Effects of Douglas fir establishment in southwestern Montana mountain big sagebrush communities

by Adam Jay Grove

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Range Science
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Abstract:
Douglas fir (Pseudotsuga menziesii (Mirb.) Franco) establishment in big sagebrush (Artemisia tridentata Nutt.) communities is often controlled by the use of prescribed fire, which also kills big sagebrush, an important component of wildlife habitat. Knowledge of the effects of Douglas fir establishment in big sagebrush communities is essential to determining when control of establishing Douglas fir by fire or other means may be warranted. Three southwestern Montana mountain big sagebrush (A.t. ssp. vaseyana [Rydb.] Beetle) communities with establishing Douglas fir were selected for study. Twenty-five sites, representing different levels of Douglas fir establishment were selected within each study area. The effects of Douglas fir growth on mountain big sagebrush and herbs were described and quantified. Data were collected on Douglas fir and mountain big sagebrush canopy cover, density, and age, Douglas fir basal area and height, mountain big sagebrush winter forage production, and herbaceous canopy cover. Models were developed for big sagebrush canopy cover and density of "large" sagebrush (average crown cover ≥ 15 cm), grass, and total herbaceous canopy cover. Negative correlation values indicated that as Douglas fir canopy cover, total basal area, and density increased, mountain big sagebrush and herbaceous variables decreased. Overall, Douglas fir canopy cover had the strongest negative relationship with sagebrush canopy cover ($r = -0.86$), density of "large" sagebrush ($r = -0.93$), sagebrush production per plant ($r = -0.87$), and total sagebrush production per transect ($r = -0.84$). While increases in Douglas fir growth caused declines in mountain big sagebrush and herbaceous variables at all study areas, models were often quite variable between study areas. Both linear and curvilinear declines in mountain big sagebrush and herbaceous variables were recorded. Model validation results indicated that individual study area models were generally not precise or accurate. Overall models for density of "large" sagebrush and sagebrush cover did predict well. Douglas fir canopy cover had the greatest impact on mountain big sagebrush and herbaceous variables. Mountain big sagebrush canopy cover declined to < 15%, as Douglas fir canopy cover increased beyond 20%. Sagebrush canopy cover declined to < 5%, as Douglas fir canopy cover increased beyond 35%.

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This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

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ABSTRACT

Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) establishment in big sagebrush (*Artemisia tridentata* Nutt.) communities is often controlled by the use of prescribed fire, which also kills big sagebrush, an important component of wildlife habitat. Knowledge of the effects of Douglas fir establishment in big sagebrush communities is essential to determining when control of establishing Douglas fir by fire or other means may be warranted. Three southwestern Montana mountain big sagebrush (*A.t.* ssp. *vaseyana* [Rydb.] Beetle) communities with establishing Douglas fir were selected for study. Twenty-five sites, representing different levels of Douglas fir establishment were selected within each study area. The effects of Douglas fir growth on mountain big sagebrush and herbs were described and quantified. Data were collected on Douglas fir and mountain big sagebrush canopy cover, density, and age, Douglas fir basal area and height, mountain big sagebrush winter forage production, and herbaceous canopy cover. Models were developed for big sagebrush canopy cover and density of "large" sagebrush (average crown cover $\geq$ 15 cm), grass, and total herbaceous canopy cover. Negative correlation values indicated that as Douglas fir canopy cover, total basal area, and density increased, mountain big sagebrush and herbaceous variables decreased. Overall, Douglas fir canopy cover had the strongest negative relationship with sagebrush canopy cover ($r = -0.86$), density of "large" sagebrush ($r = -0.93$), sagebrush production per plant ($r = -0.87$), and total sagebrush production per transect ($r = -0.84$). While increases in Douglas fir growth caused declines in mountain big sagebrush and herbaceous variables at all study areas, models were often quite variable between study areas. Both linear and curvilinear declines in mountain big sagebrush and herbaceous variables were recorded. Model validation results indicated that individual study area models were generally not precise or accurate. Overall models for density of "large" sagebrush and sagebrush cover did predict well. Douglas fir canopy cover had the greatest impact on mountain big sagebrush and herbaceous variables. Mountain big sagebrush canopy cover declined to < 15%, as Douglas fir canopy cover increased beyond 20%. Sagebrush canopy cover declined to < 5%, as Douglas fir canopy cover increased beyond 35%.

INTRODUCTION

Conifers have increased dramatically in southwestern Montana over the last 100 years (Arno and Gruell 1986). Wright and Bailey (1982) stated that Douglas fir is becoming established in grassslands throughout its range. Gruell et al. (1986) stated that Douglas fir and other conifers have become established in several million acres of Montana seral grasslands.

The establishment (encroachment, invasion, colonization) of Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) in big sagebrush (*Artemisia tridentata* Nutt.) communities poses questions for wildlife habitat management. Establishing Douglas fir may provide additional security cover to mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus nelsoni*) using big sagebrush communities for foraging. The benefits provided by additional cover will be tempered, however, if browse is lost due to Douglas fir encroachment. Land managers must balance the needs for available forage areas, cover for wildlife security, and the reduction of available forage as a result of the negative impacts of increases in tree cover. The dilemma is in ascertaining at what point the benefits provided by cover in wildlife foraging areas are being outweighed by the loss of forage due to establishing conifers.

Natural resource agencies state that establishing conifers are often controlled to maintain or improve forage for domestic livestock and wildlife, and to maintain diverse, healthy and dynamically stable ecosystems (USDA 1986, USDA 1987, BLM 1997). Prescribed burning is often used to control establishing conifers (USDA 1986, USDA 1987). However, the use of prescribed burning in big sagebrush communities where Douglas fir has become established will eliminate big sagebrush and other non-sprouting shrubs (Blaisdell 1953, Harniss and Murray 1973, Wambolt and Payne 1986), while possibly only having a marginal effect on Douglas fir. Because big sagebrush is important wildlife browse (Kufeld1973, Welch and Wagstaff 1990, Wambolt 1996), determining at what point the control of Douglas fir outweighs the loss of big sagebrush is a critical decision for land managers.

My objectives were first to examine and quantify the effects of Douglas fir establishment in mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana* [Rydb.] Beetle) communities, as to my knowledge no other study has quantified the effects of encroaching Douglas fir on mountain big sagebrush. Secondly, to determine the Douglas fir growth variable having the greatest impact on mountain big sagebrush communities, and third, to build regression models that predict the effects of Douglas fir establishment on mountain big sagebrush and herbaceous related variables. My goal was to provide land managers with information as to when control of Douglas fir in mountain big sagebrush habitats might be warranted.

LITERATURE REVIEW

Causes of Tree Establishment

Conifer establishment in mountain big sagebrush communities and rangeland succession can be viewed from 2 schools of thought: 1) the Clementsian theory of ecology (Clements 1916); and 2) the state-and-transition model (Westoby et al. 1989). Under the Clementsian theory of succession, Douglas fir establishment in mountain big sagebrush communities may be thought of as a stage in succession. Clements (1916) referred to succession as a series of invasions by different plant communities, while Sampson (1917) defined succession as "the establishment of 1 plant species by a series of invasions in a given habitat causing the replacement of 1 set of plants by another". Smith (1986) referred to succession as the "replacement of 1 community by another over time". Those who subscribe to the state-and-transition model cite increasing tree dominance as an example of a transition from one "state" (vegetative community) to an alternative "state" (Noy-Meir and Walker 1986, Westoby et al. 1989, Laycock 1991).

Regardless of the rangeland succession model one adheres to, tree establishment or encroachment generally has been perceived negatively. However, Strang and Parminter (1980) regarded encroachment of trees into grasslands as a natural process fostered by human activities, and that its perception as being "good" or "bad" depended on management objectives. Because encroachment has often been thought of as "bad" or unwanted succession, studies related to Douglas fir or other tree encroachment have generally focused on the cause or causes of encroachment. Research has identified and focused on 4 primary factors working separately or in unison to cause encroachment.

These factors are overgrazing, changes in the microenvironment, fire suppression, and climatic change. Under the state-and-transition model these factors would cause a transition between alternative steady states (Westoby et al. 1989).

Overgrazing has been cited as causing conifer encroachment by a host of researchers (Foster 1917, Rummel 1951, Springfield 1976, Vale 1975, Zimmerman and Neuenschwander 1984, Waugh and Fisser 1986). Overgrazing acts to change the microenvironment by reducing competition from herbaceous species and increasing the amount of bare mineral soil available for conifer establishment (Rummel 1951, Cooper 1960, Vale 1975, Strang and Parminter 1980, Zimmerman and Neuenschwander 1984). Exposed mineral soil is considered a prime seedbed for tree seedling establishment (Miller 1921, Shearer 1974). Shearer (1974) stated that conifer regeneration is dependent on 1) seedbed condition; 2) seed supply; and 3) environmental characteristics affecting seedling survival. Conifer seedling survival is affected by soil moisture and soil temperature (Isaac 1938, Patten 1963). Tree invasions generally occur in pulses when years of good seed production are followed by years of favorable moisture (Rummel 1951, Arnold et al. 1964, Sindelar 1971, Arno and Gruell 1986). Hoffman (1924) stated that in the Pacific Northwest, Douglas fir generally had good seed production every 2-3 years.

Douglas fir encroachment is often preceded by an increase in big sagebrush (Gruell 1983, Arno and Gruell 1983, Gruell et al. 1986). Increases in big sagebrush may be due to overgrazing, fire suppression or a combination of the two. Big sagebrush provides shaded microsites conducive to the establishment of Douglas fir seedlings (Sindelar 1971, Gruell 1983, Gruell et al. 1986). Sindelar (1971) reported that Douglas fir saplings were strongly associated with big sagebrush plants, as the majority of Douglas fir seedlings emerged directly through big sagebrush crowns. Sagebrush has been documented as acting as a "nurse" plant for other tree species as well. Burkhardt (1969) reported that western juniper (*Juniperus occidentalis* Hook.) seedlings were often found under mountain big sagebrush plants. Waugh and Fisser (1986) reported that Utah juniper (*Juniperus osteosperma* (Torr.) Little) seedlings became established predominantly under black sagebrush (*Artemisia nova* Nels.).

Direct fire suppression or the lack of fire plays a major role in tree establishment. Burkhardt (1969) attributed the establishment of western juniper in mountain big sagebrush communities directly to the cessation of periodic fires. Likewise, encroachment by Douglas fir has been attributed to fire suppression (Gruell 1980, Gruell 1983, Arno and Gruell 1986). Even when fire suppression is not considered the primary cause of encroachment, it allows encroachment to continue unabated (Rummel 1951, Cooper 1960, Patten 1963, Sindelar 1971).

Overgrazing may contribute to the lack of fire by reducing the amount or density of herbaceous vegetation, thus reducing the ability of an area to sustain or carry a fire (Foster 1917, Springfield 1976, Zimmerman and Neuenschwander 1984, Arno and Gruell 1986). While overgrazing may reduce the ability of an area to carry a fire, encroachment and the subsequent increases in tree density and canopy cover caused by overgrazing may increase the potential for catastrophic crown fires as ladder fuels increase (Zimmerman 1979, Davis et al. 1980, Gruell et al. 1982).

In addition to overgrazing, changes in the microenvironment, and fire suppression, climatic change has also been cited as a factor contributing to encroachment

(Blackburn and Tueller 1970). Johnsen (1962) stated that long droughts killed grasses and reduced competition with one-seeded juniper (*Juniperus monosperma* (Engelm.) Sarg.) seedlings. Vale (1975) stated that droughts could accentuate plant cover decreases, thereby encouraging tree establishment when more moist conditions return. Patten (1963) stated that forest advance and tree establishment in open areas was in part dependent on optimum climatic conditions.

Fire History in Douglas Fir Forests of the Rocky Mountain Region

The lack of fire in the last 100 years has resulted in extensive areas of Douglas fir and other tree encroachment in the Rocky Mountain region (Gruell 1983, Arno and Gruell 1983, Arno and Gruell 1986). Prior to fire suppression, Douglas fir habitat types were open and patchy as periodic ground fires restricted development of Douglas fir forests (Loope and Gruell 1973, Davis et al. 1980, Gruell 1983). Douglas fir was often restricted to moist sites, rocky outcrops and talus slopes (Arno and Gruell 1983, Gruell 1983, Arno and Gruell 1986). Arno and Gruell (1983) stated that with fire suppression Douglas fir has invaded grass and sagebrush habitats downslope from more rocky areas.

While fire frequency in the Rocky Mountain region is variable to begin with, fire suppression has generally resulted in longer fire-free intervals. Arno and Gruell (1983) stated that prior to 1910, mean fire intervals in the Douglas fir/grassland ecotone of southwestern Montana ranged from 35-40 years, but that no fires had been recorded in their study areas the previous 61 years. Arno and Gruell (1986) recorded a mean fire interval of 26 years in the Galena Gulch area outside of Boulder, Montana for the period 1690-1979, but only 1 fire had occurred in the 94 years prior to their study. The longest fire intervals prior to 1890 were 72-82 years. In the southern Elkhorn Mountains east of Boulder, Lehman (1995) reported post fire suppression mean fire intervals of 13.8 to 25.8 years in 4 study areas. The mean post suppression fire interval for Douglas Fir/Idaho fescue (*Festuca idahoensis* Elmer) habitat types in this area was 13.8 years compared to a historical MFI of 8.2 years (Lehman 1995). Lehman (1995) reported that over much of the study area the time elapsed since the last fire was up to 3x longer than the historic average interval between fires.

Arno (1976) found fire frequencies ranging from 6-19 years in Douglas fir/grassland habitat types on the Bitterroot National Forest of Montana. However, Arno (1976) stated that there has been a marked decrease in fire since 1920, and that in many areas the time elapsed since the last fire now exceeds the longest fire free interval from 1735-1900. Goldblum and Veblen (1992) reported a mean fire interval of 15.2 years for the period 1920-1949 for their entire study area on the Colorado front-range; however, when the time period was increased to cover the whole fire suppression era (1920-1989), the interval was 28 years. Goldblum and Veblen (1992) recorded a mean fire interval of 31.8 years prior to 1859; however, they felt that they probably had underestimated the number of fires that had occurred. Tande (1979) found that Douglas fir forests in Jasper National Park had an average mean fire return interval of 17.6 years for the period 1665-1975, but prior to 1913 when fire suppression began, the interval was only 5.5 years. Tande (1979) reported that fires covering more than 1.2% (500 ha) of the study areas had a mean fire return interval of 8.4 years, while fires covering more than 50% of the study area had a mean fire return interval of 65.5 years.

While fire frequencies historically may have been shorter than they are today, fire intensity was generally lower, and fire size was generally smaller. Loope and Gruell (1973) stated that in the Jackson Hole, WY area, very large fires were rare and crown fires only occurred under the most extreme conditions. Arno (1976) stated that short fire intervals, and the presence of multiple age class conifer stands in his 3 Bitterroot National Forest study areas, indicate that fire intensity usually was not great. Fires generally only covered a portion of the 3 study areas (Arno 1976). Tande (1979) found that only 58.5 % of the recorded fires in his 43,200-ha study area covered more than 1.2% of the study area. Sixty-eight percent of the fires covered less than 5% of the study area. Fires did not kill the entire stand, but rather left many different age classes on the landscape.

Susceptibility of Douglas Fir to Fire Treatment

Knowledge of the potential effects of fire on Douglas fir is needed if prescribed burning is to be considered for treating Douglas fir encroachment. Because of their very thick bark, mature Douglas fir are considered to be very fire resistant (Flint 1925). It takes Douglas fir approximately 40 years on moist sites to develop this corky, fire-resistant outer bark (Arno and Gruell 1983, Fischer and Bradley 1987). While generally fire resistant, mature Douglas fir may be injured by fire because of resin streaks in the bark, and because branches run the full length of the bole (Fischer and Clayton 1983). This branch habit may lead to crown fires, which are generally needed to kill mature Douglas fir (Starker 1934). In addition, dense understories created as a result of a prolonged fire-free period may provide ladder fuels for crown fires (Arno 1976, Fischer and Clayton 1983).

Although mature Douglas fir are resistant to most fires, Gruell et al. (1986) stated that seedling (trees < 1.5 m tall), sapling (2-13 cm Dbh), and pole (18-30 cm Dbh) invasions of Douglas fir may present opportunities for the use of prescribed fire. Ground or low intensity fires will kill Douglas fir seedlings (Gruell et al. 1986), and saplings (5-10 cm Dbh) are vulnerable to fire as well (Davis et al. 1980, Fischer and Clayton 1983, Fischer and Bradley 1987). Pole-sized Douglas fir are more fire resistant, and may survive low to possibly low-moderate intensity fires (Bradley et al. 1992). Kalabokidis and Wakimoto (1992) reported that fire killed 97-100% of the Douglas fir trees under 3 m on their study areas, but that only approximately 10% of the trees taller than 3 m were killed. Bushey (1986) reported that only trees taller than 1.37 m survived after 3 post-burn seasons on 2 study transects, while on another transect, Douglas fir seedlings and saplings had incurred at least 80% mortality after only 1 post-burn growing season. Moderate fires generally result in more open, park-like stands of mature trees as these fires are unlikely to kill large overstory trees (Davis et al. 1980, Fischer and Bradley 1987, Bradley et al. 1992).

While fire may kill existing trees, the use of fire may also result in a mineral seedbed perfect for Douglas fir seedlings (Davis et al. 1980, Fischer and Bradley 1987, Bradley et al. 1992). Therefore, knowledge of how Douglas fir may respond to fire is essential. Fire-related successional pathways for Douglas fir habitat types have been described for much of the Rocky Mountain region (Davis et al. 1980, Fischer and Clayton 1983, Fischer and Bradley 1987, Bradley et al. 1992). Stickney (1980) stated that "a fundamental understanding of forest succession requires basic information on the response of component species to disturbance and subsequent successional change."

Effects of Fire on Mountain Big Sagebrush and Herbaceous Species

Mountain Big Sagebrush

The decision to control Douglas fir establishment in mountain big sagebrush communities with prescribed fire may be influenced by the effects of fire on mountain big sagebrush and herbaceous species. Mountain big sagebrush is easily killed by fire (Blaisdell 1953, Harniss and Murray 1973, Bushey 1986). Because mountain big sagebrush is a non-sprouter and must recover from seed, the time required for recovery to pre-burn levels is extremely variable.

Bunting et al. (1987) stated that mountain big sagebrush might return to pre-burn density and cover levels within 15-20 years; however, no data was provided to support this claim. Studies have indicated that recovery time may be significantly longer. Walhof (1997) reported that burned mountain big sagebrush sites measured up to 16 years after burning had considerably less canopy cover and sagebrush density than paired unburned sites. Fraas et al. (1992) reported that canopy cover of mountain big sagebrush on burned sites 8 years after burning was significantly less than on paired unburned sites. Harniss and Murray (1973) reported that mountain big sagebrush made little increase in production the first 12 years following burning, and that 30 years after burning, density of mountain big sagebrush on burned areas was only 83% of unburned areas. Blaisdell (1953) found that 15 years after burning, mountain big sagebrush density on burned sites was only 50% of unburned sites.

Although it may take many years for mountain big sagebrush to reach pre-burn levels of density and cover, mountain big sagebrush may reestablish rapidly following burning (Blaisdell 1953, Bushey 1986, Walhof 1997). Walhof (1997) showed that in his study areas, peak reestablishment of mountain big sagebrush occurred within the first 4 growing seasons after the burn. Peak reestablishment occurred in either the year of treatment or the year following treatment in 4 out of 7 mountain big sagebrush locations (Walhof 1997).

How quickly mountain big sagebrush plants become reestablished may depend on burn intensity related to how many sagebrush plants survive the burn, as well as seed production prior to burning (Blaisdell 1953). Johnson and Payne (1968) reported that the number of big sagebrush (subspecies not given) surviving spraying and plowing treatments was the most important factor influencing sagebrush re-invasion. Winward (1970) noted that mountain big sagebrush often becomes dense following a disturbance. The rapid reestablishment of mountain big sagebrush may be attributed to the fact that it generally occupies more mesic sites than the other 2 subspecies of big sagebrush (Winward and Tisdale 1977). The fact that mountain big sagebrush may reach reproductive maturity in 3-5 years may also allow it to recover more rapidly in burned areas (Bunting et al. 1987).

Herbaceous Species

Because prescribed fire is often used or recommended to control encroaching Douglas fir (Bushey 1986, USDA 1986), knowledge of the effects of fire on herbaceous species is important for deciding on an individual case basis whether or not to use prescribed fire. Lyon (1971) reported that the number of herbaceous species doubled within 7 years after burning a Douglas fir community in south central Idaho. Bushey (1986) reported that herbaceous production doubled and even tripled following burning of mountain big sagebrush and Douglas fir. Wambolt and Payne (1986) found that basal cover of perennial grasses was over 6 times higher 3 years after burning Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis* Beetle and Young), and that production was still 3 times higher than the pre-burn level 17 years after burning. Blaisdell (1953) and Mueggler and Blaisdell (1958) also documented increases in herbaceous production following burning of big sagebrush-grasslands.

While burning may have a positive effect on herbaceous species, the positive effects often decrease over time. Blaisdell (1953) stated that increases in grass and forb production might be short-lived. Fraas et al. (1992) reported that while there were more herbaceous species in burned areas 8 years after burning, total grass canopy cover was similar to unburned sites. Harniss and Murray (1973) reported that herbaceous species increased the first 12 years after burning, but that 30 years after burning, relative yields of various grasses were near pre-burn levels. Walhof (1997) reported that perennial grass canopy cover was not significantly different overall between 13 paired burned and unburned areas of different ages.

The effects of fire on Idaho fescue and bluebunch wheatgrass (*Agropyron spicatum* (Pursh) Scribn. & Smith) are of particular importance, because these 2 species often form habitat types with big sagebrush and Douglas fir (Pfister et al. 1977, Mueggler and Stewart 1980). These 2 species are also important forage species for both wildlife and cattle. Idaho fescue can be seriously injured by fire (Blaisdell 1953, Countrymen and Cornelius 1957, Conrad and Poulton 1966, Harniss and Murray 1973, Young 1982). The amount of damage to Idaho fescue appears to be related to factors affecting burn intensity such as the amount of sagebrush cover and the amount of dead fescue present (Conrad and Poulton 1966, Gruell et al. 1986). While initially damaged, Idaho fescue may recover from injury fairly quickly, as Antos et al. (1983), in grasslands, and Bushey (1986), in Douglas fir encroached mountain big sagebrush communities, reported that Idaho fescue had recovered to at least pre-burn levels 3 years after burning. Walhof (1997), in southwestern Montana big sagebrush-grasslands, reported that overall Idaho fescue cover did not differ significantly between burned and unburned areas of different ages. Gruell et al. (1986) recommended burning Idaho fescue in the spring or in the fall, after plants are dormant to reduce injury.

Bluebunch wheatgrass appears to be less affected by fire than Idaho fescue in big sagebrush-grasslands (Conrad and Poulton 1966). Antos et al. (1983) reported no significant difference in bluebunch wheatgrass cover between burned and unburned sites 1 growing season after burning. Conrad and Poulton (1966) reported that fire affected bluebunch wheatgrass plant size and not plant density. Blaisdell (1953) reported that bluebunch wheatgrass was slightly affected initially by fire in mountain big sagebrush-grasslands, but responded well following burning. Harniss and Murray (1973) reported that after an initial decline, production of bluebunch wheatgrass was nearly double 12 years after burning of mountain big sagebrush-grasslands. However, Fraas et al. (1992) reported no difference in bluebunch wheatgrass canopy cover between paired burned and unburned sites 8 years after burning of mountain big sagebrush-bluebunch wheatgrass communities.

Importance of Cover to Big Game

Cover, a significant component of big game habitat in the northwestern United States, may influence the microclimate of an area and obstruct visibility, thereby providing security from a distance (Peek et al. 1982). Lyon and Jensen (1980) stated that in the case of clear-cuts, the willingness of animals to enter an opening is influenced by requirements for security. Security (hiding) cover appears to be a requirement for elk in the presence of human disturbance (Peek et al. 1982). Gruell et al. (1982) stated that sparse cover subjects elk to harassment and allows hunters to be more effective. Lyon (1979) found that tree cover modified use of a roaded area by elk, and that increasing tree cover resulted in elk losing less habitat effectiveness due to roads. While elk use of different tree canopy cover classes in northwestern Montana is season and sex dependent, they use areas with no tree canopy cover the least (Marcum 1975, Marcum et al. 1984).

Thomas et al. (1979) defined hiding cover as "vegetation capable of hiding 90% of a standing adult deer or elk from the view of a human at a distance equal to or less than 61m (200 feet). The distance away an elk or deer (*Odocoileus* spp.) is seen is called the site distance. Site distance is a function of the horizontal shielding effects of low growing vegetation and tree stems (Skovlin 1982). Lyon and Jensen (1980) stated that elk in western Montana would not fully utilize a clear-cut until vegetation reached a minimum of 1 m in height. Lyon and Jensen (1980) found that deer use of clear-cuts in western Montana dramatically increased after vegetation height exceeded 0.3 m, and then leveled off after vegetation reached 1 m in height. For eastern Montana deer, use of clear-cuts increased as vegetation height in the opening increased (Lyon and Jensen 1980). Maximum use occurred shortly before growth reached 1 m in height. Canfield et al. (1986) stated that steep viewing angles reduce the effectiveness of cover, but that this is buffered by tree height due to a trigonomic layering effect. Taller trees provide better security cover with a threshold appearing to exist at around 7.6-9.1 m.

The security benefits of increases in canopy cover may be tempered by reduced forage amounts. Lyon and Jensen (1980) stated that both elk and deer preferred clear-cuts with cover in the opening except when cover inhibited forage growth. Suring and Vohs (1979) reported that communities that provided both hiding cover and forage were more heavily utilized by Columbian white-tailed deer (*Odocoileus virginianus leucurus*) than communities providing only hiding cover or forage. Loft and Menke (1984) stated that Columbian black-tailed deer (*Odocoileus hemionus columbianus*) used plots that had low tree canopy cover, high amounts of hiding cover from vegetation 0-0.5 m tall, high herbaceous production, and higher amounts of shrubs. Short et al. (1977) and Reynolds (1964) reported that mule deer and elk use of pinyon-pine (*Pinus edulis* Engelm.)-juniper (*Juniperus* spp.) woodlands decreased as tree density increased and shrub and herbaceous production decreased. However, Reynolds (1964) reported that pellet group numbers were lowest where densities of trees and shrubs were the lowest, indicating that some tree presence was desired.

While increasing security cover may benefit elk and mule deer, it has a negative impact on bighorn sheep (*Ovis canadensis*). Visibility is an important habitat variable to bighorn sheep (Risenhoover and Bailey 1980, Wakelyn 1987, Etchberger et al. 1989). Bighorn sheep avoid tall vegetation habitat types that reduce visibility such as Douglas fir habitat types (Risenhoover and Bailey 1985). Wakelyn (1987) stated that encroachment of tall dense shrublands and forest in the absence of fire has led to the loss of bighorn sheep habitat. Lotan and Brown (1985) reported that burning forest canopy improved bighorn sheep habitat conditions in British Columbia.

The Importance of Douglas Fir and Mountain Big Sagebrush to Wildlife

Cover and Habitat

Douglas fir establishment in mountain big sagebrush communities may provide valuable habitat to wildlife. Reports by Schwarzkoph (1973) and Steerey (1979) indicate that Douglas fir and big sagebrush habitat types play an important role for mule deer in the Bridger Mountains of Montana. Douglas fir and mountain big sagebrush may benefit wildlife by providing both security cover and forage. Douglas fir provides good security cover because 1) branches are moderately low and dense in saplings, and run the length of the bole in mature trees (Flint 1925, Fischer and Bradley 1987); and 2) Douglas fir generally forms moderate to dense stands (Flint 1925).

Big sagebrush may provide protective cover for mule deer fawns and elk calves, nesting habitat for songbirds, and habitat for small mammals (Peterson 1995). Sagebrush also provides the dominant nesting cover for sage grouse (Wallestad and Pyrah 1974). Sagebrush habitats are also important during the sage grouse breeding and brood rearing periods (Klebenow 1969, Wallestad and Schladweiler 1974). Roberson (1984) stated that sage grouse are sagebrush obligates.

Forage

The use of big sagebrush as forage by mule deer and elk has been well documented (Wilkins 1957, Schwarzkoph 1973, Kufeld 1973, Kufeld et al. 1973, Steerey 1979, Welch and Wagstaff 1990, Wambolt 1996). Wambolt (1996) and Personius et al. (1987) reported that mountain big sagebrush was the preferred subspecies of big sagebrush. However, Striby (1985) stated that Wyoming big sagebrush was the most preferred subspecies based on form class assessment, with mountain big sagebrush being the second most preferred. In addition to being important to big game, big sagebrush is also the primary staple for sage grouse, especially during the winter (Wallestad et al. 1975).

Mule deer use of small to trace amounts of Douglas fir as forage has been reported in the Bridger and Little Belt Mountains of Montana (Wilkins 1957, Lovass 1958, Schwarzkoph 1973, Steerey 1979). While damage to Douglas fir seedlings by both black-tailed deer and elk (*Cervus elaphus roosevelti*) has been reported in western Washington and Oregon (Campbell and Evans 1975), mule deer generally select needles from larger diameter trees rather than smaller diameter ones (Tucker et al. 1976, Dawson et al. 1990). Mule deer also select needles from trees in openings over those in gullies (Tucker et al. 1976).

METHODS

Study Areas

Selection

Three mountain big sagebrush communities with encroaching Douglas fir were selected for study. Aerial photographs taken over a number of years verified that encroachment was actually occurring. Study areas were geographically separated to increase the study's range of inference. The minimum distance between study areas was 41 km, while the maximum distance was 148 km. The areas exhibited the following characteristics: 1) levels of encroachment that were characterized by a range of Douglas fir canopy cover from low to high; 2) an area large enough to contain 25 study sites 30.5 x 30.5 m in size; and 3) no recent impacts on the communities from burning, spraying or cutting.

Description

Study areas were located in southwestern Montana (Fig. 1). Legal descriptions and sampling dates for each study area are in Table 1. All study areas were classified as Douglas fir/Idaho fescue habitat types (Pfister et al. 1977). The Douglas fir/Idaho fescue habitat type is part of the cool-dry Douglas fir habitat types fire group (Fischer and Clayton 1983).

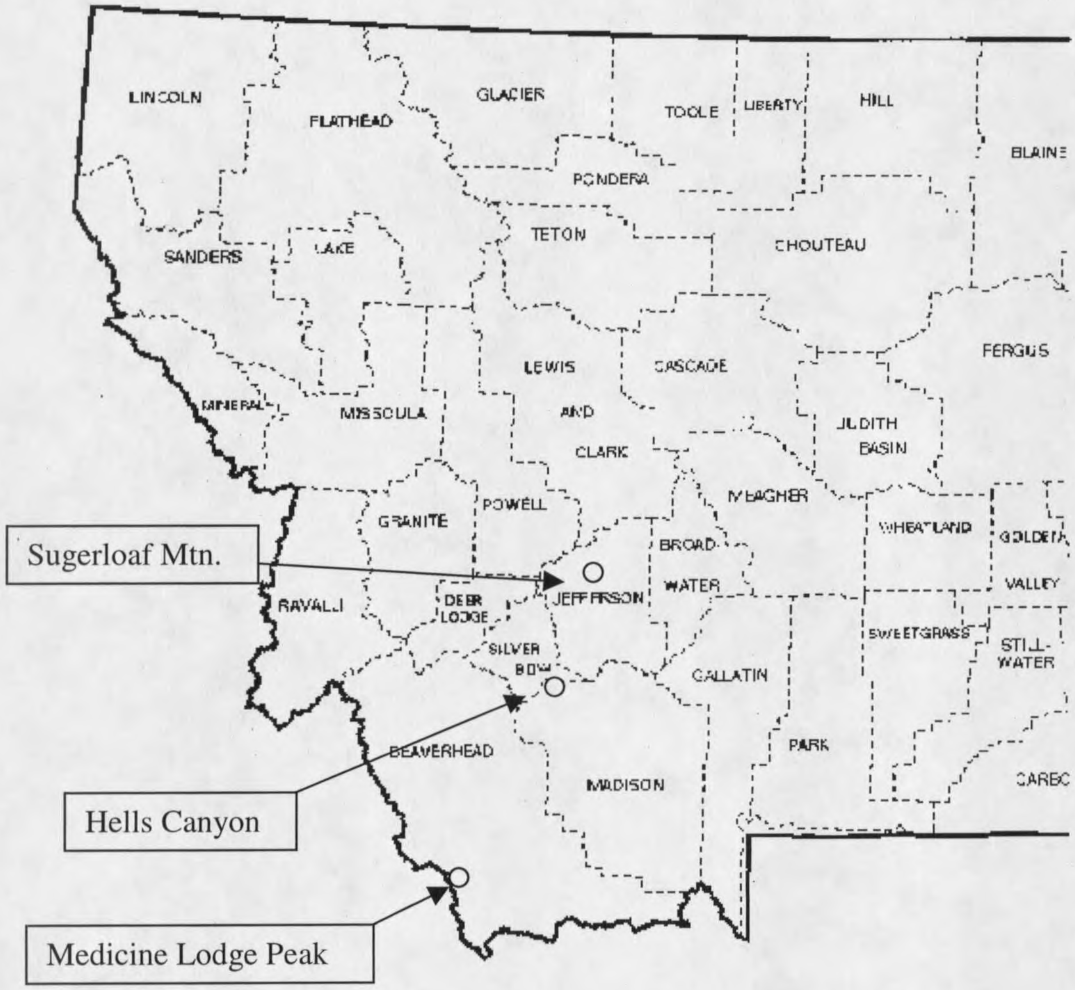

Figure 1. Map of 3 study areas in southwestern Montana.

Table 1. Legal descriptions and sampling dates for 3 study areas.

SITE	Legal Description	Date Sampled
Sugerloaf Mountain	6N 4W SW S7, NW S18 6N 5W SE S12, NE NE S13	July, August, 1996
Medicine Lodge Peak	12S 12W S SE S10, SE SE SW S10 12 S 12W NE S15, NE NE NW S15	June, July, 1997
Hells Canyon	1S 6W NW SW S32, SW NW S32 1S 6W NW S31	July, August, 1997

Sugerloaf Mountain Study Area

The Sugerloaf Mountain (SLM) study area is located in Jefferson County approximately 6 km northwest of Boulder, MT on Bureau of Land Management (BLM) land. The area is part of the Boulder Batholith formation and is several kilometers north of Arno and Gruell's (1986) Galena Gulch study area. Parent material is intrusive igneous rock (Montagne et al. 1982). The soil was classified as a sandy, mixed Typic Cryoboroll (Soil Survey Staff 1994). Slope, aspect, elevation and A horizon soil texture for each of the 25 study sites are given in Table 2. Mean annual air temperature is 3.9° C (Caprio et al. 1994). The length of the freeze-free season is 70-90 days with an annual precipitation of 357-406 mm, with most of this occurring May-July (Caprio et al. 1994). A vegetation species list for the area is provided in Appendix A. Cattle graze the area in the summer. Mule deer appears to be the primary big game species using the area, although some elk fecal pellets were seen in the study area.

Table 2. Slope, aspect, elevation, and A horizon soil texture of Sugerloaf Mountain study sites.

Site#	Degree of Slope	Aspect	Elevation (m)	Soil Texture of A Horizon
1	12	E	1820	Loamy Medium Sand
2	9	E-NE	1829	Loamy Medium Sand
3	14	E	1823	Loamy Medium Sand
4	9	E	1817	Loamy Medium Sand
5	10	SE	1811	Loamy Medium Sand
6	12	E	1811	Loamy Coarse Sand
7	11	E-SE	1804	Loamy Medium sand
8	7	S	1792	Loamy Medium Sand
9	6	S	1817	Loamy Medium Sand
10	9	S	1817	Loamy Medium Sand
11	10	S	1823	Loamy Medium Sand
12	14	E-SE	1826	Medium Sandy Loam
13	12	E	1826	Loamy Coarse Sand
14	14	E	1826	Loamy Medium Sand
15	14	E	1835	Loamy Medium Sand
16	16	E	1841	Loamy Medium Sand
17	17	NE-E	1853	Medium Sandy Loam
18	11	SE	1841	Loamy Medium Sand
19	9	SE-S	1841	Loamy Medium Sand
20	12	SE	1853	Coarse Sandy Clay Loam
21	10	SE	1853	Loamy Coarse Sand
22	9	S	1823	Loamy Coarse Sand
23	13	E-SE	1865	Coarse Sandy Loam
24	14	NE-E	1871	Coarse Sandy Loam
25	14	E	1868	Coarse Sandy Loam

Medicine Lodge Peak Study Area

The Medicine Lodge Peak (MLP) study area is located in Beaverhead County approximately 24 km south of Grant, MT on BLM land. The study area lies in the foothills of the Beaverhead Mountains. Parent material is intrusive igneous rock (Montagne et al. 1982). The soil was classified as a clayey-skeletal, mixed Argic Cryoboroll (Soil Survey Staff 1994). Slope, aspect, elevation and A horizon soil texture for each of the 25 study sites are given in Table 3. The mean annual air temperature is 1.7° C (Caprio etal., 1994). The length of the freeze-free season is 10-30 days with

Table 3. Slope, aspect, elevation, and A horizon soil texture of Medicine Lodge Peak study sites.

Site#	Degree of Slope	Aspect	Elevation (m)	Soil Texture of A Horizon
1	10	288	2231	Silty Clay Loam
2	11	344	2231	Clay Loam
3	10	324	2231	Clay Loam
4	9	344	2243	Clay Loam
5	11	304	2237	Clay Loam
6	12	324	2256	Clay Loam
7	18	314	2268	Clay Loam
8	7	357	2231	Clay Loam
9	10	352	2256	Clay Loam
10	9	346	2237	Clay Loam
11	18	328	2268	Clay Loam
12	14	318	2243	Clay Loam
13	12	332	2237	Silty Clay Loam
14	14	320	2249	Clay
15	14	342	2280	Clay Loam
16	10	0	2286	Silty Clay Loam
17	9	334	2259	Silty Clay Loam
18	10	0	2261	Clay Loam
19	9	0	2261	Clay Loam
20	9	356	2268	Silty Clay Loam
21	7	354	2262	Clay Loam
22	7	4	2274	Clay Loam
23	8	348	2274	Clay Loam
24	7	24	2280	Sandy Clay Loam
25	14	342	2280	Clay Loam

annual precipitation of 357-406 mm, with most of this occurring from April-July. A vegetation species list for the area is provided in Appendix A. Cattle graze the area in the summer. Mule deer were seen in the study area, and elk were seen in the vicinity of the study area. Mule deer, elk, and moose fecal pellets were seen in the study area.

Hells Canyon Study Area

The Hells Canyon (HC) study area is located in Madison County approximately 20 km northwest of Twin Bridges, MT on BLM and Big Sky Lumber Company land.