 1998). Deer have greater influence through browsing by eliminating understory vegetation than without removing understory vegetation (Gordon et al. 1995). Northern red oak seedlings that grew above the herb layer were browsed intensively, leading to seedling height identical to the herb layer for many years (Gottschalk and Marquis 1983). Deer concentrate on areas like clear-cuts that increase concentrations of forage (Tilghman 1989, Johnson et al. 1995) with seedlings of high palatability (Blair et al. 1983).

Clear-cutting creates severe alterations in the food supply of deer (Johnson et al. 1995). The result of this type of manipulation is the complete loss of forage produced by the forest canopy. The most important cost is loss of acorn production. Complete acorn production will not peak for close to 40 years in forest stands (Fowells 1965) and oaks do not reestablish properly in stands after clear-cutting (Loftis 1988). However, clear-cutting creates a direct response of sprout growth from stumps and an invasion of early successional species, both woody and herbaceous. Edges associated with clear-cuts are where the most intense deer grazing occurred on herbaceous plants (Wentworth et al. 1990b).

Cutting as a Tool for Bur Oak Regeneration

Clear-cutting is a harvest and regeneration of the forest and is done to improve stand quality, growth, genetics and species composition (Belt and Campbell 1999). As a management device clear-cutting has been a practical forest tool for many years (Smith 1962). It is one of many cutting methods available to foresters and where there is a poor history of forest fire, clear-cutting is the best and occasionally the only viable method (Belt and Campbell 1999). Clear-cutting's effects on the biota would be of great aid to foresters in determining post-cutting management (Horn 1980).

Clear-cutting can be used to improve forest health, productivity and quality. Clear-cutting causes a timber stand to compete from ground level; the future stand will consist of the fastest-growing, healthiest, straightest, tallest and best quality trees (Belt and Campbell 1999). "Clear-cutting is a successful and reliable method for regeneration of oak forests; whereas, single-tree openings lack adequate sunlight for sufficient reproduction and growth to permit overstory recruitment" (Larsen and Johnson 1998). The herbaceous layer was the location of vast inter-site alteration following clear-cutting in Southern Appalachian woods. Some herbaceous communities displayed minimal effects from clear-cutting, while others decreased dramatically. After cutting, Southern Appalachian woods regenerate rapidly. Red maple and yellow poplar were the leading dominants after clear-cutting while climax species such as white and chestnut oak, white pine (*Pinus strobus*), and hickories (*Carya* spp.) were present but in smaller quantities (Horn 1980). The new canopy consisted of fewer oaks than in the unlogged woodlands and consisted of larger contributions from red maple and yellow poplar (Horn 1980).

A study in the Shawnee National Forest scored oak and hickory trees with a DBH of 5cm and greater for straightness (Bey 1964). Tree age was 70 to 90 years old and 75 percent consisted of white oak, black oak, scarlet oak, and hickory in the treatments. Twenty-seven years after fast development the dominant and co-dominant trees were straight, in potentially excellent condition, and matured quickly (Bey 1964). A vast number of the oak seedlings that grew following cutting developed from older root systems (Leffelman and Hawley 1925, Merz and Boyce 1956). Where the conditions were optimal for increased oak reproduction, the outcome of clear-cutting was a large fraction of quick-maturing, straight stemmed oaks (Liming and Johnson 1944, Merz and Boyce 1958).

Boring et al. (1981) conducted a study on the Coweeta Basin in the Nantahala Mountains, part of the Blue Ridge province in the southern Appalachians. Roughly 16.4 ha were clear-cut in the winter of 1976-1977 and tested for regeneration over the following growing season. Sprout density was more than two times the density of seedlings. Once logging was completed the majority of previously established seedlings that were broken during logging sprouted prolifically (Boring et al. 1981).

Regeneration of Bur Oak

It has been hypothesized that the renewal niche of oak has been limited since European colonization because of elevated competition with fire intolerant plant species (Crow 1988, Lorimer 1989, Abrams 1992, Van Learn and Watt 1993). The physical features, adaptive abilities and renewal niches of oaks differ greatly among oak species. In eastern North America where forests are dominated by oaks, species variations were most noticeably limited by soil moisture relations, which strongly control the dynamics of the regeneration process. Accordingly, it is fitting to categorize oaks species as xerophytes, mesophytes, or hydrophytes. The capability to die back and resprout to regenerate is greatly a function of drought tolerance of xerophytic oaks. Dry environments create little competition from other species in fire prone regions where xerophytic oaks usually grow. Large root systems are characteristic of oaks that have a high tolerance of both fire and drought (Larsen and Johnson 1998). Larsen and Johnson (1998) also stated that growth of the mesophytic and hydrophytic oaks is less dependent on root mass than for xerophytic oaks.

Competition level is more intense now than in pre-colonization woods for northern red oak seedlings and saplings. Herbs, shrubs and trees of the understory become significant competitors with other tree species after elimination of the overstory (Johnson and Jacobs 1981, Crow and Isebrands 1986, Loftis 1990, Teclaw and Isebrands 1991, 1993). The most successful sites for northern red oak occur on moderately productive sites where disturbances and stresses limit the abundance and type of competition (Kolb et al. 1990, Kotar 1991, Hodges and Gardiner 1993, Lorimer 1993).

Natural regeneration of northern red oak and sessile oak is frequently profitable in industrial forestry operations in western and central Europe. Soil scarification, supplemental sowing and planting, and continuous control of competing vegetation are some of the intensive measures used (Klepac 1981, Evans 1982).

Models of colonization and succession were dependent upon multiple site variables after a forest disturbance including the seasonal timing of the disturbance, the area's size, shape, topographic features, amount of leaf and canopy debris, degree of disturbance to litter and topsoil, species present before disturbance and intact plants after disturbance (Boring et al. 1981). Natural and man-made disturbances range dramatically in their pressure on site variables. Perhaps one of the most important factors affecting forest regeneration is minimal disturbance of the soil, litter and plant roots.

Conclusion of Bur Oak Management Strategies

Management efforts like mechanical cutting and clearing, livestock grazing, fire and the introduction of non-native grasses and forbs are key inputs on oak forests. These practices have changed the make-up of the oak forest understory (Swiecki and Bernhardt 1992).

STUDY AREA

Location

This study was conducted within the North Unit of the Gilbert C. Grafton North Dakota Army National Guard Training Base in eastcentral North Dakota (Figure 2). Camp Grafton North (CGN) is located approximately 9.7 km south of Devils Lake, North Dakota, in southern Ramsey County (Figure 3). According to the United States Geological Survey (USGS) 7.5 Minute Quadrangle Map Camp Grafton, North Dakota, CGN is located between 48.036 to 48.062 N Lat. and -98.891 to -98.934 W Long. on sections NE1/2 19 and 29; W1/2 21 and 28, 20; T153N; R64W. Located on the north shore of Devils Lake, CGN has a land area of approximately 613 hectares (Barker et al. 2000).

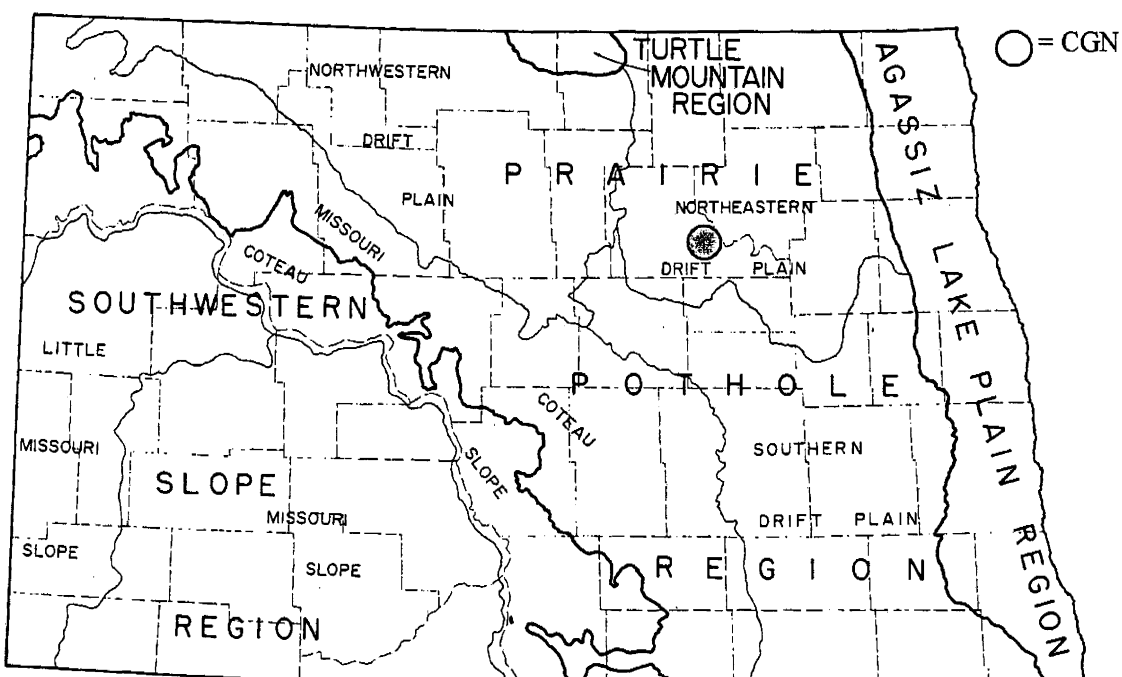

Figure 2. Location of Camp Grafton North (CGN) Unit by physiographic region in North Dakota (Modified from Stewart 1975).

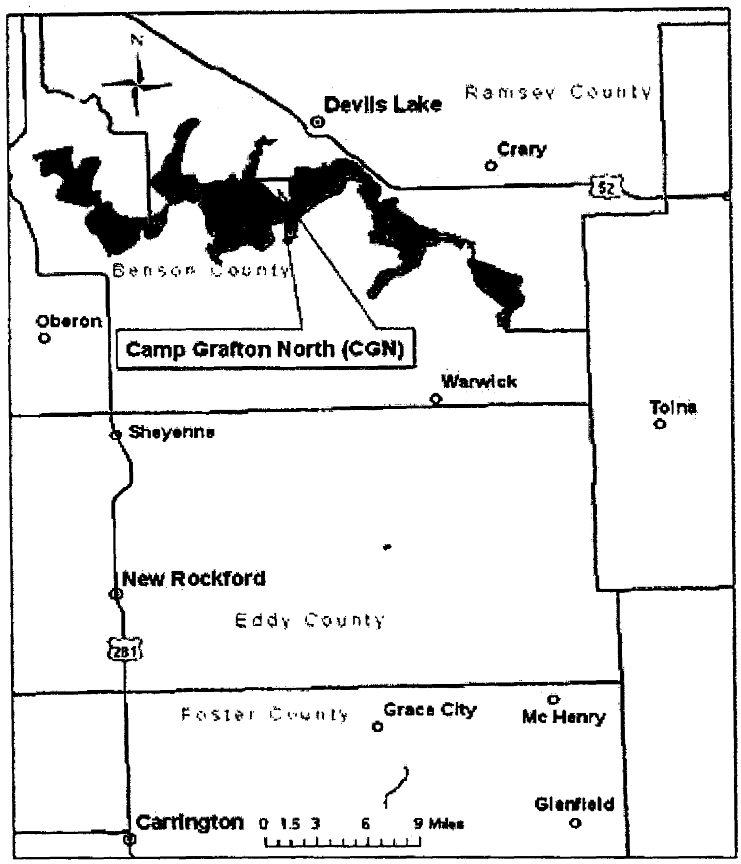

Figure 3. Location of Camp Grafton North (CGN) Unit within Benson County, North Dakota (Modified from Barker et al. 2000).

Camp Grafton North contains a large white-tailed deer population, with an estimated population of 45 deer/km^2 based on an aerial survey conducted by the North Dakota Game and Fish Department in early March, 2007. White-tailed deer are often in view when traveling within CGN. The only method used to control the deer population prior to the initiation of this study was hunting by disabled veterans. However, an archery season was opened to the general public in 2007 to help reduce deer numbers. The season continued in 2008 through 2009.

Vegetation

Camp Grafton North is dominated by a bur oak/green ash forest community (Table 1) with 90% of CGN considered a hardwood forest plant community (Figure 4). Other tree species found in this community include American elm, boxelder, quaking aspen, balsam poplar (*P. balsamifera*) and eastern cottonwood (Barker et al. 2000). The main species in the understory shrub community consist of chokecherry, juneberry, Virginia creeper (*Parthinocissus quinquefolia*), poison ivy (*Toxicodendron radicans*) and western snowberry.

Table 1. Number of trees per hectare by species at various heights at Camp Grafton North during the pre-study survey in 2006.

Tree height	Bur Oak	Green Ash	Hackberry	American Elm	Boxelder
[illegible]0 cm	14.6	20.7	20.7	18.3	12.2
30-150 cm	0	3.7	1.2	0	0
[illegible]00 cm	2.5	3.7	0	0	0
> 300 cm	265.6	383.7	40.3	20.7	2.5

Forb species common within the forest community include wild sarsaparilla (*Aralia nudicaulis*), stickywilly (*Galium aparine*), hog peanut (*Amphicarpea bracteata*), yellow wood sorrel (*Oxalis stricta*), American stickseed (*Hackelia deflexa* var. *americana*), stinging nettle (*Urtica dioica*), Canadian wood violet (*Viola canadensis*), field sow thistle (*Sonchus arvensis*), beggars tick (*Bidens frondosa*), purple meadow-rue (*Thalictrum dasycarpum*), black mustard (*Brassica nigra*) and catnip (*Nepeta cataria*).

Common graminoid species found include Sprengel's sedge (*Carex sprengelii*), wooly sedge (*Carex pellita*), Kentucky bluegrass (*Poa pratensis*), smooth brome (*Bromus inermis*), eastern bottlebrush grass (*Elymus hystrix* var. *bigeloviana*), little seed ricegrass

(*Piptatherum micranthum*), wirestem muhly (*Muhlenbergia frondosa*), reed canarygrass (*Phalaris arundinacea*), squirrel tail (*Sitanion hystrix*), Canada wildrye (*Elymus canadensis*) and Virginia wildrye (*Elymus virginicus*) (Prosser 1998, Rush 2007). Plant species were referenced from Flora of the Great Plains (Great Plains Flora Association 1986) and the Natural Resource Conservation Service Plants Database (USDA-NRCS 2009b).

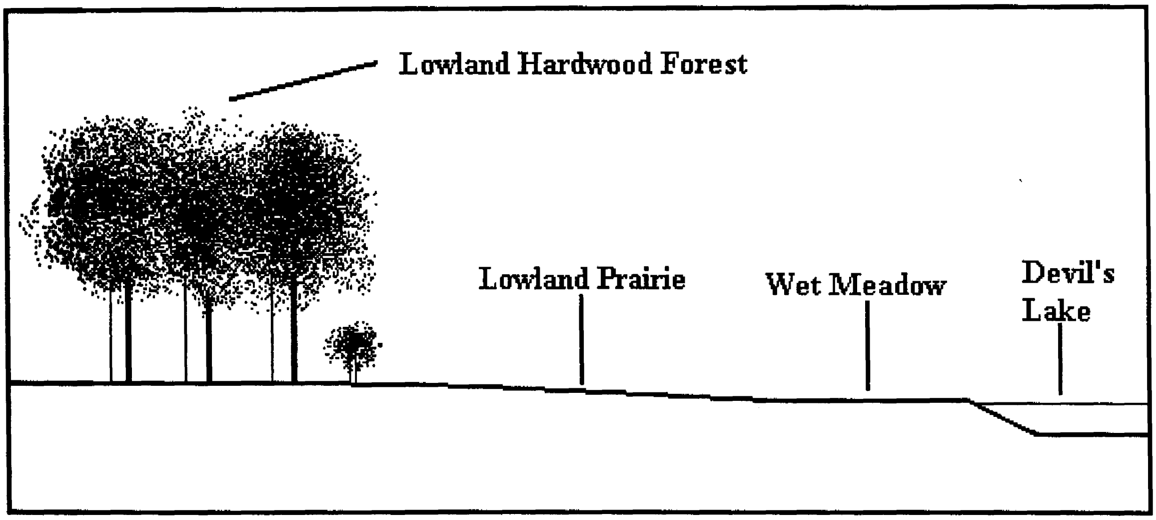

Figure 4. Location of plant communities found on the landscape at Camp Grafton North Unit, North Dakota (Goertel 1999).

Soils

Camp Grafton North is comprised of loamy soils. The four study sites were characterized as the Bottineau loam series, two with slopes of zero to three percent and two with slopes of three to six percent.

Geographically, the Bottineau soil series are found on level to hilly uplands with slopes ranging from 0 to 25 percent with elevations ranging from 305 to 762m (USDA-NRCS 2009b). The Bottineau series consists of very deep, well drained, moderately slow permeable soils that formed in loamy glacial till (USDA-NRCS 2009b). The horizon consists of an Oi 0-5cm, A 5-23cm, Bt1 23-38cm, Bt2 38-51cm, Bt3 51-69cm, Btk 69-114cm, and a C 114-152cm. The taxonomic classification of the Bottineau series is a fine-loamy, mixed, superactive, frigid Alfic Argiudoll (USDA-NRCS 2009b). Common land use for these soils is pastureland and timber production, with some small grains, alfalfa, and native hay production (USDA-NRCS 2009b).

Climate

Climate at CGN is categorized as sub-humid continental with warm summers and cold winters, with mean annual temperature and precipitation 4.4 C° and 48.7cm; respectively. The frost free period ranges from 100 to 120 days (USDA-NRCS 2009b). Most of the precipitation falls between the months of April and November (Jensen 1998). Precipitation and temperature for CGN were measured at the Crary weather station in Ramsey County (NDAWN 2010, NOAA 2010).

Average temperatures at the Crary weather station in 2006 and 2007 were above the 30-year mean of 4.4° C. The average temperature was 5.5° C in 2006 and 4.1° C in 2007. The monthly average temperatures for May, June, July, and August in 2006 and 2007 were above or equal to the 30-year mean temperature; however, the average temperature for August 2007 fell below the 30-year mean (Table 2) (NDAWN 2010).

Average temperatures at the Crary station in 2008 were 3° C below the 30 year mean of 4.4° C. The monthly average temperatures for May, June and July in 2008 were below or equal to the 30 year mean temperature and December through March average temperatures were well below the 30 year mean temperature August rose above the 30 year mean (Table 2) (NDAWN 2010).

Table 2. Average monthly temperature (°C) for the 30-year average, 2006 - 2009 at the weather stations near Devils Lake, North Dakota (NDAWN 2010).

Month	30-year Average	2006	2007	2008	2009
uary	-15	-5	-13	-16	-18
February	-11	-14	-17	-16	-14
rch	-4	-5	-3	-6	-8
April	5	9	5	4	3
ay	13	13	13	10	10
June	18	19	19	16	16
ly	20	22	22	20	18
August	19	20	17	20	19
ember	13	13	14	14	18
October	6	4	8	7	4
ember	-4	-3	-3	-3	3
December	-12	-7	-13	-17	-14

Average temperatures at the Crary station in 2009 were below the 30 year mean of 4.4° C. The average temperature was 3° C in 2009. The monthly average temperatures for May, June, July, and August in 2009 were below or equal to the 30 year mean temperature; however, September's average temperature was well above the 30 year mean and winter months well below the 30 year mean (Table 2) (NDAWN 2010).

The 30-year mean annual precipitation at the Crary weather station is 48.7cm. The annual precipitation amount for this weather station was lower than the 30-year mean in 2006 at 44.6cm. The monthly average precipitation amounts for May, June, July, and August were below the 30-year monthly precipitation means. However, the monthly average precipitation for the same months was above the 30 year mean monthly precipitation in 2007 (Table 3).

The annual precipitation amount for this station was below the 30-year mean in 2008 at 48cm. The monthly average precipitation amounts for May, June, July, and August were below the 30-year mean monthly precipitation. However, the monthly average precipitation for September through November was more than double the 30-year mean (Table 3).

The annual precipitation amount for this station was below the 30-year mean in 2009 at 47.6cm. The monthly average precipitation amounts for August, September and October were above the 30-year mean monthly precipitation. However, the winter months of February and November 2009 the monthly average precipitation was below the 30-year mean monthly precipitation (Table 3).

Table 3. Average monthly precipitation (cm) at Crary, North Dakota, 2006 - 2009 (NDAWN 2010, NOAA 2010).

Month	30-year mean	2006	2007	2008	2009
[illegible]ary	1.5	0.7	0.3	0	0.3
February	1.3	1.2	0.3	0.9	0.1
[illegible]ch	2.0	2.7	3.2	1	1
April	2.3	3.5	0.5	2.3	4.2
[illegible]y	5.4	4.7	17.0	1.8	4.5
June	9.7	3.3	9.0	8.8	10.2
[illegible]	8.4	3.8	12.4	7.1	2.3
August	6.1	5.2	5.7	4.6	7.8
[illegible]ber	4.7	3.2	2.2	10.8	9.0
October	3.7	2.3	1.4	5.8	8.2
[illegible]ber	2.1	0.2	1.1	4.9	0
December	1.4	2.0	1.9	0	0
[illegible]L	48.6	32.6	55.0	48	47.6

MATERIALS AND METHODS

This study was designed using a randomized complete block, split-plot design with four blocks (replications). Within each of the four blocks, four plots were delineated and one of four treatments randomly assigned to a plot. The four treatments were clear-cut (CC), selective cut (SC), prescribed burn (PB), and a non-manipulated control (CO) (Figure 5). Each block was 100m x 50m in size with each plot treatment 25m x 50m in size. Each plot was split to test browsing by white-tailed deer. One split was accessible by deer to freely browse and the other fenced-off to eliminate deer use, with each split-plot treatment randomly assigned to one half of the entire 100 m x 50 m block (Figure 5).

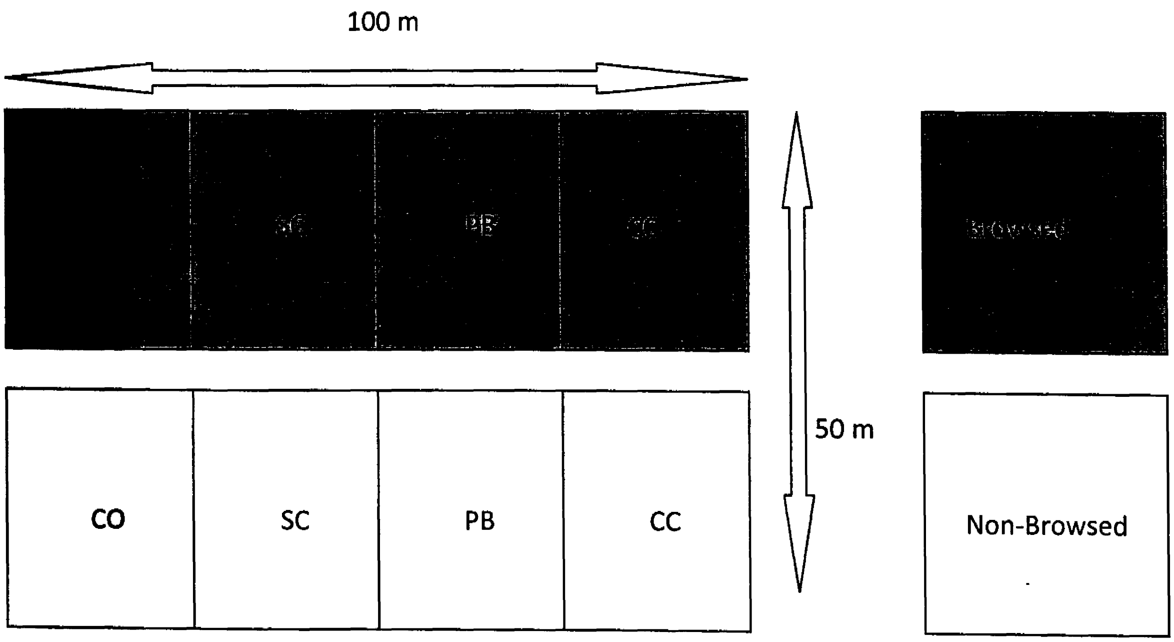

Figure 5. Example of experimental layout using a randomized complete block, split-plot design on Camp Grafton North near Devils Lake, North Dakota. Abbreviations indicate treatment: CO Control, SC Selective cut, PB Prescribed burn, and CC Clear-cut.

The clear-cut (CC) treatment was total removal of all trees at a stump height of 15-25 cm. Tree cutting occurred over the three week period 15-16 December 2006 and 6-7 January 2007. All shrubs, chokecherry and juneberry were removed to ground level in July and August, 2006.

The selective cut (SC) treatment was designed to remove all trees except bur oak at stump height of 15-25 cm in July and August, 2006. All shrubs, chokecherry and juneberry were also removed during this time and cut to ground level.

The prescribed burn (PB) treatment was conducted 26 October 2006. A backfire was used to create a black line and a head fire was allowed to move through the plot burning all fine fuels and some larger fuels such as herbaceous vegetation, dead trees and shrubs from the site. Table 4 describes the weather conditions at various times during the prescribed burn.

Table 4. Environmental conditions for October 26, 2006 during the prescribed burn treatments on Camp Grafton North near Devils Lake, North Dakota.

Replicate	**Time**	**Humidity**	**Air temperature (°C)**	**Wind (kmph)**
[redacted]	10:00 AM	81%	3.9	NW 8-16
3, 1	12:00 PM	66%	7.8	NW 8-16
[redacted]	1:00 PM	61%	8.9	NW 8-16

A deer fence was built to create the non-browsed exclosure treatment within each plot in May, 2007. This split-plot created fenced and unfenced plots of equal size to represent (free roaming) deer browsing and no browsing. The 2.3 m tall fence was constructed using 2.4 m T-posts, a double-stranded wire used as a top-anchor, and green plastic netted fencing. Zip ties were used to secure the top border of the plastic netting to the anchoring wire and ground stakes used as anchors for the bottom border of the plastic netting. A four-strand smooth-wire electric fence was assembled surrounding the plastic netting with a portable solar-paneled/battery-operated electric energizer in August, 2007. Each non-browsed exclosure was supplemented with the electric fence to prevent deer from entering the exclosures. Fencing was checked monthly and repairs made when necessary.

An aerial survey was conducted by the North Dakota Game and Fish Department on 12 March 2007 to estimate the deer population within the boundary of Camp Grafton North. This boundary included those areas within a permanent fence and totaled 365 hectares.

Since re-sprouting is a primary mechanism for oak reproduction, bur oak cookie (cross-section) samples were examined for sprouting ability of felled trees in relation to age, diameter outside of the bark (DOB) and season of cut. Samples were collected in the summer 24 and 36 months after treatment (MAFT). One hundred sixteen samples were collected; 63 samples from a cleared area on the north side of the main entrance road (summer harvest) and 53 samples from a cleared area on the south side of the main entrance road (winter harvest). The stumps were chosen at random from both sites.

Bur oak stump sprouts were counted on each stump and recorded. Tree rings were counted from each cookie to determine age. Diameter outside bark was measured at right angles using a grid. The frequency distribution of the ages, bark outside diameter, and number of sprouts (with each category containing a sample size of one hundred sixteen samples) from the stump samples were analyzed using the Kolmogorov-Smirnov test for normal distribution [$D = \max_{1 \leq i \leq N} (F(Y_i) - i/N, i/N - F(Y_i))$]. Frequency samples that were not different ($P>0.05$) did not differ from a normal distribution ($p>0.05$) (Chakravart et al. 1967).

Measurements

Mature tree [diameter at breast height (DBH) ≥ 10cm] density was collected pre-treatment in June, 2006 and 12 MAFT in June, 2007. Mature tree data was collected using one 40.5m^2 sample plot per treatment placed in the center of the treatment area. Plant species composition was collected within each sample plot; DBH and estimated height recorded for every mature tree.

Sapling (height > 13.7dm, DBH 2.5cm – 10cm) data was collected pre-treatment in June and July, 2006 and repeated 12, 24, and 36 MAFT. Sapling density was collected within four sample plots, each measuring 40.5m^2 (Figure 6). Two transects were located in each treatment plot, with two of the four sample plots located on each transect. The starting point of the two transects in each plot were 8 and 16m along the north edge of the treatment, running parallel to each other at a length of 25m. Data collected included plant species, DBH of every tree, and estimated height of every fourth tree.

Seedling (height < 13.7dm, or height > 13.7dm and DBH < 2.5cm) data was collected pre-treatment in June, 2006 and repeated 12, 24, and 36 MAFT. Seedling density was collected from four sample subplots, each measuring 4.1m^2. Seedling sample plots were located concentric with the sapling sample plots (Figure 6). Species and number of seedlings were recorded. Seedlings were distinguished from sprouts based on the origin. Seedlings are from seeds that germinated in the past few years, while sprouts arise from stumps of recently-harvested mature trees.

Sprout density was collected after treatments were implemented in 2007. Sprouts occurred in clumps, with differing numbers of clumps per stump and differing numbers of sprouts per clump. Sprout counts on the selective cut, prescribed burn, and control were recorded from the base of live, mature bur oak trees; however, sprout counts from the clear-cut were from stumps of cut trees, as no mature bur oak trees remained in this treatment. Sprout data was collected in the same sample subplots used for saplings. Height of tallest sprouts was determined for each browsed and non-browsed subplot.

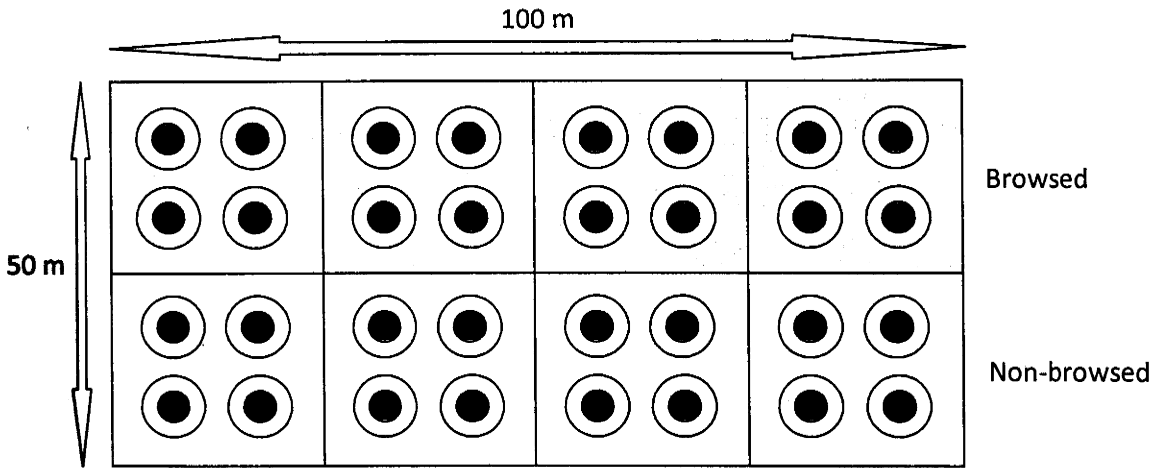

Figure 6. Diagrammatic representation of the sampling design for saplings and seedlings within each treatment on Camp Grafton North near Devils Lake, ND. Seedling subplot is represented by the (smaller black circle), and the sapling and sprout subplots represented by the (larger white circle). The seedling subplot is located within the sapling and sprout subplot.

Herbaceous vegetation data were collected using two methods. The first method involved one 0.25 m^2 frame with a 0.10 m^2 nested frame recorded every two meters along each 25 m transects (2) per treatment for a total of 24 frames per treatment plot. Frame data included graminoid species frequency (presence or absence) within the 0.10 m^2 nested frame and forb species density (number of each plant species) within the 0.25 m^2 frame. Herbaceous species richness and evenness, and herbaceous plant species diversity were determined using the 0.25 m^2 frame method. The second method involved collecting understory herbaceous vegetation using the ten-pin point frame as modified by Smith

(1959) and described by Mueller-Dombois and Ellenberg (1974). Point frames were collected every meter along each 25 m transect (2) for a total of 50 frames or 500 points. Point frame data was used to determine plant species basal cover, litter, bare ground, plant species richness and evenness, and plant species diversity (Levy and Madden 1933).

Statistical Analysis

Seedling density, clump density, sprouts/clump, height of sprouts, sapling density, basal cover, alpha species richness, alpha species evenness and alpha species diversity were analyzed using PROC MIXED repeated measures in SAS (SAS Inst. Inc, Cary, NC). The four treatments (CC, SC, PB and CO) were treated as the whole plot factor while the browsed or non-browsed plots level treated as the split plot. Year effects were treated as repeated measures and were modeled with an autoregressive (1) covariance. All effects were treated as fixed. Degrees of freedom were calculated using the Satterthwaite approximation when year effects were included in the analysis. Mean separations test was performed using the LSMEANS procedure with the Tukey adjustment. A p-value ≤ 0.05 was used to test for significant differences for all data.

Species diversity, a combination of richness and evenness, was calculated using Shannon-Wiener's index (McCune and Grace 2002). The Shannon-Wiener ($H'=\Sigma P_i \ln P_i$) index of diversity was used as a measure of diversity across treatments. P_i is the proportion of each species in the sample.

RESULTS

Herbaceous Vegetation

Richness

Seventy-one plant species were found on the study treatments plots during this four year project (Appendix A). There was a treatment, year and browsing-by-year interaction for herbaceous species richness (Table 5). Species richness was greater ($P \leq 0.05$) on the CC treatment than CO and SC during the pre-treatment year 2006 and remained greater in 2007, 2008 and 2009 (Figure 7). The CC and PB treatments had similar number of plant species pre-treatment in 2006 and 12 MAFT (Figure 7). However, species richness was lower on the PB compared to the CC treatment 24 and 36 MAFT. The PB and SC treatments had similar species richness to the CO during the pre-treatment year, 24 and 36 MAFT.

Table 5. Herbaceous species richness F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	0.06	0.80
Treatment	6.10	0.005
Year	36.43	0.0001
Browsing x Year	2.60	0.05
Treatment x Year	0.43	0.92
Browsing x Treatment	0.36	0.78
Browsing x Treatment x Year	0.34	0.96

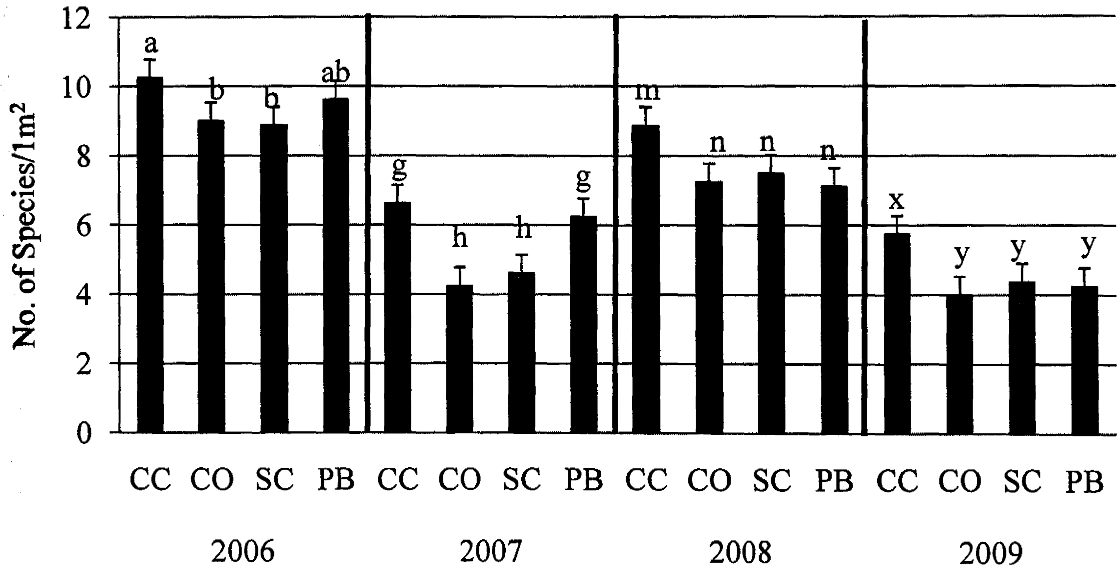

Figure 7. Herbaceous species richness by treatment on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Treatments within year with same letter are not different (P>0.05).

Species richness was not different (P>0.05) between browsed (B) and un-browsed (UnB) plots pre-treatment in 2006, with both the B and UnB decreasing (P≤0.05) 12 MAFT (Figure 8). Species richness on the B plots was lower (P≤0.05) 24 and 36 MAFT. In comparison, species richness increased starting 12 MAFT on the UnB plots, with no differences in species richness found 24 and 36 MAFT on these plots (Figure 8).

Evenness

There was a year and browsing-by-year interaction for herbaceous species evenness (Table 6). Species evenness was not different (P>0.05) between B and UnB plots pre-treatment in 2006 (Figure 9). Plant species evenness decreased (P≤0.05) on both B and UnB plots 12 MAFT. Species evenness was similar (P>0.05) to pre-treatment levels 24

MAFT on the B and UnB plots. However, evenness was lower (P≤0.05) than pre-treatment levels 36 MAFT on B plots, whereas UnB was similar (P>0.05) to pretreatment levels 36 MAFT.

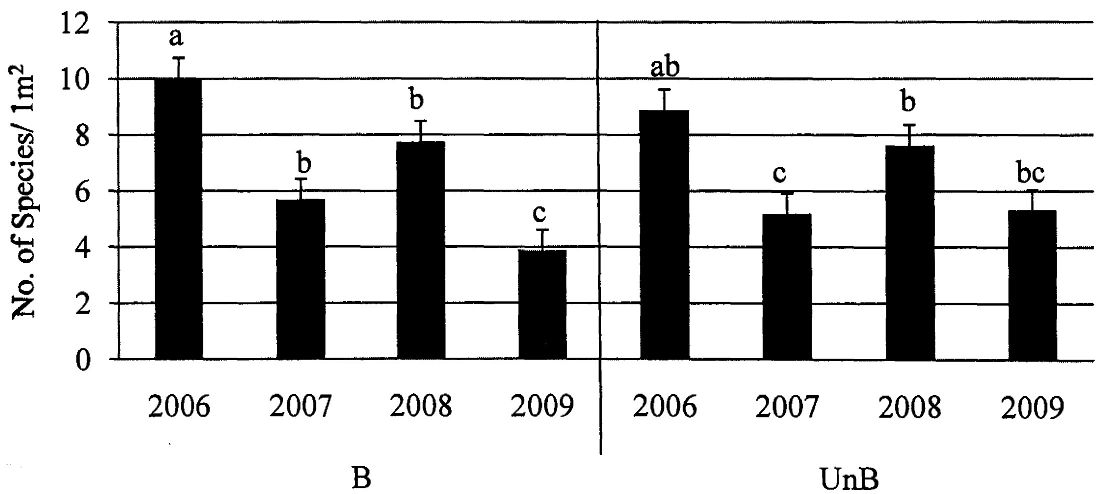

Figure 8. Herbaceous species richness by year on browsed (B) and unbrowsed (UnB) plots **on** Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same **letter** within treatment are not different (P>0.05).

Table 6. Herbaceous species evenness F and P values for each variable tested on Camp **Grafton** North, 2006 – 2009.

Effect	**F Value**	**P Value**
Browsing	2.03	0.12
Treatment	0.49	0.53
Year	46.88	0.0001
Browsing x Year	5.23	0.003
Treatment x Year	1.29	0.26
Browsing x Treatment	0.55	0.66
Browsing x Treatment x Year	1.31	0.25

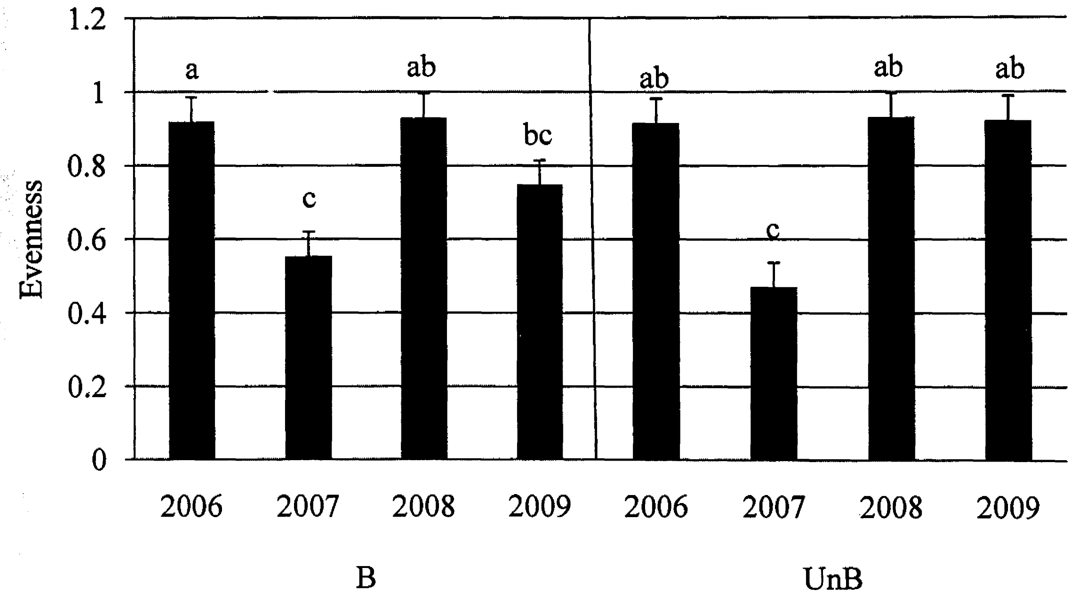

Figure 9. Herbaceous species evenness by year on browsed (B) and unbrowsed (UnB) plots on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter within treatment are not different (P>0.05).

Diversity

There was a treatment, year and browsing-by-year interaction for herbaceous species diversity (Table 7). Herbaceous species diversity was not different (P>0.05) between treatments in the pre-treatment year, 12 and 24 MAFT (Figure 10). Species diversity on the SC and PB treatments were not different (P>0.05) from the CO at any time during the three years following treatment application. Herbaceous species diversity was greater ($P\leq0.05$) on the CC than CO, SC and PB 36 MAFT.

Plant species diversity was not different (P>0.05) between B and UnB plots pre-treatment in 2006, with both decreasing ($P\leq0.05$) 12 MAFT (Figure 11). Species diversity on the B and UnB plots increased ($P\leq0.05$) to pre-treatment levels 24 MAFT and 36 MAFT on the UnB plots. However, species diversity decreased ($P\leq0.05$) 36 MAFT on the B plots.

Table 7. Herbaceous species diversity F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	0.58	0.50
Treatment	3.37	0.04
Year	47.92	0.0001
Browsing x Year	3.85	0.01
Treatment x Year	0.60	0.80
Browsing x Treatment	0.28	0.84
Browsing x Treatment x Year	0.17	1.00

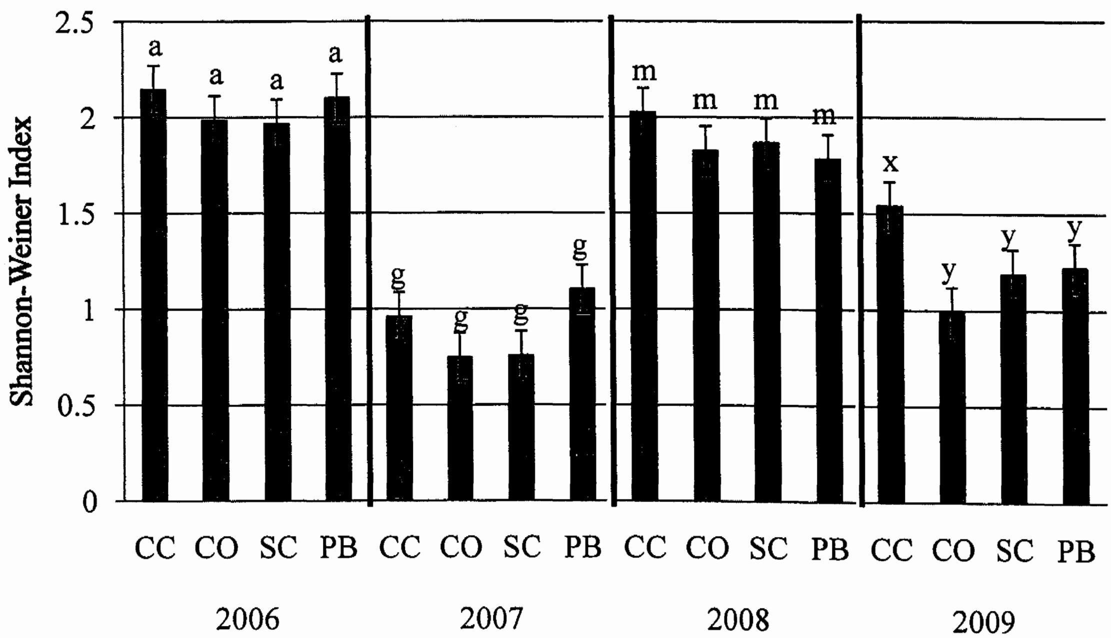

Figure 10. Herbaceous species diversity by treatment on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Treatments within year with same letter are not different (P>0.05).

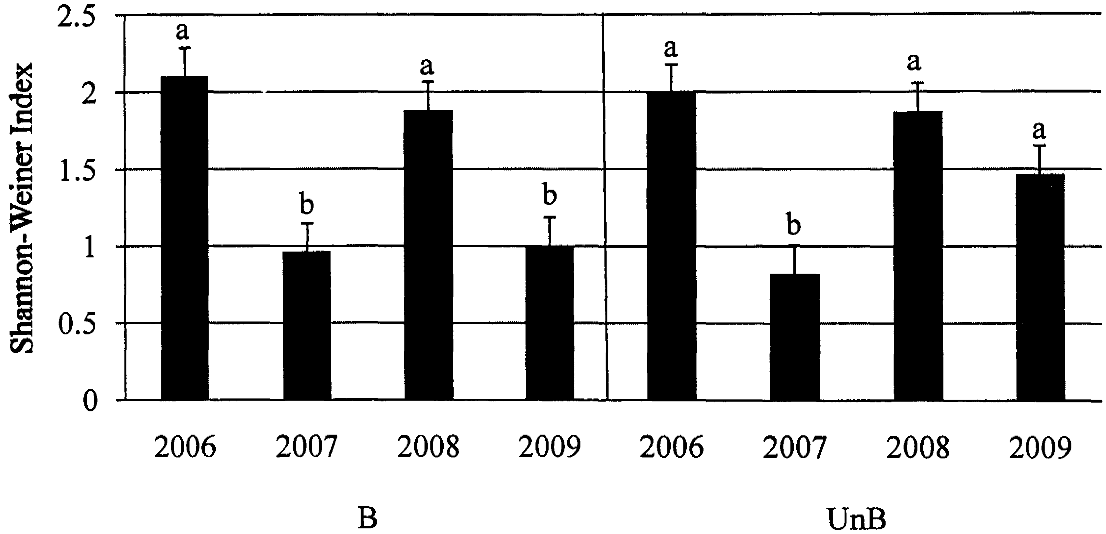

Figure 11. Herbaceous species diversity by year on browsed (B) and unbrowsed (UnB) plots on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter within treatment are not different (P>0.05).

Graminoids

Sixteen graminoid species were recorded, with Sprengel's sedge the dominant native and Kentucky bluegrass the dominant introduced species at frequencies of 96 and 49 percent, respectively. Little seed ricegrass was found at a frequency of 24 percent, with eastern bottlebrush grass, Virginia wildrye and smooth brome at frequencies of 15, 25 and 27 percent, respectively.

Eastern bottlebrush grass, smooth brome and Virginia wildrye frequency was not different (P>0.05) between browsing plots (browse and un-browsed) and among treatments (P>0.05). There was no interaction between browsing and treatment (P>0.05), browsing and year (P>0.05), treatment and year (P>0.05), and among browsing, treatment and year (P>0.05) for these grass species. However, eastern bottlebrush grass, smooth brome, and Virginia wildrye frequency was different (P≤0.05) among years.

The frequency of occurrence of smooth bromegrass and Virginia wildrye decreased ($P \leq 0.05$) on all treatments 12 MAFT (Figure 12). Smooth bromegrass frequency at 36 MAFT was similar to the levels achieved 12 MAFT; however, Virginia wildrye was absent from all study treatments 36 MAFT. Although yearly fluctuations occurred for eastern bottlebrush grass, no difference ($P>0.05$) in frequency was found from the pre-treatment year in 2006 and 36 MAFT.

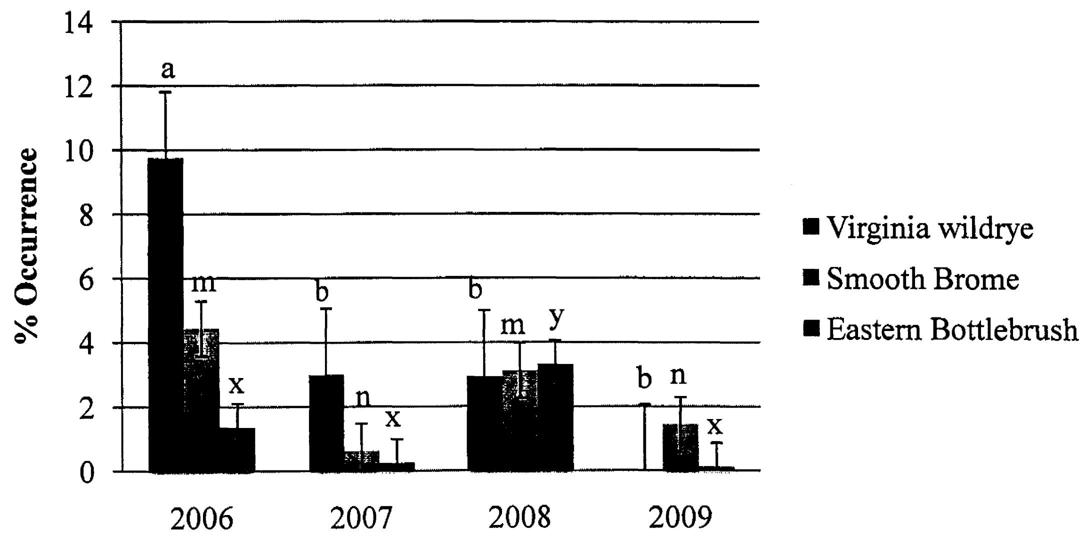

Figure 12. Eastern bottlebrush grass, smooth brome, and Virginia wildrye frequency on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter within each grass species are not different ($P>0.05$).

Sprengel's Sedge

There was a year and treatment by year interaction for Sprengel's sedge frequency (Table 8). Sprengel's sedge frequency did not change ($P>0.05$) on the CO 36 MAFT (Figure 13). Sprengel's sedge frequency decreased ($P \leq 0.05$) 24 MAFT on the PB treatment and 36 MAFT on the CC and SC treatments. All the other treatments had a decreasing frequency 36 MAFT while the CO did not change in frequency.

Table 8. Sprengel's sedge F and P values for each variable tested on Camp Grafton North **2006** – 2009.

Effect	F Value	P Value
Browsing	0.06	0.82
Treatment	2.22	0.12
Year	70.27	0.0001
Browsing x Year	1.26	0.29
Treatment x Year	3.27	0.002
Browsing x Treatment	0.09	0.96
Browsing x Treatment x Year	0.13	1.00

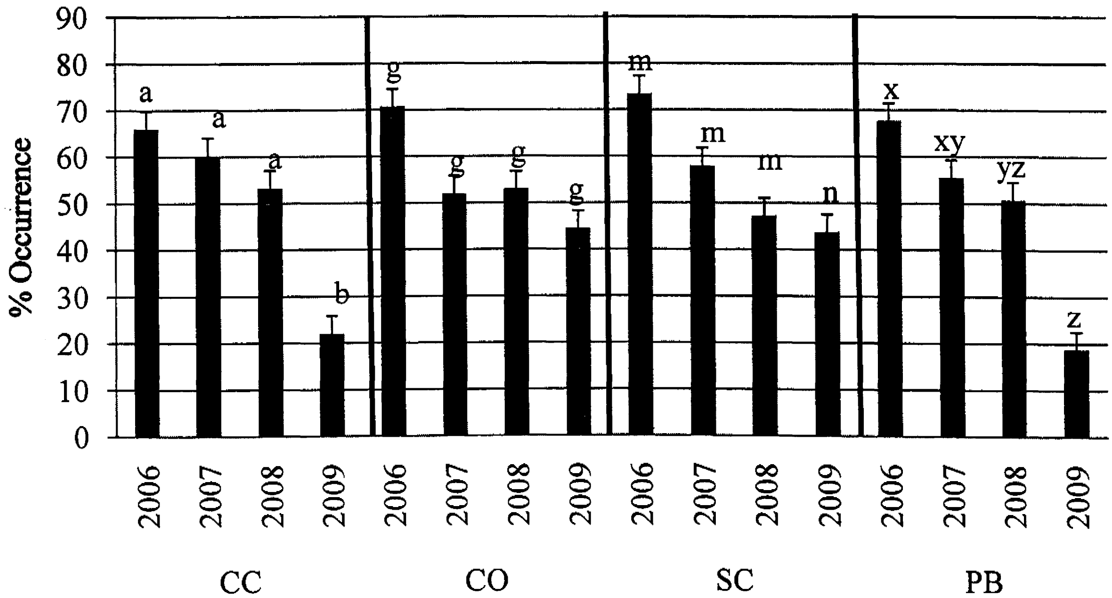

Figure 13. Sprengel's sedge frequency (percent occurrence) by treatment on Camp Grafton **North** near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, **SC** selective cut, and PB prescribed burn. Treatments within year with same letter are not **different** (P>0.05).

Kentucky Bluegrass

There was a year and browsing-by-year interaction for Kentucky bluegrass frequency (Table 9). Kentucky bluegrass frequency was not different among years on the B plots. However, after an initial decrease in frequency 12 MAFT, Kentucky bluegrass frequency increased 36 MAFT on the UnB plots (Figure 14).

Table 9. Kentucky bluegrass frequency F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	1.10	0.37
Treatment	1.20	0.34
Year	8.66	0.0001
Browsing x Year	3.60	0.02
Treatment x Year	0.88	0.54
Browsing x Treatment	0.48	0.70
Browsing x Treatment x Year	0.43	0.91

Little-seed Ricegrass

There was a browsing, year and browsing-by-year interaction for little-seed ricegrass frequency (Table 10). Little-seed ricegrass frequency was not different ($P > 0.05$) between pre-treatment year 2006 and 12 MAFT on the B and UnB plots; however, frequency increased ($P \leq 0.05$) on both plots 24 MAFT (Figure 15). The low frequency of little-seed ricegrass pretreatment and 12 MAFT could be due to a misidentification, or lack of identification. Frequency of little-seed ricegrass decline to pre-treatment levels on the B plots; however, remained greater ($P \leq 0.05$) than pre-treatment levels on the UnB plots. Little-seed ricegrass frequency increased ($P \leq 0.05$) on all treatments, irrelevant of white-tailed deer browsing in this study after two years.

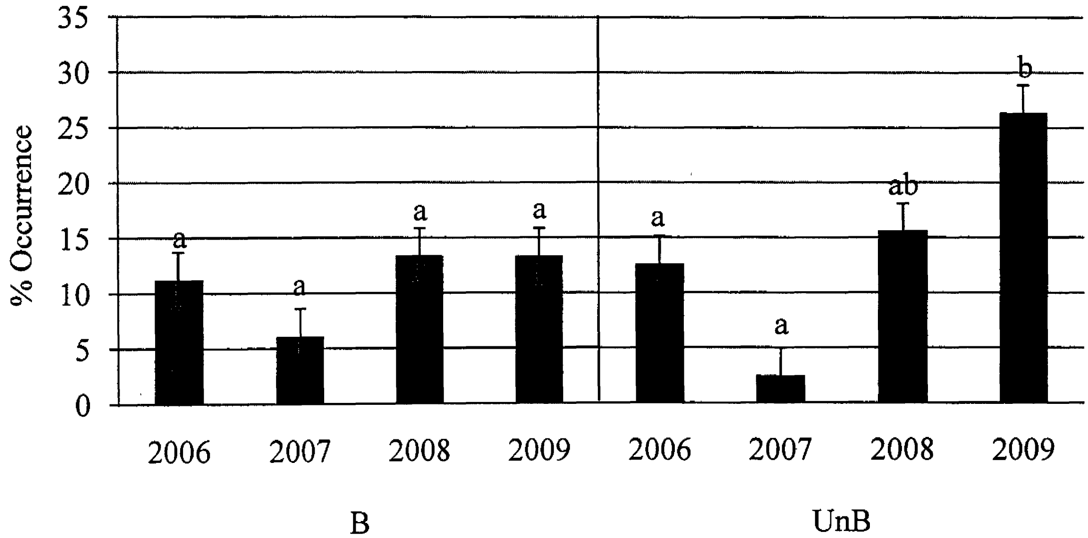

Figure 14. Kentucky blue grass frequency by year on browsed (B) and unbrowsed (UnB) plots on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter within treatments are not different (P>0.05).

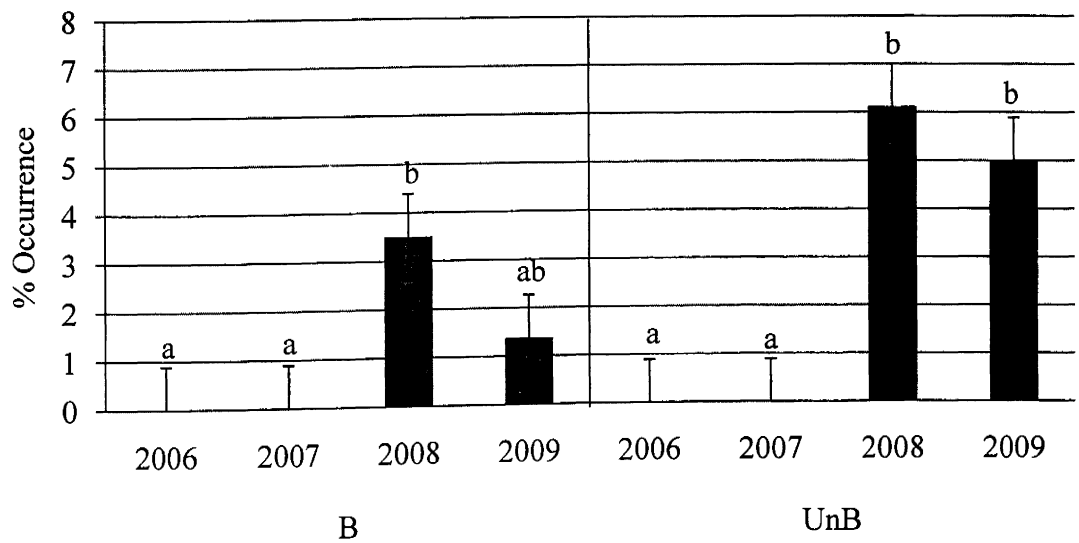

Figure 15. Little seed ricegrass frequency by year on browsed (B) and unbrowsed (UnB) plots on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter within treatments are not different (P>0.05).

Table 10. Little-seed ricegrass frequency F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	2.11	[illegible]
Treatment	0.34	0.80
Year	22.31	0.0001
Browsing x Year	3.27	0.03
Treatment x Year	0.19	0.99
Browsing x Treatment	0.22	0.88
Browsing x Treatment x Year	0.26	[illegible]

Bare Ground and Litter

Bare Ground

There was a year interaction ($P \leq 0.05$) for bare ground frequency (Table 11). Bare ground increased ($P \leq 0.05$) (Figure 16) 12 MAFT, but returned to pretreatment levels 24 and 36 MAFT. The highest occurrence of bare ground occurred on all treatments in 2007.

Litter

There was a year and treatment by year interaction ($P \leq 0.05$) for litter frequency (Table 12). Litter frequency was similar on all treatments during all years except CO. Litter frequency increased ($P \leq 0.05$) in the CC 12 MAFT, and SC and PB 24 MAFT; returning to pretreatment levels on all three treatments 36 MAFT (Figure 17). Only the CC treatment increased litter occurrence 12 MAFT, with the SC and PB similar to the CO in litter frequency throughout the study. Litter frequency of CC responded similarly to the CO after 24 months.

Table 11. Bare ground density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	0.60	0.50
Treatment	0.71	0.56
Year	17.70	0.0001
Browsing x Year	0.95	0.42
Treatment x Year	0.64	0.76
Browsing x Treatment	0.31	0.82
Browsing x Treatment x Year	0.71	0.70

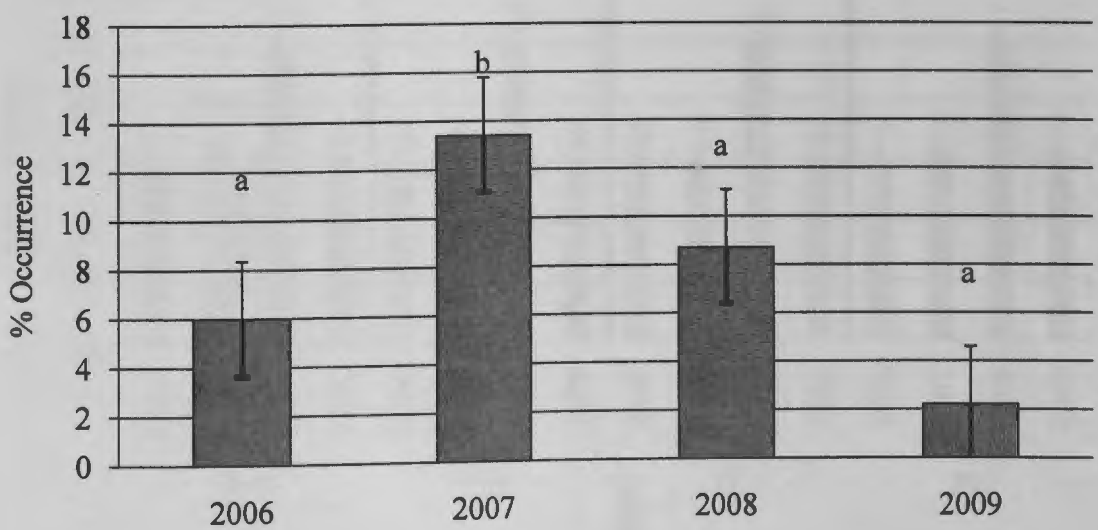

Figure 16. Bare ground frequency by year on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter are not different (P>0.05).

Table 12. Litter frequency F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	0.07	0.81
Treatment	0.54	0.66
Year	606.47	0.0001
Browsing x Year	1.11	0.35
Treatment x Year	2.50	0.02
Browsing x Treatment	0.54	0.66
Browsing x Treatment x Year	0.75	0.66

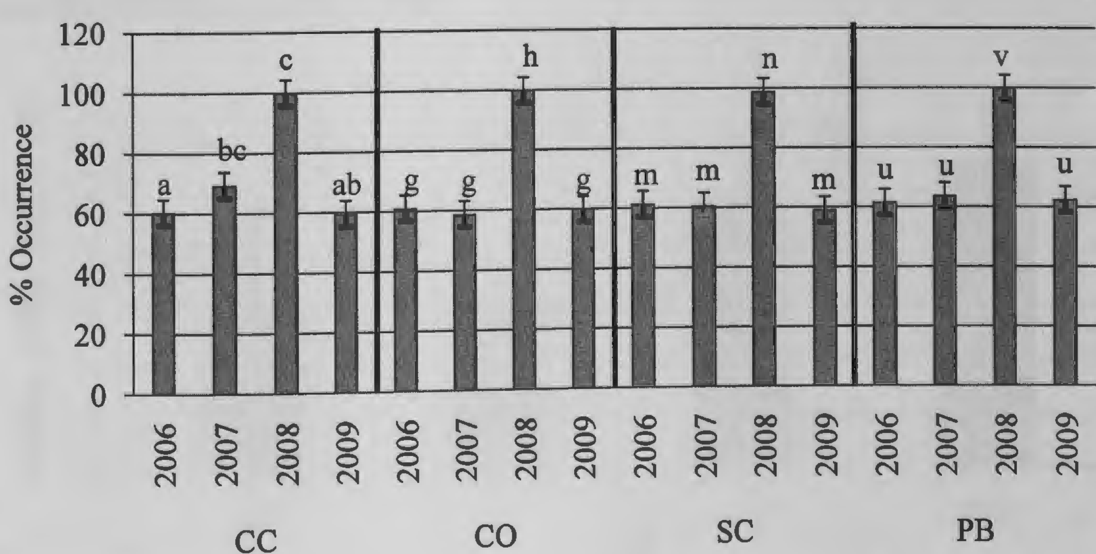

Figure 17. Litter frequency by treatment on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Years within treatments with same letter are not different (P>0.05).

Forbs

There were forty-one different forb species found on all treatments over the duration of the study (Appendix A). The most common forbs were wild sarsaparilla, American stickseed and stickywilly. Annual and perennial forb frequency increased ($P \leq 0.05$) during the study on all treatments. Frequency of annuals averaged 24 percent while perennials averaged 38 percent. The dominant understory forb on Camp Grafton North was wild sarsaparilla at an average frequency of 76 percent. Canada thistle was the dominant introduced forb at an average frequency of 34 percent.

Annual Forbs

There was a year interaction for annual forb species (Table 13). Annual forb species increased ($P \leq 0.05$) from six plants per m^2 in 2006 to 11.5 plants per m^2 36 MAFT (Figure 18). Number of annual forb species increased ($P \leq 0.05$) from 24 to 36 MAFT, irrelevant of treatment and browsing.

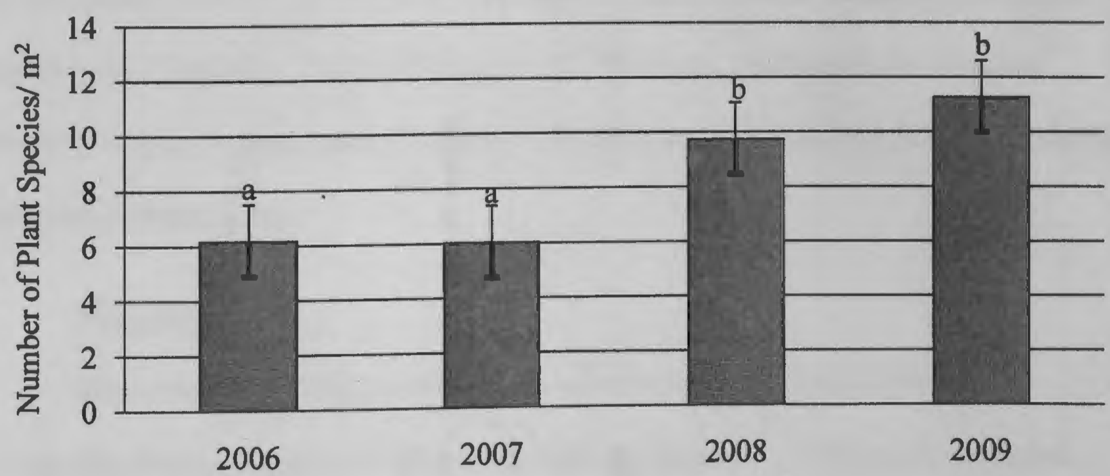

Figure 18. Mean number of annual forb species on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter are not different (P>0.05).

Table 13. Mean richness of annual forb species F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	0.18	0.70
Treatment	0.87	0.47
Year	5.03	0.003
Browsing x Year	0.10	0.96
Treatment x Year	0.71	0.70
Browsing x Treatment	0.99	0.42
Browsing x Treatment x Year	1.30	0.26

Perennial Forbs

There was a year interaction ($P \leq 0.05$) for perennial forb species (Table 14). Perennials increased ($P \leq 0.05$) from 14.5 plants/ m^2 in 2006 to 33 plants/ m^2 12 MAFT and were 27.7 plants/ m^2 36 MAFT (Figure 19). However, perennial forbs remained similar ($P \leq 0.05$) between years 2, 3 and 4 with all years significantly different ($P \leq 0.05$) than pretreatment levels.

Wild Sarsaparilla

There was a browsing, treatment and year interaction ($P \leq 0.05$) for wild sarsaparilla density (Table 15). Wild sarsaparilla increased ($P \leq 0.05$) on all treatments except CC 12 MAFT (Figure 20). Although wild sarsaparilla density increased on the PB, SC and CO 12 and 24 MAFT (Figure 20); they are declined ($P \leq 0.05$) to similar ($P > 0.05$) levels observed in 2006, or 36 MAFT. The CC treatment was the only treatment to reduce

($P \leq 0.05$) wild sarsaparilla densities 36 MAFT (Figure 20) and remain lower ($P \leq 0.05$) than all other treatments 36 MAFT (Figure 21).

Table 14. Richness of perennial forb species F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	2.64	0.20
Treatment	2.30	0.11
Year	14.68	0.0001
Browsing x Year	1.00	0.40
Treatment x Year	1.30	0.25
Browsing x Treatment	0.26	0.86
Browsing x Treatment x Year	0.43	0.92

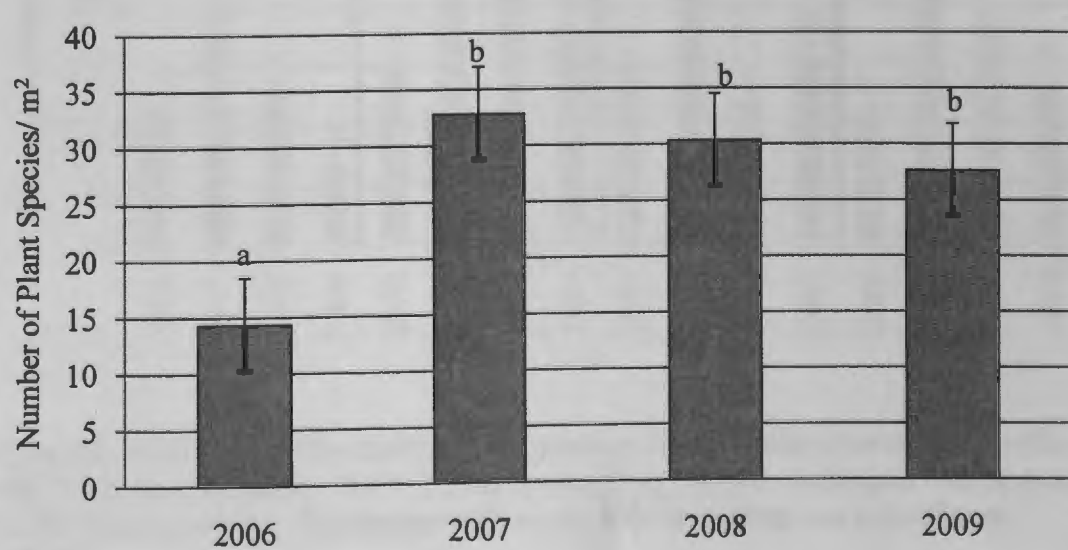

Figure 19. Mean number of perennial forb plant species on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter are not different (P>0.05).

Table 15. Wild sarsaparilla density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	7.10	0.08
Treatment	5.03	0.01
Year	12.01	0.0001
Browsing x Year	1.04	0.38
Treatment x Year	1.40	0.42
Browsing x Treatment	0.52	0.67
Browsing x Treatment x Year	0.83	0.60

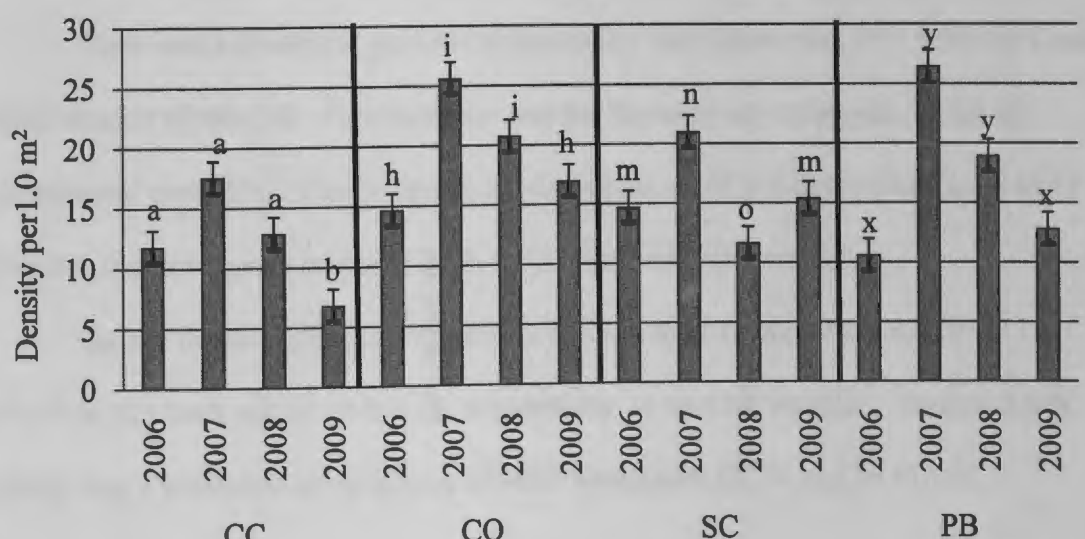

Figure 20. Wild sarsaparilla density by treatment on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Treatments within year with same letter are not different (P>0.05).

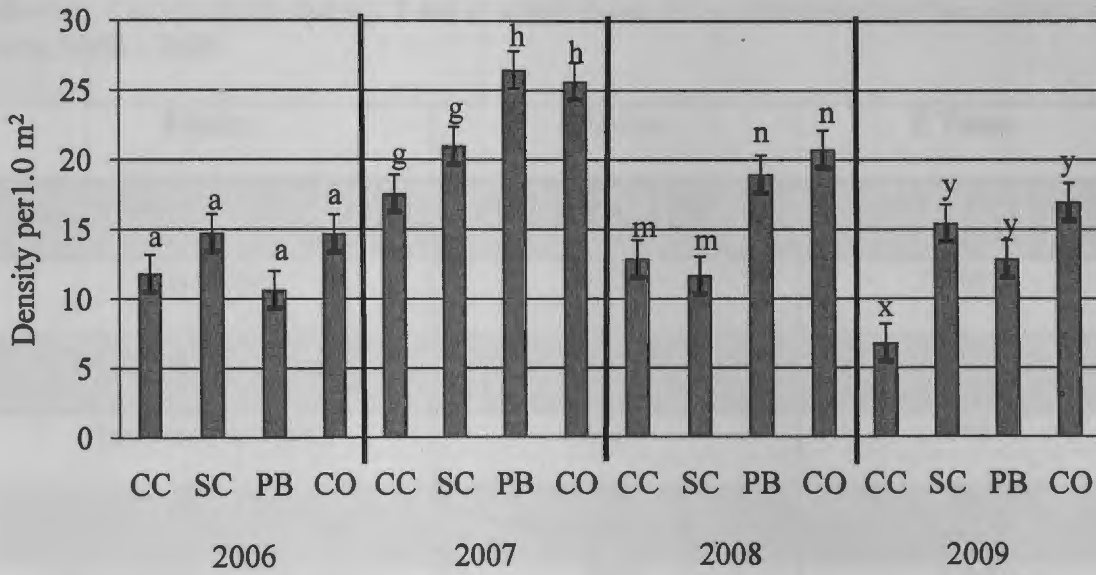

Figure 21. Wild sarsaparilla density by year on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Treatments with same letter are not different (P>0.05).

Canada Thistle

There was a treatment, year and treatment by year interaction ($P \leq 0.05$) for Canada thistle density (Table 16). Canada thistle was not found in any treatment during the pretreatment year 2006. Canada thistle density increased ($P \leq 0.05$) on the CC 24 MAFT; however, did not change on the SC, PB, or CO 36 MAFT (Figure 22).

Canada thistle density increased ($P \leq 0.05$) in the CC from 2 plants per m^2 12 MAFT to 12 plants per m^2 36 MAFT; respectively, or an 83% increase. Canada thistle density was 2 plants per m^2 or less on all other treatments 12, 24 and 36 MAFT.

Table 16. Canada thistle density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	0.14	0.73
Treatment	12.30	0.0001
Year	8.66	0.0001
Browsing x Year	0.50	0.68
Treatment x Year	6.57	0.0001
Browsing x Treatment	0.04	0.99
Browsing x Treatment x Year	0.45	0.90

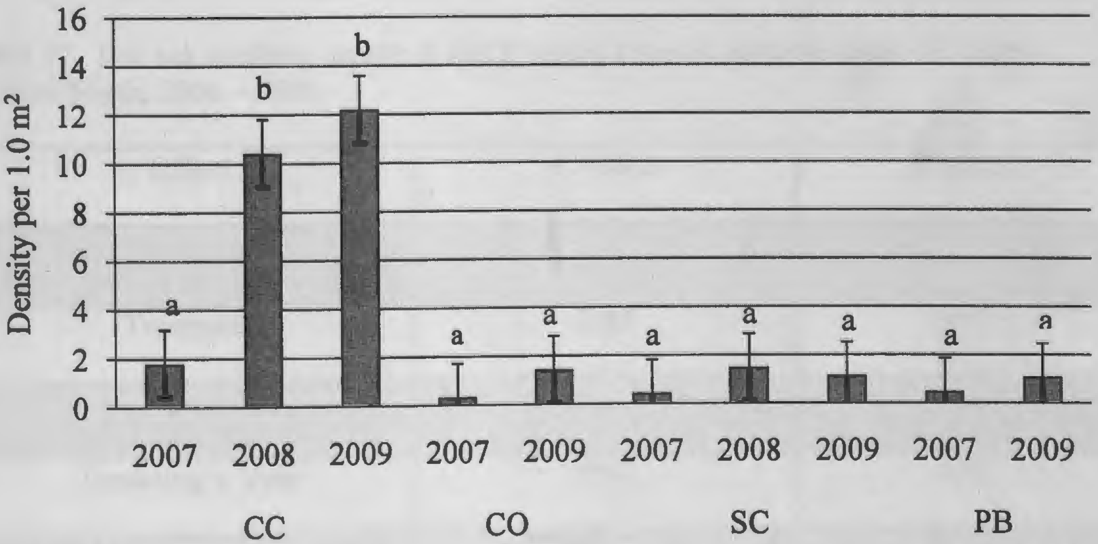

Figure 22. Canada thistle density by treatment on Camp Grafton North near Devils Lake, North Dakota, 2007 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Years within treatments with same letter are not different (P>0.05).

Tree Composition and Impacts

Seedlings

There were six trees and eight shrub species found on all treatments over the duration of the study. The most prevalent seedling tree and shrub species found were bur oak, green ash, American elm, chokecherry and western snowberry.

Bur Oak

There was a year effect ($P \leq 0.05$) for bur oak seedling density (Table 17). The CC treatment was trending to increasing ($P = 0.08$) bur oak seedlings 12 MAFT and 36 MAFT; whereas, the SC and PB were similar ($P > 0.05$) to the CO (Figure 23). Bur oak seedling density increased ($P \leq 0.05$) from 610 seedlings per hectare pretreatment in 2006 to 23479 seedlings per hectare 12 MAFT, 11454 seedlings per hectare 24 MAFT and 17,900 36 MAFT (Figure 24).

Table 17. Bur oak seedling density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	0.75	0.59
Treatment	2.63	0.08
Year	39.25	0.0001
Browsing x Year	0.35	0.79
Treatment x Year	0.57	0.82
Browsing x Treatment	1.04	0.40
Browsing x Treatment x Year	0.87	0.55

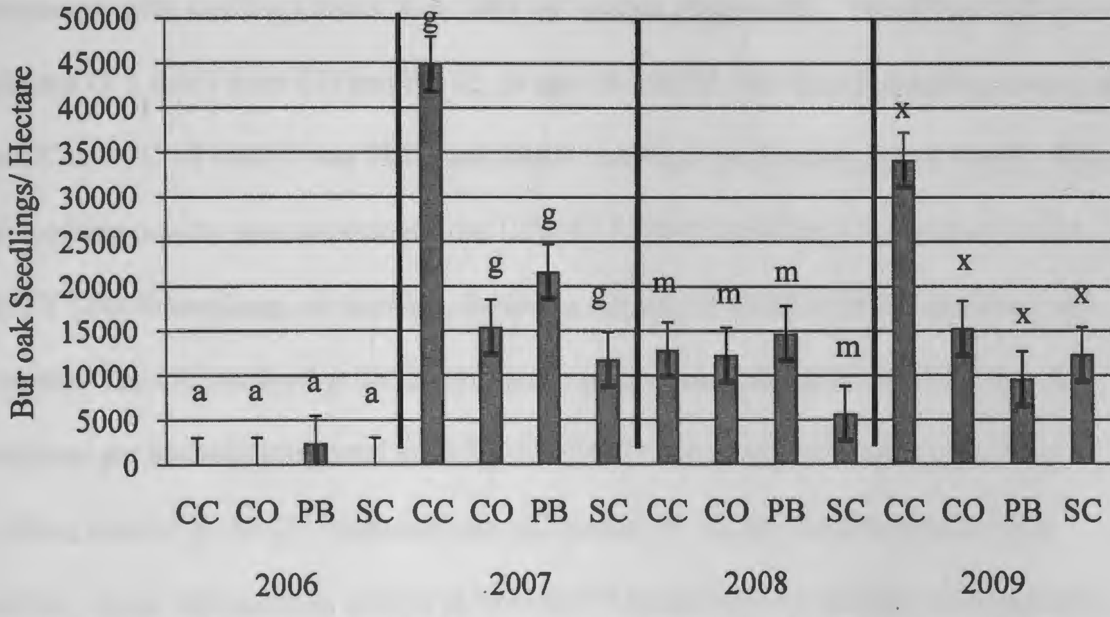

Figure 23. Bur oak seedling density by treatment on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Years with same letter are not different (P>0.05).

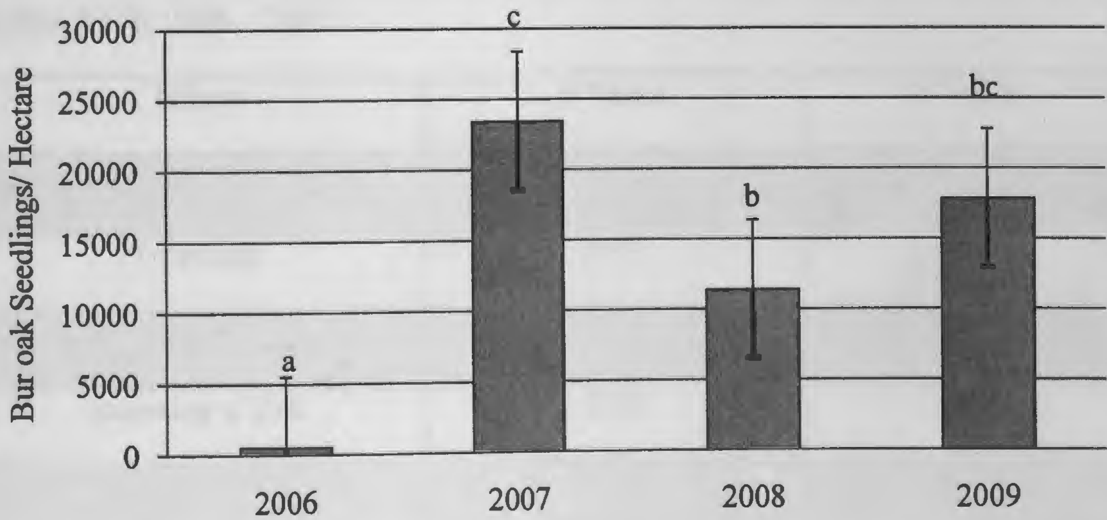

Figure 24. Bur oak seedling density by year on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter are not different (P>0.05).

Green Ash

There were a treatment and year effects ($P \leq 0.05$) for green ash seedling density (Table 18). Pretreatment green ash seedlings were not different ($P > 0.05$) among treatments, with seedlings fewer than 5000 per hectare (Figure 25). The SC and CC were different ($P \leq 0.05$) from CO and PB 12, 24 and 36 MAFT. Green ash seedling density on the CC and SC 12 MAFT was 38211and 26423 seedlings per hectare, respectively. Select cut seedling density was consistent from 12 MAFT (26422 seedlings per hectare) to 24 MAFT (24970 seedlings per hectare), declining slightly 36 MAFT (19985 seedlings per hectare). The CC declined at 24 (27642 seedlings per hectare) and 36 MAFT (18,336 seedlings per hectare) compared to 12 MAFT (38211 seedlings per hectare); however, seedling density in the CC treatment was still greater ($P \leq 0.05$) than the pretreatment density. Green ash seedling density in CO and PB treatments was similar for all years of the study (Figure 25).

Table 18. Green ash seedling density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	2.32	0.23
Treatment	6.47	0.004
Year	12.32	0.0001
Browsing x Year	2.12	0.11
Treatment x Year	1.46	0.18
Browsing x Treatment	2.02	0.15
Browsing x Treatment x Year	1.40	0.20

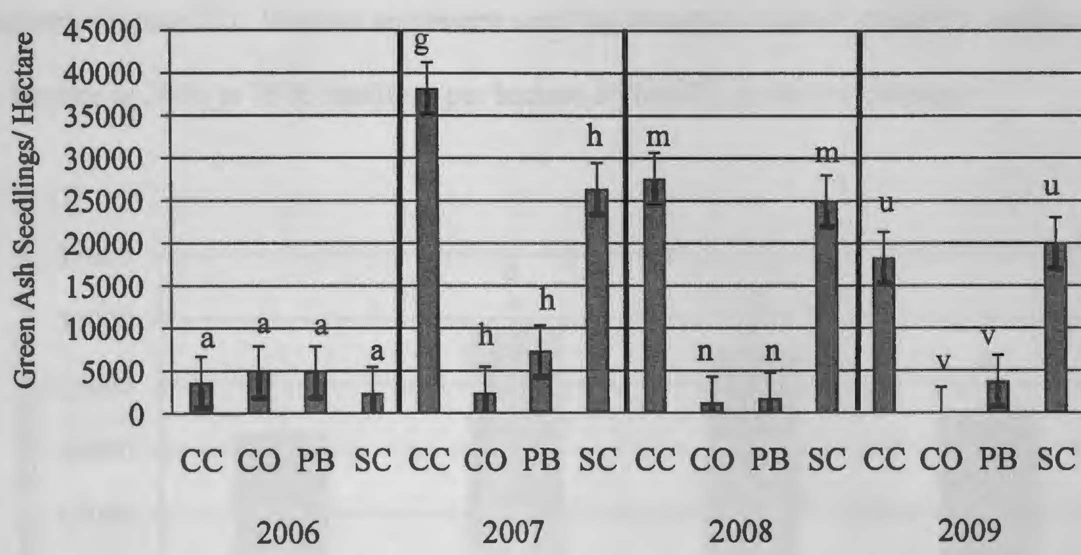

Figure 25. Green ash seedling density by treatment on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Treatments within year with same letter are not different ($P>0.05$).

Chokecherry

Chokecherry seedling density was not different ($P>0.05$) between browsed plots, among treatments or years. Chokecherry seedling density averaged 21303 seedlings per hectare pretreatment, 29420 seedlings per hectare 12 MAFT, 27448 seedlings per hectare 24 MAFT and 22357 seedlings per hectare 36 MAFT (Figure 26). Chokecherry seedling density averaged 28398 seedlings per hectare in the CC, 25546 seedlings per hectare in the CO, 18556 seedlings per hectare in the PB and 28029 seedlings per hectare in the SC.

Western Snowberry

There was a year effect for western snowberry seedling density (Table 19). Western snowberry seedling density increased ($P \leq 0.05$) from pretreatment year to 12 MAFT. Western snowberry seedling density was not different ($P > 0.05$) between years at 12, 24 and 36 MAFT, indicating the initial increase was a one-time influx independent of treatment (Figure 27). Western snowberry seedling density increased from 915 seedlings per hectare in 2006 to 7876 seedlings per hectare 36 MAFT, an 861% increase.

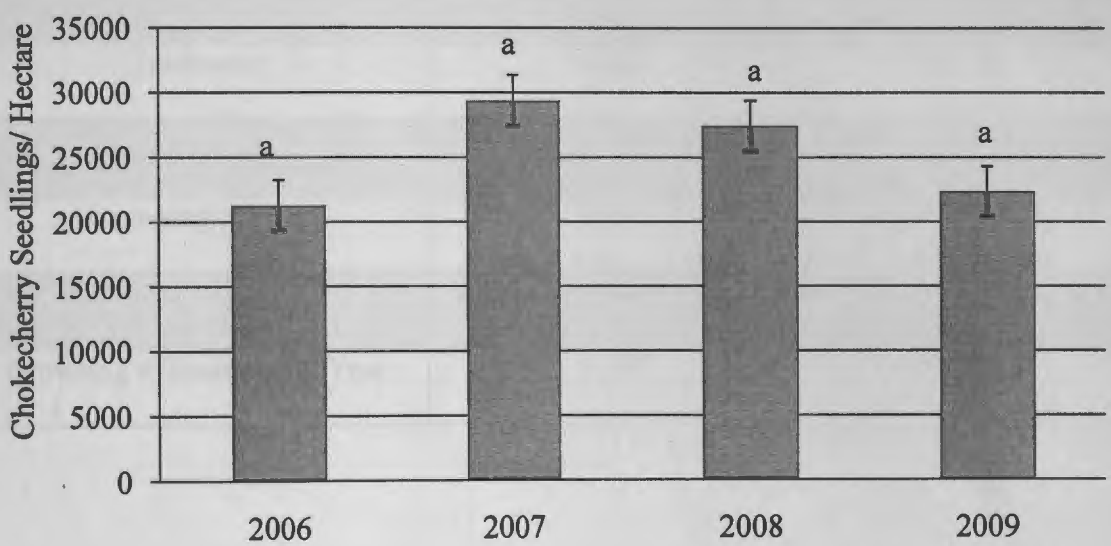

Figure 26. Chokecherry seedling density by year on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter are not different (P>0.05).

American Elm

There was a year effect and browsing-by-year interaction ($P \leq 0.05$) effect for American elm seedling density (Table 20). American elm seedling density increased ($P \leq 0.05$) in the browsed plots 12 MAFT that continued 24 and 36 MAFT (Figure 28). There was no difference ($P > 0.05$) between the pretreatment year, 2007, 2008 and 2009 on the UnB. There was no difference ($P > 0.05$) between 2007, 2008 and 2009 on the B.

American elm seedling density increased ($P \leq 0.05$) from 2388 seedlings per hectare pretreatment to 7851 seedlings per hectare 12 MAFT and 4414 seedlings per hectare 36 MAFT on the browsed plots (Figure 28). American elm seedling density did not change on UnB plots ($P > 0.05$).

Table 19. Western snowberry seedling density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	6.33	0.08
Treatment	0.37	0.78
Year	8.40	0.0001
Browsing x Year	2.35	0.08
Treatment x Year	0.58	0.81
Browsing x Treatment x Year	0.97	0.48

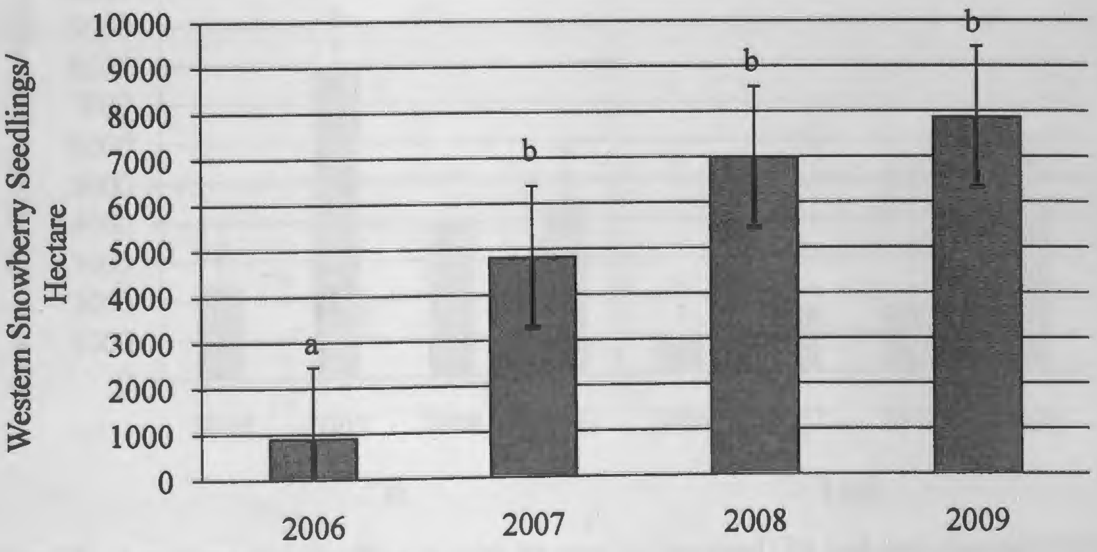

Figure 27. Western snowberry seedling density by year on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter are not (P>0.05) different.

Table 20. American elm seedling density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	2.61	0.20
Treatment	0.89	0.47
Year	5.17	0.003
Browsing x Year	2.91	0.04
Treatment x Year	1.33	0.24
Browsing x Treatment x Year	1.64	0.10

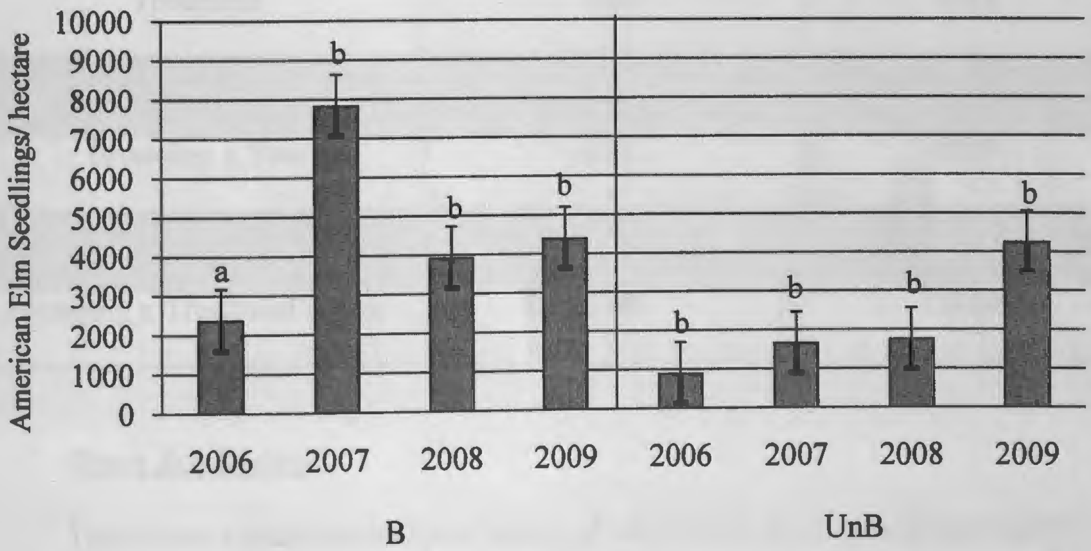

Figure 28. American elm seedling density by year on browsed (B) and un-browsed (UnB) plots on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter within treatments are not different (P>0.05).

Saplings

Bur Oak Saplings

There was a year effect ($P \leq 0.05$) on bur oak sapling density (Table 21). Saplings decreased ($P \leq 0.05$) from 5273 saplings per hectare pretreatment in 2006 to 766 saplings per hectare 36 MAFT, or a decline of 85% (Figure 29). Saplings did not decrease ($P > 0.05$) 12 to 36 MAFT, with saplings densities 1964 and 766 saplings per hectare in 2007 and 2009, respectively.

Table 21. Bur oak sapling density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	**F Value**	**P Value**
Browsing	0.57	0.51
Treatment	0.29	0.83
Year	5.12	0.003
Browsing x Year	1.02	0.39
Treatment x Year	1.30	0.25
Browsing x Treatment x Year	0.98	0.48

Green Ash Saplings

There were a treatment and year effects ($P \leq 0.05$) for green ash sapling density (Table 22). Green ash sapling density was not different ($P > 0.05$) among treatments during pretreatment in 2006 (Figure 30). The CC and SC decreased ($P \leq 0.05$) 12 (0, 0 saplings per hectare), 24 (0, 0 saplings per hectare) and 36 (1787, 0 saplings per hectare) MAFT; respectively, compared to pretreatment. Green ash sapling density remained similar to pretreatment on the PB at 7996 per hectare in 2006 to 6729 per hectare in 2009. Green ash saplings on the CO were not different (P > 0.05) between the pretreatment levels 12, 24 and 36 MAFT; 8114, 6410, 7635, and 5483 seedlings per hectare, respectively. Green ash sapling numbers increased in the CC treatment 36 MAFT.

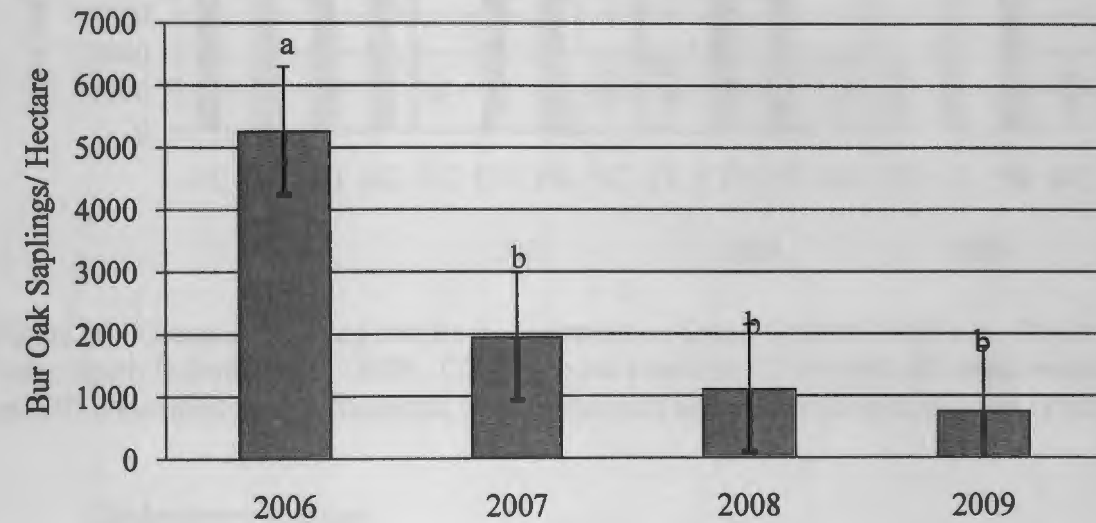

Figure 29. Bur oak sapling density on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter are not different (P>0.05).

Table 22. Green ash sapling density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	3.06	0.18
Treatment	13.14	0.0001
Year	7.62	0.0002
Browsing x Year	1.04	0.38
Treatment x Year	1.52	0.16
Browsing x Treatment	0.36	0.79
Browsing x Treatment x Year	0.87	0.58

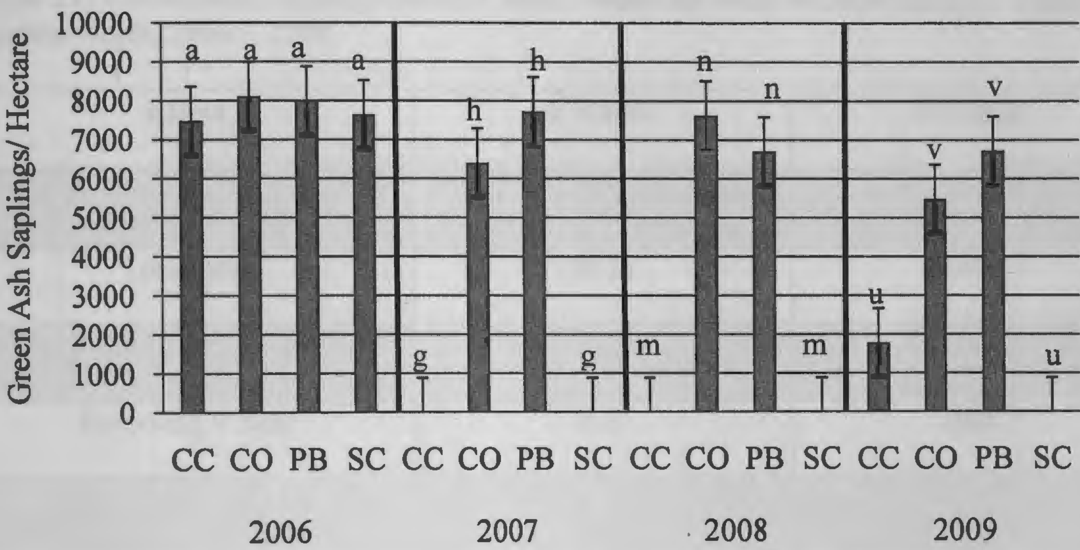

Figure 30. Green ash sapling density by treatment on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Treatments within year with same letter are not different (P>0.05).

Chokecherry Saplings

There were treatment, year and treatment-by-year effects ($P \leq 0.05$) for chokecherry sapling density (Table 23). Chokecherry sapling density was not different ($P > 0.05$) between treatments during the pretreatment year 2006 (Figure 31). The sapling density in the CO and PB increased ($P \leq 0.05$) 36 MAFT (3432 and 3656 saplings per hectare) and was greater ($P \leq 0.05$) than in the CC and SC 24 (0 and 0) and 36 (0, 963) MAFT. Chokecherry trees and saplings on the CC and SC were removed in 2006, so these lower densities were expected is subsequent years. The PB and CO sapling densities were similar between the pretreatment year in 2006 and 12 MAFT at 1659 and 1396 saplings per hectare; however, chokecherry sapling density increased ($P \leq 0.05$) 24 MAFT to 5395 and 5045 saplings per hectare on the PB and CO treatments, respectively.

Table 23. Chokecherry sapling density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	0.02	0.90
Treatment	10.71	0.003
Year	7.65	0.0002
Browsing x Year	0.57	0.63
Treatment x Year	0.23	0.0008
Browsing x Treatment	0.29	0.83
Browsing x Treatment x Year	0.32	0.98

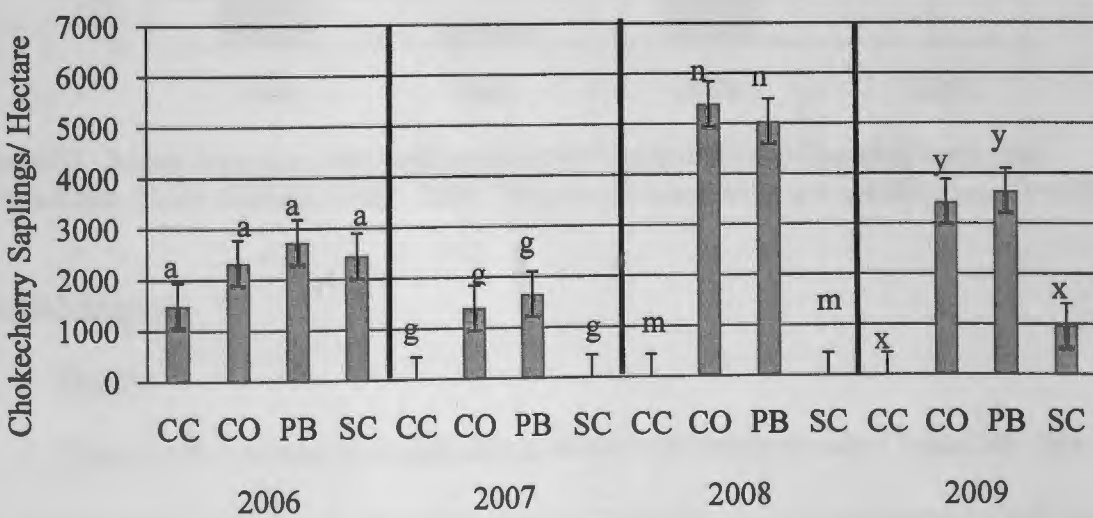

Figure 31. Chokecherry sapling density by treatment on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Treatments within year with same letter are not different (P>0.05).

American Elm Saplings

American elm sapling density was not different (P > 0.05) among treatments or between browsing plots; however, there was a year effect (P ≤ 0.05). American elm sapling density averaged 1291 saplings per hectare in the 2006, declining (P ≤ 0.05) to 0 saplings per hectare 36 MAFT (Figure 32).

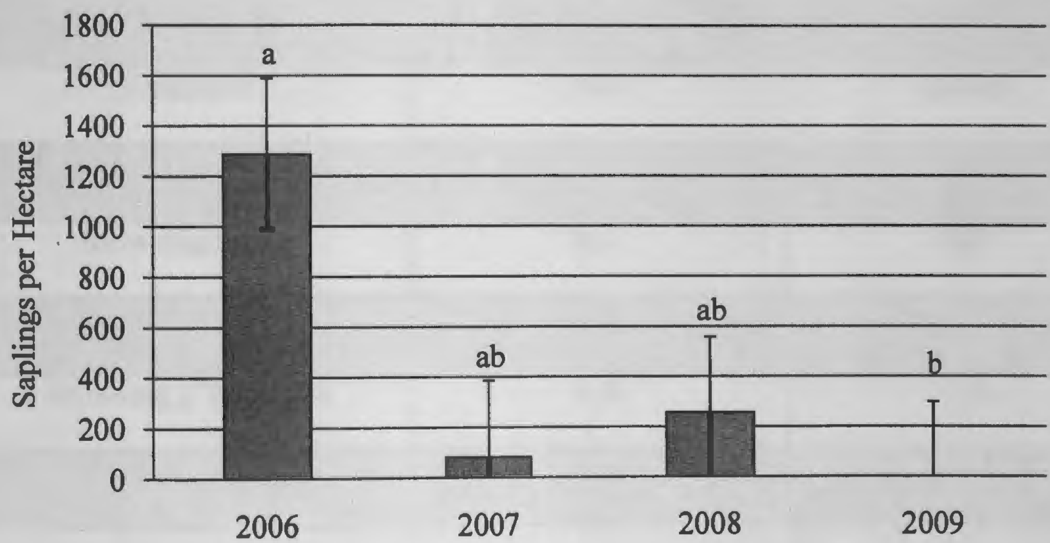

Figure 32. Mean American elm sapling density by year on Camp Grafton North near Devils Lake, North Dakota, 2006 - 2009. Years with same letter are not different (P>0.05).

Bur Oak Sprouts

Density

There were treatment and year effects for bur oak sprout density (Table 24). Bur oak sprout density was greater (P ≤ 0.05) in the CC than in all other treatments in 2007 and 2009 (Figure 33). Sprout density in the CO was not different (P > 0.05) between 2007 (2080 per hectare) and 2009 (2165 per hectare); whereas, in the PB sprouts declined (P ≤ 0.05) from 3099 in 2007 to 272 sprouts per hectare in 2009. Sprout density was not different (P > 0.05) on the PB (3079 sprouts per hectare) or SC (1114 sprouts per hectare) compared to CO (2080 sprouts per hectare) 12 MAFT. However, the number of sprouts on the PB treatment (272 sprouts per hectare) was lower ($P \leq 0.05$) than the CO treatment (2165 sprouts per hectare) 36 MAFT.

Table 24. Bur oak sprouts density F and P values for each variable tested on Camp Grafton North, 2006 – 2009.

Effect	F Value	P Value
Browsing	0.34	0.60
Treatment	9.86	0.0005
Year	5.57	0.03
Browsing x Year	0.11	0.74
Treatment x Year	0.37	0.70
Browsing x Treatment	0.40	0.76
Browsing x Treatment x Year	1.06	0.36

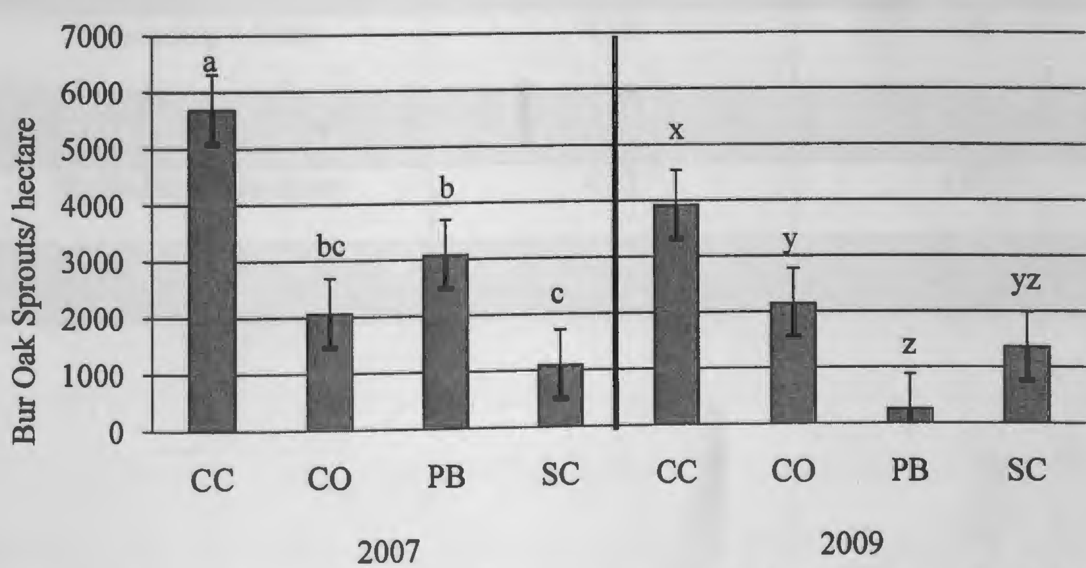

Figure 33. Bur oak sprouts density by treatment on Camp Grafton North near Devils Lake, North Dakota, 2007 and 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. Treatments within year with same letter are not different ($P>0.05$).

Sprout Height

There were browsing, treatment, treatment-by-year and browsing-by-treatment effects ($P \leq 0.05$) on bur oak sprout height (Table 25). The un-browsed CC treatment produced the tallest ($P \leq 0.05$) bur oak sprouts (Figure 34). Sprout height was greater ($P \leq 0.05$) in the UnB CC than the B CC with an average height of 124cm compared to 35cm, respectively. Bur oak sprout height in the UnB CO was ($P \leq 0.05$) greater than B CO, with an average of 55 and 16cm, respectively. Sprout height on the PB and SC was not ($P > 0.05$) different between B and UnB.

Table 25. Bur oak sprouts height F and P values for treatments on Camp Grafton North 2006 – 2009.

Effect	F Value	P Value
Browsing	17.48	0.03
Treatment	15.59	0.0001
Year	2.62	0.12
Browsing x Year	0.10	0.76
Treatment x Year	11.18	0.0003
Browsing x Treatment	4.13	0.02
Browsing x Treatment x Year	0.14	0.87

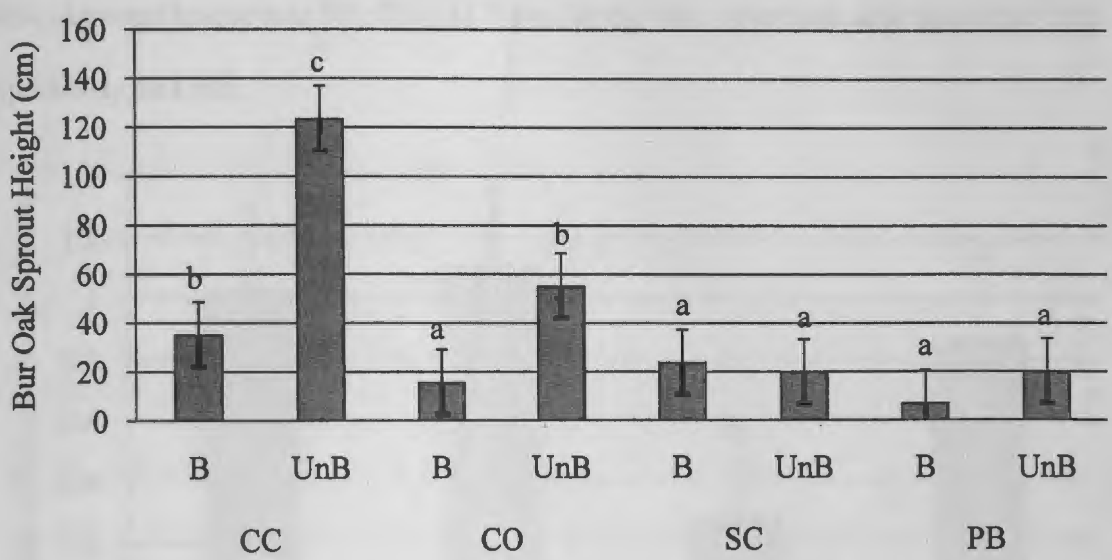

Figure 34. Mean bur oak sprout height on browsed (B) and un-browsed (UnB) plots on Camp Grafton North near Devils Lake, North Dakota, 2007 and 2009. CC represents clear-cut, CO control, SC selective cut, and PB prescribed burn. B and UnB with same letter are not different (P>0.05).

Clumps

Bur oak sprout clumps were measured only in the CC treatment. The number of bur oak sprout clumps was not different ($P > 0.05$) 12 MAFT between the B and UnB (Figure 35). However, clump density was different ($P \leq 0.05$) between B and UnB 36 MAFT. Clump density was greater ($P \leq 0.05$) in B 2007 than B 2009; however, not different ($P > 0.05$) between the UnB 2007 than UnB 2009.

Mature Trees

There was no difference (P>0.05) in tree density, DBH, and height among treatments pre-treatment in 2006. Densities of bur oak, green ash, and American elm were 356.1, 114.3, and 12.6 trees per hectare; respectively, in 2006 (Table 26). Average DBH was 32.3, 27.3, and 27.0 cm for bur oak, green ash, and American elm, respectively in

2006. Average height was 9.0, 7.5, and 7.8 m for bur oak, green ash, and American elm; respectively, in 2006.

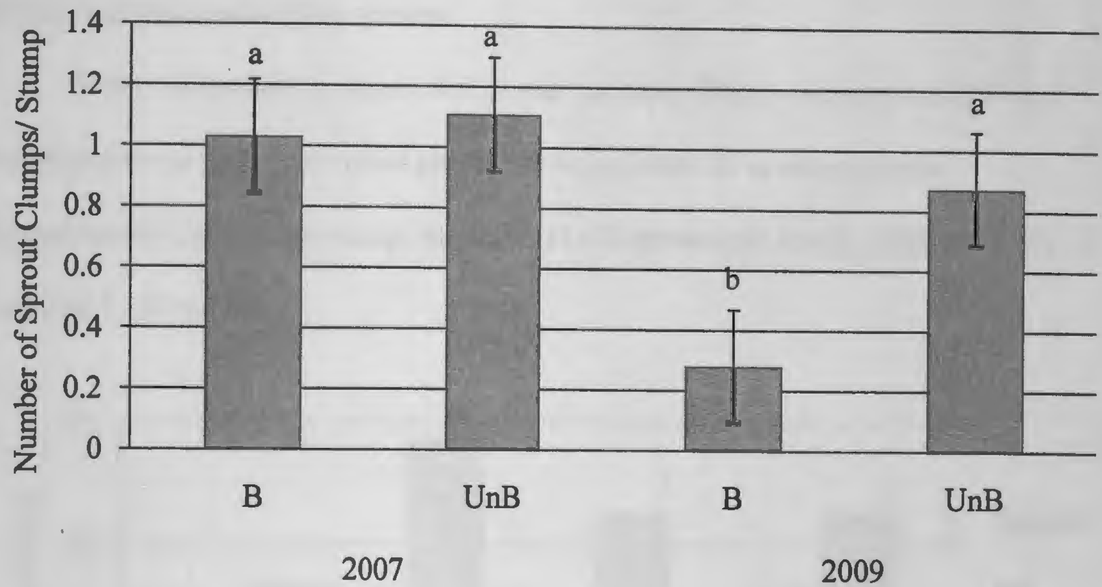

Figure 35. Mean bur oak sprout clumps on browsed (B) and un-browsed (UnB) plots only in clear-cut treatment on Camp Grafton North near Devils Lake, North Dakota, 2007 and 2009. B and UnB with same letter are not different (P>0.05).

Table 26. Tree density per treatment and mean for all treatments by species on Camp Grafton North near Devils Lake, North Dakota in 2006.

TREATMENT	Bur Oak[1]	Green Ash[1]	American Elm[1]
		Density (trees/ha)	
Control	438.6	123.6	21.7
Clear-cut	345.9	114.4	9.4
Prescribed Burn	287.4	154.4	6.2
Selective Cut	352.1	64.9	12.4
Mean (SE)	356.1 (13.1)	114.3 (10.1)	12.6 (4.3)

[1]No treatment effect (P>0.1) occurred for any species of mature trees.

Bur Oak Stump Sprouting

Season of Harvest

Season of harvest (summer vs. winter) had no effect on the sprouting ability of bur oak trees (P > 0.05). For the summer harvest, most of the stumps (43%) produced 21 or more sprouts (Figure 36). Twenty four percent (24%) of stumps produced 11–20 sprouts while 25% of the stumps produced 1–10 sprouts. Eight percent (8%) of stumps in the summer harvest produced no sprouts.

In the winter harvest sprout ability was variable. Thirty–four percent (34%) of stumps produced no sprouts while another 34% produced 21 or more sprouts. Approximately 24% of the stumps produced 11–20 sprouts per stump, while only 8% produced 1–10 sprouts.

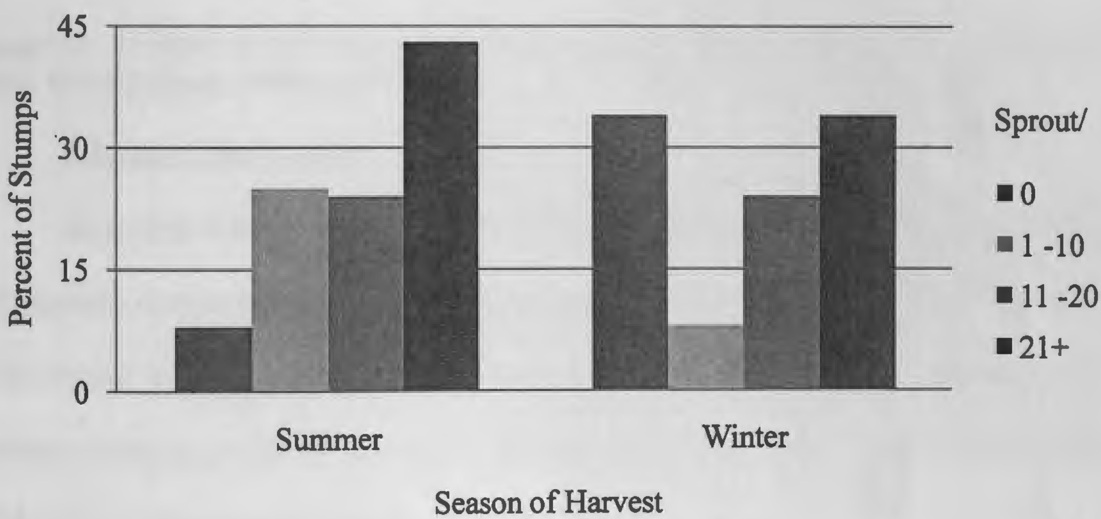

Figure 36. Number of sprouts per stump by season of cut of bur oak trees on Camp Grafton North near Devils Lake, North Dakota, 2008 and 2009.

Age

The sprouting ability was not different among the different age classes ($P > 0.05$). It appears, however, that production of the highest number of sprouts (21+) increases with age (Figure 37). It must be noted that sample size for the 226+ years was only five. The percent of stumps producing zero sprouts appears to increase with age to 126-175 years, declining after thereafter.

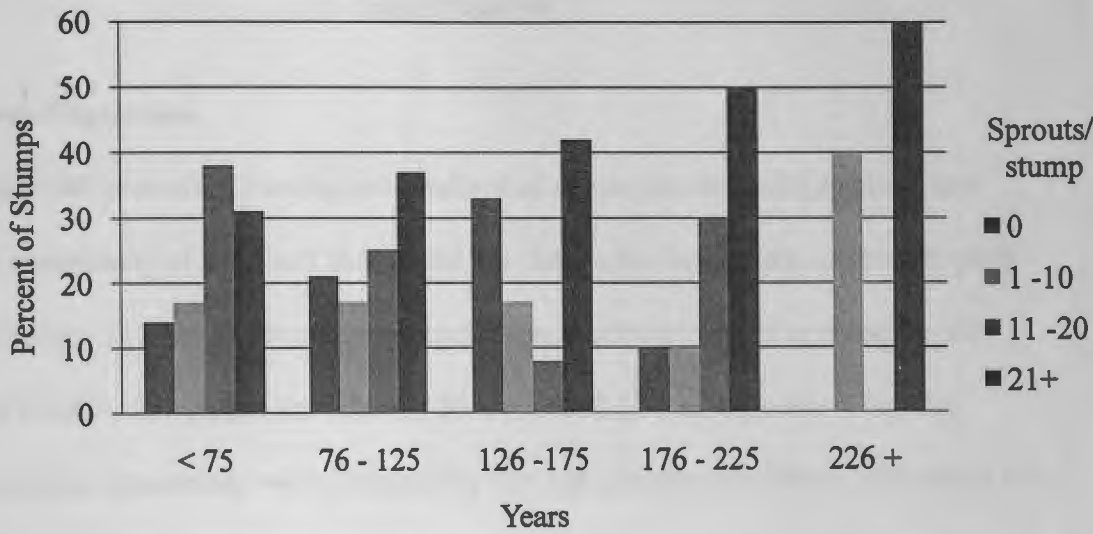

Figure 37. Number of sprouts per stump by age of tree on Camp Grafton North near Devils Lake, North Dakota, 2008 and 2009.

Diameter Outside Bark

Sprouting ability was not different (P > 0.05) among trees of different size classes. All diameter classes were dominated by stumps producing 21 or more sprouts (Figure 38). In the largest diameter class (6.2dm), all three stumps produced 21 or more sprouts. The percent of stumps producing zero sprouts ranged from 16% (3.1 – 4.1dm diameter range) to 27% (4.1 – 5.1dm diameter range).

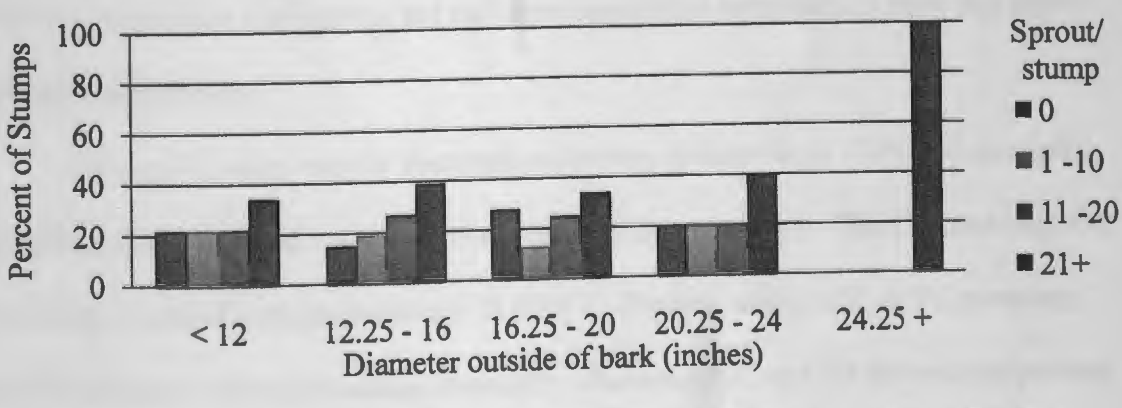

Figure 38. Stump percentage in relation to diameter outside of bark on Camp Grafton North near Devils Lake, North Dakota, 2008 and 2009.

DISCUSSION

Herbaceous Vegetation

Early-fall prescribed burning and mechanical cutting in a bur oak/green ash tree woodland community of the North Dakota did not change the herbaceous understory plant species diversity 12 MAFT. However, the herbaceous understory plant community did change 36 MAFT. In a study conducted by Biondini et al (1998), they also found the herbaceous plant community was influenced by fire and moisture conditions. Exclusion of white-tailed deer from browsing allowed an increased frequency of little-seed ricegrass after three years, irrelevant of treatment; however when deer browsing continued, frequency after three years remained at pre-treatment levels.

Clear-cutting treatments had an influence on the herbaceous understory plant community and tree species. Opening the canopy using the CC treatment decreased understory forbs, graminoids and tree species seedlings, except green ash which increased, and increased Canada thistle frequency and density. In contrast, Larson and Johnson (1998) reported clear-cutting was a successful, reliable method to regenerate oak forests; however, single-tree breaks were not sufficient enough for reproduction rates that allow overstory recruitment.

Sprengel's sedge was the dominant understory graminoid on CGN and showed a significant response to all treatments that disrupted the understory. The CO treatment did not change Spengel's sedge frequency 36 MAFT. Cutting, using a CC or SC treatment, and PB decreased Sprengel's sedge frequency. However, CC and PB showed the greatest decline in Sprengel's sedge by 70% from 36 MAFT, indicating opening of the canopy of a bur oak/green ash woodland community and fall fire will decreased Sprengel's sedge in two to three years. The time of the burn could have affected its ability to respond to the disruption in its life cycle and the increased sunlight under clear-cutting created a competition advantage to invasive plants.

Wild sarsaparilla was the dominant herbaceous understory forb on CGN and had a significant response to time and treatments. All treatments, including the CO, had an initial increase in plant density 12 MAFT, but plant numbers declined to near pretreatment levels 36 MAFT in the CO, PB and SC, but not CC. The CC treatment exposed wild sarsaparilla to sunlight, reducing its density 36 MAFT.

Grazing effects of white-tailed deer in the CC treatments could be an underling factor for the decline in wild sarsaparilla, as the browsing treatment was trending ($P = 0.08$) to reducing the density of this plant. Deer vastly alter the plants they utilize and disturb. Deer are a keystone species that can reorganize whole ecological communities (McShea and Rappole 1992, Waller and Alverson 1997, Paine 2000, Rooney 2001). Wild sarsaparilla is forage utilized by white-tailed deer and most desirable in June, with palatability decreasing as the growing season progresses. June is highest in plant digestibility and protein content with a gradual decline over the growing season (Lesage et al. 2000).

Canada thistle was not found in any treatment in the pretreatment year and increased 83% in the CC from 12 MAFT to 36 MAFT. Canada thistle remained at low levels in all other treatments including the CO. The limited sunlight and native understory plants ability to compete for resources can explain why Canada thistle density was low in the SC, PB and the CO. Opening up the canopy and the inability of native understory plants to compete due to increased sunlight allow Canada thistle to increase and flourish. Once

Canada thistle is established it creates densely packed colonies growing up to 30m in diameter and root fragments as small as 1.3cm can develop into new plants (USDA-Forest Service 2010).

As Canada thistle colonies grow, they displace desirable native vegetation, degrade wildlife habitat and compete with crops and pastures (USDA-Forest Service 2010). In this study, Canada thistle appears to have displaced wild sarsaparilla in a clear-cutting treatment. The North Dakota Department of Agriculture - Noxious Weeds Division (2009) also stated that in extreme cases Canada thistle may take over entire pastures and displacing desirable plant communities.

Bur Oak

Bur oak seedling density was low during the pre-treatment year, with few seedlings found on any treatment, including the CO. However, all treatments, including the CO, had increases in bur oak seedlings 12 MAFT. However, 24 MAFT bur oak seedlings decreased, and then increasing 36 MAFT to near the level of seedlings found 12 MAFT. The overall year effect is more likely due to a good acorn crop. Time, environmental conditions and the survival of viable seedlings were the driving factors in the increase and decline of the seedlings; however, CC was showing a positive trend in bur oak seedling establishment. Every two to three years a viable seed crop occurs with low or no crop in intervening years (Johnson 1990, USDA-NRCS 2009a). Part of the temporal variability in seedlings is due to the variable seed crop and their ability to survive.

Seedlings in their first year are poor competitors with limited height growth (Larsen and Johnson 1998). Seedlings present in the understory have trouble competing unless there are small openings in the canopy or major disturbances occur (Beck 1970, Ehrenfeld

1980, McGee 1984, Beck and Hooper 1986). Bur oak seedling survival is determined by site characteristics, such as microclimate and agents that can destroy the shoot (Swiecki and Bernhardt 1992). Acorns have a higher rate of germination where soil disturbance creates contact with bare soil (Sieg and Wright 1996). The reasons stated previously could explain the decrease in seedling numbers in the SC, PB and CO 12 and 36 MAFT during the study and increase on the CC 12 and 36 MAFT.

In other studies seedling concentrations have been noted to be greatest at intermediate deer densities, and lowest in very low and very high deer densities in the conifer-hardwood forests of the Great Lakes (Rooney and Waller 2003). This trend was noticeable on CGN as the deer population was estimated at 45 deer/km^2 in 2007 and bur oak seedlings were at 45,000 per hectare in CC 2007. The deer population decreased due to hunting and bur oak seedling numbers increased over treatment years.

Bur oak sapling density decreased 85% at 36 MAFT. Saplings numbers were low throughout the entire study, indicating the maturity (mature tree [diameter at breast height (DBH) $\geq$ 10 cm]) of the stand was detrimental to sapling survival, high density of white-tailed deer or any combination. Saplings growing to the mature stage are another reason for a decrease in numbers. The sapling stage is the transitional stage between seedling and overstory trees. The speed at which saplings grow is heavily dependent upon soil moisture and browsing (Swiecki and Bernhardt 1992). Bur oak saplings rarely live under dense canopy (Swiecki and Bernhardt 1992). In the CC treatment, a few bur oaks have reached sapling stage.

Sprout density decreased from 2007 to 2009 in the CC treatment where density was the greatest. Sprout numbers on the CO remained similar between 2007 and 2009. Sprouts on the PB declined from 2007 and 2009. The sprout density was similar in the SC among the two years observed. Sprout density is linked to sprout height which was reduced by deer browsing.

Browsing effect on bur oak sprouts by deer were high in the CC where there was continuous browsing. Old scars from past browsing were visible along with new shoots that were freshly browsed. Seedlings within cutting and burned treatments appear more prone to white-tailed deer browsing than non-cut and non-burned areas in our study. Tilghman (1989), Johnson et al. (1995) and Blair et al. (1983) also documented that patches of CC are the main focus of deer browsing due to increased forage densities and seedling palatability.

Continuous browsing harms the plant more than any one episode, with decreasing height growth and stem structure creating an imbalance in the shoot-root ratio (Eiberle 1980, Gill 1992, Gill and Beardall 2001). Oak can survive continuous elimination of aboveground parts by re-sprouting and/or staying in a suppressed state for long periods (Anderson 1991, Braithwaite and Mayhead 1996, Gerber and Schmidt 1996, Harmer 2001). Deer strongly impact woody species abundance, both absolute and relative (Leopold et al. 1947, Webb et al. 1956). The decrease in height by deer is controlled greatly by intensity, consistency of browsing, fluctuation between tree species, and a trees' ability to rebound from damage (Eiblerle 1978, Roth 1996, Gill et al. 2000).

Cutting and prescribed burn treatments did not impact seedling/sprout production of western snowberry or chokecherry 36 MAFT. These findings indicate impacts on seedling development differ among shrub species by treatment and environmental condition, at least within a three year period. White-tailed deer browsing did not impact seedling/sprout density 36 MAFT for western snowberry and chokecherry, indicating a lack of preference for these shrub species when bur oak trees dominate the available browse.

In this study, as a tree aged the amount of sprouts it produced upon cutting increased. This finding contradicts findings by Johnson (1977) that sprouting ability decreases with larger stem diameter. "The negative association of oak sprouting with increasing size may be related to the thicker bark of older trees and/or decline in sprouting ability and low carbohydrate reserves associated with senescence in trees (Johnson 1977)."

Re-sprouting is one of the key elements in regeneration under natural or human manipulations (Retana et al. 1991). All oaks can resprout after removal of the stem (Johnson 1975). The summer harvest had more stumps with twenty-one plus sprouts per stump compared to the winter harvest which had a similar number of zero sprouts per stump to twenty-one plus sprouts per stump. This would indicate that season of cut is important for increased sprouting ability for a more successful regeneration of bur oaks from stumps. The 6.1 +dm diameter outside bark had a higher percentage of twenty-one plus sprouts per stump compared to all other diameter classes. This would indicate that increased tree diameter is important in determining sprouting ability.

SUMMARY AND CONCLUSIONS

Selective cutting for bur oak trees was detrimental to seedling establishment and provided no benefit for sprout development on CGN after three years. A single early-fall prescribed fire did not demonstrate any benefits to bur oak seedling establishment. Sapling height and density, along with native understory forbs and graminoids, decreased in the CC browsed treatment.

Bur oak sprout establishment occurred only on the browsed and un-browsed CC treatments; however, sprout development in the form of growth height only occurred on the un-browsed CC treatment. White-tailed deer browsing impacted height growth on bur oak sprouts. The CC treatment appears to be trending towards an increase of bur oak seedlings along with an increase of green ash seedlings. This bur oak sprouting ability from tree stumps also increases with increased age of tree, diameter and a growing season harvest in eastern North Dakota forests.

Herbaceous plant species diversity followed similar trends pre-treatment on browsed and non-browsed plots. The PB treatment appears to enhance species richness 12 MAFT, but was not different from the CO at 24 and 36 MAFT. The CC treatment was the only treatment to increase plant species diversity within the herbaceous understory of the bur oak/green ash community and only after 36 MAFT. Species evenness was the same between 36 MAFT and pretreatment on the browsed and un-browsed plots.

A major increase of invasive plants, mainly Canada thistle, was observed on the CC browsed and un-browsed plots three years after treatment. The CC treatment also decreased wild sarsaparilla density after three years. It appears, clear-cutting a bur oak/green ash woodland community will create an environment that causes Canada thistle to displace wild sarsaparilla within three years.

MANAGEMENT RECOMMENDATIONS

1. Continue monitoring the white-tailed deer population to determine the continued effects of browsing on seedling establishment and on height and density of bur oak sprouts and saplings.
2. Narrowing the distance between removal of mature bur oak trees in a clear-cutting treatment to see if there would be a different effect on seedling establishment and how it would affect Canada thistle invasion and wild sarsaparilla density.
3. Monitor Canada thistle and other invasive plant species to determine the density and length of infestation. Control Canada thistle through natural methods like weevils since chemicals or herbicides can be detrimental to the forest community
4. Increasing burns could help control invasive plants and will also result in a better understanding of the impacts on oak establishment.
5. Monitor wild sarsaparilla, Sprengel's sedge and little-seed ricegrass to determine if there is a continued decline in density and determine how they interact with invasive species or specific treatments. Continue monitoring of Kentucky bluegrass to see if specific treatments are effective in controlling its invasion rate.

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APPENDIX A

Various plant species found in treatment areas on Camp Grafton North in 2006 - 2009.

Plant list

Scientific	Common
Agropyron repens	Quack grass
Alisma subcordatum	Water plantain
Amaranthus retroflexus	Rough pigweed
Amelanchier alnifolia	Saskatoon service-berry
Amphicarpea bracteata	Hog peanut
Aralia nudicaulis	Wild Sarsaparilla
Aster simplex	Panicled aster
Bidens frondosa	Beggar-ticks
Brassica nigra	Black mustard
Bromus inermis	Smooth Brome
Bromus latiglumis	Early leaf brome
Carex brevior	Fescue sedge
Carex lanuginosa	wooly sedge
Carex sprengelli	Long beaked sedge
Celtis occidentalis	Hackberry
Chenopodium album	Lamb's quarters
Chenopodium rubrum	Alkali blite
Cirsium arvense	Canada thistle
Convolvulus arvensis	Field bindweed
Corydalis aurea	Golden corydalis
Elymus canadensis	Canada wild rye
Elymus virginicus	Virginia wild rye
Euphorbia esula	Leafy spurge
Fraxinus pennsylvania	Green ash
Galium aparine	Catchweed bedstraw
Geum aleppicum	Yellow avens
Hackelia deflexa	Stick seed
Heracleum sphondylium	Cow parsnip
Hystrix patula	Bottle brush grass
Lactuca oblongifolia	Blue lettuce
Lappula redowskii	Stickseed
Medicago lupulina	Black Medick
Melilotus officinalis	Yellow sweet clover
Muhlenbergia frondosa	Wirestem muhley

Nepeta cataria	Catnip
Oryzopsis micrantha	Little-seed ricegrass
Oxalis stricta	Wood sorrel-yellow
Panicum capillare	Common witchgrass
Parthinocissus uinquefolia	Virginia creeper
Phalaris arundinacea	Reed canary grass
Plantago major	Common plantain
Poa pratensis	Kentucky Blue Grass
Populus deltoides	Cottonwood
Prunus virginiana	Chokecherry
Quercus macrocarpa	Burr Oak
Ribes cynosbati	Dogberry
Ribes setosum	Bristly gooseberry
Rosa woodsii	Western wild rose
Rubus idaeus	Red raspberry
Rumex crispus	Curly dock
Rumex mexicanus	Golden dock
Setaria glauca	Yellow foxtail
Smilacina racemosa	False spikenard
Smilax herbacea	Carrion flower
Solidago gigantea	Late goldenrod
Sonchus arvensis	Field sow thistle
Spartina pectinata	Prairie cordgrass
Spirea alba	meadow sweet
Stachys palustris	Hedge nettle
Sympharicarpos occidentalis	Western snowberry
Taraxacum officinale	Dandelion
Thalictrum dasycarpum	Purple meadow rhue
Toxicodendron radicans	Poison ivy
Ulmus americana	American Elm
Urtica diocia	Stinging nettle
Vicia americana	American vetch
Viola adunca	Hook-spurred violet
Viola canadensis	Canadian wood violet
Viola canadensis L. var. *rugulosa*	Creepingroot violet
Xanthium strumarium	Cocklebur