ASPEN RESPONSE TO PRESCRIBED FIRE IN SOUTHWEST MONTANA

by

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APPROVAL

of a thesis submitted by

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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, citation, bibliographic style, and consistency, and is ready for submission to the Division of Graduate Education.

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Daniel Avery Durham

December 2008

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

A collaborative effort by the BLM, MAES and MFWP, the Whitetail Watershed Restoration Project used prescribed fire in 2005 and 2006 to address aspen decline, conifer encroachment and altered hydrologic function in a forested watershed within Jefferson County, MT. As part of this effort quaking aspen response to fire was evaluated in two sub-drainages of the Whitetail Basin three years after treatment. Unburned stands were first surveyed to determine whether regeneration was occurring and to measure the distribution of aspen stems by size class. This information was then compared to stem response in burned stands. Big game and cattle impacts on aspen sucker height and density were measured using a series of 3-part ungulate exclosures in a sub-sample of burned stands. Regeneration was occurring in only1 of 40 unburned stands suggesting aspen was declining in this area. Sucker density increased dramatically in the burned stands after three years increasing the likelihood for regeneration. Within the first three years post-fire big game and the combination of big game and cattle did not affect sucker density in the burned stands. Although sucker height was significantly less in plots used by ungulates we did not feel it was enough to prevent regeneration. This assertion was supported by sufficient annual growth rates and the recruitment of individual regeneration stems into stands outside of protected plots. While it appears fire has increased the potential for aspen regeneration in the Whitetail Basin, early growth rates have allowed for some individual stem to surpass browse height to date, suggesting future monitoring will be necessary to learn if the current recruitment levels are sufficient to regenerate the majority of stands.

INTRODUCTION

Quaking aspen (*Populus tremuloides* Michx.) has declined throughout the western United States over the past century (Bartos 2001, Baker et al. 1997, White et al. 1998). The extent of this decline varies from 49-96% among the western states while it is estimated at 64% in Montana (Bartos 2001). Although a variety of factors contribute to aspen decline, fire suppression and severe ungulate herbivory prevent stand regeneration most often (Despain et al. 1986, Romme et al. 1995, White et al. 1998, Kay and Bartos 2000). The loss of aspen is a concern due to the important ecological and social values associated with this cover type, including high biodiversity, forage production and water yield (Mueggler 1985a, DeByle 1985b, Kay 1997, McCool 2001, LaMalfa 2008). Land managers often attempt to restore aspen by returning fire to the landscape despite inconsistent results.

Fire was important historically for stimulating aspen reproduction and preventing conifer establishment in the Northern Rockies region (Jones and DeByle 1985a). Aspen is a clonal species, reproducing primarily by root suckering following disruption of apical dominance (Schier et al. 1985). In the absence of fire or other disturbance, most aspen stands in the Northern Rockies are seral to Douglas-fir [*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco] or other conifer species (Mueggler 1985b). Fire can also create the appropriate conditions for aspen seedling establishment (Turner et al. 2003). However, due to fire suppression and the elimination of indigenous burning, fire return intervals have increased greatly throughout the west (Arno and Gruell 1983, 1986). In southwest Montana fires have not burned in many areas for close to 100 years, despite historic intervals of 25-40 years (Arno and Gruell 1983, 1986). The effect of fire suppression on aspen regeneration is likely compounded by ungulate herbivory.

Rocky Mountain elk (*Cervus elaphus nelsoni* Bailey), mule deer (*Odocoileus hemionus* Rafinesque), white-tailed deer (*Odocoileus virginianus* Zimmerman), moose (*Alces alces* Peterson), and domestic livestock feed on aspen twigs and leaves (DeByle 1985c). Severe ungulate browsing can prevent aspen stems from replacing the overstory (Kay 2001), reduce sucker densities (Bartos and Mueggler 1981), and is often associated with fungal infections of aspen stems (Hart and Hart 2001). Ungulates can also cause severe damage to aspen stands by rubbing and gnawing on fragile aspen bark (Keigley and Frisina 2008). While elk are most often blamed for aspen decline, livestock can also prevent regeneration or exacerbate wildlife impacts (Kilpatrick et al. 2003, Kay and Bartos 2000). While control of ungulate populations could potentially allow for aspen regeneration, it is often not desired by the public or land managers.

Instead, prescribed fire is often recommended for restoring aspen woodlands because it stimulates prolific suckering and provides optimal growing conditions for young aspen (Sheppard 2001). However, controlled burns have produced mixed results due to severe browsing of suckers following treatment (Bartos and Mueggler 1981, Kay 2001, Kilpatrick et al. 2003). In some situations the combination of fire and severe ungulate use has eliminated stands, prompting researchers to suggest prescribed fire could hasten the demise of aspen (White et al. 1998, Kay 2001). However, many of these reports come from national parks, elk feedgrounds or adjacent areas that are known to have high big game densities.

The purpose of this study was to determine whether fire is an effective tool for regenerating aspen in areas with relatively low elk density (less than 1 elk/km^2) (White et al. 1998), managed cattle grazing and sparse aspen cover in southwest Montana. These conditions were met in the Whitetail basin in Jefferson County. The Bureau of Land Management used prescribed fire to restore vegetation and hydrologic function in two sub-drainages of this watershed in 2005 and 2006. The objective of our study were to determine 1) if aspen was regenerating naturally in the study area prior to the burn, 2) if the fire increased sucker density, 3) if sucker height was affected by big game or a combination of big game and cattle in the burned drainages, and 4) if sucker density was affected by big game or big game and cattle in the burned drainages.

LITERATURE REVIEW

Aspen Distribution

Aspen is the most widespread deciduous tree in North America (Perala 1990) covering more than 1.5 million ha across the interior western states (Jones 1985). Relatively small aspen woodlands occupy approximately 80,000 ha (Bartos 2001) in riparian areas, meadow fringes, and conifer/grassland transitions throughout western Montana (Mueggler 1985a). While aspen has a wide range across the region, it is relatively rare in the Greater Yellowstone Ecosystem, occupying only 1.4% if the landcover (Brown et al. 2006).

Although aspen stands are typically limited to moist sites, this species is able to inhabit a wide variety of elevations and aspects throughout its range (Jones 1985). Aspen is one of very few plants that can be found in all mountain vegetational zones from the alpine to basal plain (Daubenmire 1943). As a result, aspen is a component of a wide diversity of plant associations (Mueggler 1985a). Unlike other poplar species, aspen is able to survive across such a large gradient due to its high stress tolerance, resulting from unique morphological and physiological traits (Lieffers et al. 2001). Specifically, aspen is different than other trees due to it clonal growth form.

Aspen Reproduction and Growth

Throughout much of their range aspen stands consist of one or more clones of genetically identical individuals (Barnes 1966). In the West seedling events have been considered rare (Kemperman and Barnes 1976, McDonough 1985, Despain et al. 1986), although recent reports indicate seedlings do establish in this region following fire (Shirley and Erickson 2001, Turner et al. 2003, Romme et al. 1995). More often reproduction occurs when young ramets (suckers) are produced from shallow lateral roots (Schier et al. 1985, DesRochers and Lieffers 2001, Frey et al. 2003). Suckers initially depend on the parent root system for water and nutrients (Jones and DeByle 1985b) although this dependence declines as these stems develop their own roots (Zahner and DeByle 1965). Passing of this extensive root system between generations enhances tolerance to climate stress (Lieffers et al. 2001) and gives aspen suckers a growth and survival advantage over seedlings of other species (Jones and DeByle 1985b).

While individual genotypes may persist indefinitely on a particular site (Barnes 1966), adult aspen stems are short lived in comparison to conifers (Jones and Schier 1985). Although stands have been found that contain dominant trees averaging more than 200 years old (Jones and Schier 1985), adult aspen typically succumb to succession, stem decay and disease from age 80-100 (Mueggler 1989, Sheppard et al. 2006). In the Northern Rockies most aspen stands are seral, and will eventually be replaced by Douglas-fir or other conifers in the absence of disturbance (Mueggler 1985b). Aspen is a shade-intolerant species making it very sensitive to competition from conifers (Shepperd 2004). Therefore, aspen clones tend to thrive where regular and frequent disturbance prevents conifer establishment and promotes suckering (Jones and DeByle 1985b).

Fire is one of several disturbances that can stimulate aspen suckering by reducing the flow of auxins to the root system (Schier et al. 1985). Adult aspen trees are highly vulnerable to fire due to their thin bark (Jones and DeByle 1985a). Therefore, prolific suckering can occur after fire kills mature trees (Bartos and Mueggler 1981, Bartos et al. 1991, Kilpatrick 2003). The rate of sucker production and growth varies considerably among stands following adult stem mortality (Frey et al. 2003). Post-fire sucker densities have been reported to range from 3,000 to 147,000 suckers/ha (Bartos and Mueggler 1981, Brown and DeByle 1989, Bartos et al. 1991, Walker 1993, Bartos et al. 1994, Kilpatrick 2003). Growth rates can also vary significantly following disturbance (Romme et al. 1995, Renkin and Despain 1996). Genetic differences among clones explain some of this variation in response (Schier and Campbell 1978). Although less understood, soil characteristics, carbohydrate reserves and environmental conditions also impact sucker production and success (Schier et al. 1985; Frey et al. 2003, Renkin and Despain 1996).

Several soil properties can influence sucker dynamics. Higher soil temperatures result in earlier suckering but not increased sucker numbers (Frey et al. 2003). Excessively wet or dry soil conditions at the time of suckering alter the degree of sucker initiation (Schier et al. 1985). This suggests that differences in precipitation prior to the normal time of suckering may influence suckering as much as the overall moisture class used to describe the site (Frey et al. 2003). Soil composition may also be an important factor affecting aspen suckering and stability. In Colorado, self-sustaining aspen stands are only found on a few subgroups of the mollisol soil order (Cryer and Murray 1992). If stands are not rejuvenated by disturbance, soils hold less water and are more acidic, becoming more conducive to conifer establishment than aspen sucker survival (Cryer and Murray 1992).

Sucker growth is related to availability of soil nutrients. Fertilization with ammonia nitrate and calcium sulfate increased sucker growth but not sucker numbers (Fraser et al. 2002). Growth also increases and decreases with levels of calcium (Lu and Sucoff 2001). Nutrient availability may indirectly increase sucker numbers following disturbance by increasing competitive ability and survival of suckers (Fraser et al. 2002). Growth is also affected by light availability, declining under conditions of low light (Farmer 1963). Therefore, disturbances that remove the entire overstory result in the highest growth rates (Huffman et al. 1999).

Pre-disturbance stand conditions likely play a large role in sucker production. The number of post burn suckers is positively related to the number of pre-burn suckers (Bartos et al. 1991). Sucker density following fire is also positively correlated to pre-burn basal area (Renkin and Despain 1996; Greene and Johnson 1999; Wang 2003). Optimal suckering occurred when pre-burn basal area was approximately 25 m^2/ha which correlated to a root biomass of 20 tons/ha (Renkin and Despain 1996). DesRochers and Lieffers (2001) also demonstrate a relationship between increased root biomass and sucker density although they question which of these conditions is driving the other.

Greene and Johnson (1999) used basal area as an indicator of carbohydrate reserves, which explained 62% of the total variation in post fire sucker density. Others have found growth is strongly correlated with total non-structural carbohydrate (TNC) reserves but not sucker density (Landhausser and Lieffers 2002). TNC levels peak in late summer gradually declining over the winter to the lowest levels around spring leaf flush (Landhausser and Lieffers 2003). Starch concentrations are 10 times higher in fall than spring. As a result of these patterns,sucker growth is greater when stems are cut in the fall than spring due to greater reserves at the time of regrowth (Landhausser and Lieffers 2002). However, Bartos et al. (1991) showed that spring burning produced more suckers than fall burning despite this pattern. They suggest this was due to a hotter and more destructive fall burn. This was supported by Wang (2003) who found plots that burned severely produced fewer and shorter suckers than moderate burns following a spring fire.

Aspen Values

Aspen communities provide numerous ecological and social values, often as a result of the lush understory these stands support. The vegetative associations that are formed in these stands vary considerably with geography and between stands of different seral stages (Mueggler 1985b). However, over 300 plant species can potentially exist in the aspen understory (Houston 1954). Aspen communities in the western US are second only to riparian areas in terms of overall biodiversity (Kay 1997). Over 130 birds and more than 50 mammals use aspen for food or shelter (DeByle 1985c). Game species such as grouse, deer, elk, and moose use aspen, as do several species of concern including the northern goshawk and grizzly bear. As aspen stands change to conifer forest there is a marked change in both flora and fauna (Bartos 2001).

Aspen stands provide excellent grazing for livestock (Mueggler 1985a). Forage production in the aspen understory ranges from 560 kg/ha to over 4480 kg/ha, with most stands producing 1-2,000 kg/ha (Houston 1954). Understory production in the northern Rockies is often dominated by forbs, followed by grasses and few shrubs (Bartos and Mueggler 1979, Mueggler and Campbell 1982). On average 95% of this forage production was considered desirable or intermediate. Forage production decreases progressively as the proportion of conifers in a stand increases (Mueggler 1985a). Stand production can be reduced by 50% when conifers occupy as little as 15% of the basal area (Mueggler 1985b).

In addition to forage production, aspen may have benefits to agricultural operations in terms of water yield and quality. These stands use less water annually than conifers due to a shorter photosynthetic period (Sheppard et al. 2006). This differential use between species may lead to greater downstream water yields from aspen compared to conifer forest (Gifford et. al. 1984). A hydrologic model created by Harper et al. (1981) showed a 5% decrease in water yield when conifers replace aspen. A recent study in Utah found that the difference in downstream yield resulted from higher snow pack and infiltration in aspen stands (LaMalfa 2008.) Furthermore, aspen soils are porous, neutral and have high infiltration rates resulting in higher water quality from these stands compared to conifers (DeByle 1985b).

In addition to forage and water production, aspen woodlands provide numerous social values. Aspen is well recognized for its scenic beauty (Johnson et al. 1985), recreational, and spiritual uses (McCool 2001). In some areas aspen is becoming an important timber product, used to produce dimensional lumber, plywood, particle board, pulp, animal bedding, animal feed, fuel and tourist items (Mackes and Lynch 2001).

Moreover, aspen is now recognized as an “asbestos forest type,” providing natural firebreaks which reduce fire intensity and severity and allow better control by fire managers (Jones and DeByle 1985a, Kilpatrick et al. 2003).

Aspen Decline

In light of these important values, the loss of aspen is a major concern for land managers. Aspen has declined 49-96% across the western United States since Euro-American settlement (Bartos 2001). Some authors have suggested aspen may actually disappear from parts of the west (Kay 1997, White et al. 1998, Shirley and Erickson 2001). However recent regional studies indicate aspen trends are spatially variable, suggesting previous reports of widespread decline are incorrectly based on localized studies (Suzuki et al. 1999, Kulakowski et al. 2004, Zier and Baker 2006, Brown et al. 2006).

Similar inconsistency has been found throughout Montana. Aspen has declined across the state by approximately 64% according to data collected by the Forest Inventory and Analysis program (Bartos 2001). However a regional study of the Greater Yellowstone Ecosystem indicates aspen has only decreased 10% across this area and is increasing in some places (Brown et al. 2006). This is supported by Sankey (2008) who showed aspen are regenerating successfully in the Centennial Valley of southwest Montana. Yet in nearby Gallatin National Forest on the Northern Yellowstone Winter Range aspen decline continues at 0.8% annually (Kimble 2007).

Aspen decline typically occurs when decadent stands are unable to recruit young stems into the overstory and therefore fail to regenerate. Successful regeneration is often defined by the presence of stems that have grown above the reach of browsing ungulates. The following browse heights have been recommended for common herbivores: sheep (3 ft), cattle and deer (5 ft.), elk and horses (6ft. or more) (Jones et al. 2005b). In the Northern Rockies, where elk are the primary ungulate herbivore, regeneration stems are often classified as those stems > 2m tall and < 5 cm diameter at breast height (Kay 1985, Bartos et al. 1991).

There are many different definitions of stand regeneration. By Kay's definition (1985), the ratio of recruitment stems (≥ 2 m tall and < 5 cm DBH) to mature stems (≥ 2 m tall and >5cm DBH) must be ≥ 1.0 for an untreated stand to remain stable or increase in size or density. In a later report Kay (2001) considered stands to be regenerating if they had at least one regeneration stem following fire. By other accounts the presence of any stems that have grown about the accepted browse height indicates successful stand regeneration in unburned stands (Jones et al. 2005a). These evaluations are biased in favor of successful regeneration (Kay 2001). Kilpatrick et al. (2003) defined successful regeneration by the presence of > 2471 stems/ha > 3.1m tall, after 10 years. This criteria also suggested sucker height should increase 0.31 m/year. Suzuki et al. (1999) contend that regenerating stands contain stems that are younger than 30 years old and taller than 2.5 m. Kulakowski et al. (2004) identified persistent aspen stands by the presence of ≥ 100 saplings/ha. Saplings were defined as stems ≤ 3 cm at breast height and ≥ 30 cm in height.

Stem density has also been used to indicate stand regeneration potential or persistence. Undisturbed stands with over 2,470 suckers/ha have the potential to replace themselves (Mueggler 1989; Baker et al. 1997). However, the number of suckers being produced may be irrelevant if the stand is being invaded by conifer or overbrowsed (Mueggler 1989). For example, 14,000-20,000 suckers/ha was not a sufficient number of suckers for stems to reach regeneration height under heavy elk browse following fire in Wyoming (Bartos and Mueggler 1981).

Variability in the estimates of decline and methods of quantifying regeneration suggests generalizations are difficult when describing aspen dynamics. However, the lack of aspen regeneration in many areas is attributed to browsing by native and domestic ungulates, altered fire regimes, changing climatic conditions, or a combination of these factors (Despain et al. 1986, Romme et al. 1995, White et al. 1998, Hessel and Graumlich 2002, Bartos 2001, Kay 2001).

Fire Suppression

Historically, wildfire has been important in stimulating aspen sucker production and maintaining aspen woodlands (Jones and DeByle 1985a). Under the right conditions, fire can also expose mineral soil and encourage seedling establishment leading to new genets (Turner et al. 2003, Shirley and Erickson 2001, Romme et al. 2005). While pre-settlement fire frequencies varied with vegetation type and climate, many parts of southwest Montana likely experienced fires every 25-40 years (Arno and Gruell 1983, Arno and Gruel 1986, Despain et al. 1986). Over the last century suppression efforts have prevented wildfires, burning by indigenous people has been eliminated, and periods of overgrazing have reduced fine fuels needed to carry fires. As a result of human induced changes, fires have not burned in many parts of southwest Montana for over 90 years (Sindelar 1971, Arno and Gruell 1983, Arno and Gruell 1986, Habeck 1992).

Without disturbance stands may fail to produce sufficient suckers to replace decadent adults and to escape herbivory (Despain et al. 1986). The lack of fire has also allowed shade tolerant conifers to expand into adjacent plant communities, sometimes outcompeting aspen stands (Arno and Gruell 1986). Allowing fire to burn may be sufficient to regenerate aspen in some situations (Despain 1986, Kilpatrick and Abendroth 2001, Kilpatrick et al. 2003). Fire may not be necessary to regenerate aspen if browsing is controlled or ungulate densities are low (Kay 2001).

Ungulate Herbivory

While fire suppression has certainly influenced patterns of aspen regeneration, native and domestic ungulates are regularly cited as the primary cause of aspen decline (Kay 1985, Baker et al. 1997, White et al. 1998, Kay and Bartos 2000, Kay 2001). In fact, unprotected aspen stands often fail to regenerate following fire due to ungulate herbivory (Bartos and Mueggler 1981, Bartos et al. 1991, Bartos et al. 1994, White et al. 1998, Kay 2001). Elk, deer, moose and livestock eat aspen twigs, leaves, and even bark (DeByle 1985a) causing decreased sucker establishment and growth rates (Bartos et al.

1991, Kay and Bartos 2000, Kaye et al. 2005). Rubbing and gnawing of fragile aspen bark can also
introduce fungal pathogens and hasten the death of stems (Hart and Hart 2001, Keigley and Frisina 2008).

Elk are most commonly implicated in studies of aspen decline (Baker et al. 1997, White et al. 1998). Elk impacts appear to depend on density and patterns of seasonal use (Suzuki et al. 1999, White et al. 1998, Kay 2001, Barnett and Stohlgren 2001). It is unclear at what density elk have negative impacts on aspen regeneration. Particularly since many of the studies that show elk preventing regeneration were conducted in national parks or other areas known to have abnormally large elk populations (Baker et al. 1997, White et al. 1998, Kay 2001). White et al. (1998) indicated elk use of aspen becomes moderate at a density of 1-3 elk/km^2. Kimble (2007) found only 27% and 16% of aspen stands were regenerating when elk density was 2.4 elk/km^2 and 5.2 elk/km^2. However, elk did not prevent regeneration at densities of 1.36-1.92 elk/km^2 in the Centennial Valley (Sankey 2008). Some stands were able to regenerate near heavily used elk grounds in Wyoming (Barnett and Stohlgren 2001). In Colorado and Wyoming, dendrochronological records indicate historic periods of aspen regeneration correspond to eras when elk density was low (Olmsted 1979, Baker et al. 1997, Hessl and Graumlich 2002) although they do not provide exact densities.

Livestock can also inhibit aspen regeneration, particularly when using the same area as wildlife. Cattle and sheep grazing have historically been the primary consumptive use of aspen in the West (DeByle 1985a). In Utah, wildlife reduced sucker production by 30% in one year, while the combination of wildlife and livestock reduced the number of suckers by 59% (Kay and Bartos 2000). In the Northern Yellowstone Winter Range, fewer stands were regenerating on cattle allotments than adjacent ungrazed areas (St. John 1995, Kimble 2007). A Canadian study found continuous June-July cattle grazing reduced aspen regeneration compared to ungrazed exclosures (Dockrill et al. 2004). Although sheep have become less common on Montana rangelands, they have a much greater impact on aspen suckers than cattle under similar grazing intensities (DeByle 1985a).

Climate

Climatic changes have also been suggested as a potential factor in aspen decline (Despain et al. 1986, Romme et al. 1995). While their clonal roots system allows aspen to avoid drought stress, long term drought can eventually impact growth of suckers and mature stems (Schier et al. 1985). Singer et al. (1998) suggest that declines of aspen may have occurred in Yellowstone National Park despite elk management due to increased temperatures and aridity in that region. Renkin and Despain (1996) found that some stands simply grew too slow in Yellowstone to escape herbivory under the growing conditions there. It is also possible aspen has become more susceptible to damage from herbivores as the climate has changed (Hogg 2001).

On the other hand, some authors do not accept climate as a cause of aspen decline. Hessl and Graumlich (2002) found that decadal periods of aspen regeneration coincide with periods of extremely high precipitation near Jackson Hole Wyoming, yet they argue drought has played little or no role in aspen dynamics. Likewise, exclosure studies indicate that climatic variation did not affect regeneration inside protected plots in Utah or Wyoming (Kay and Bartos 2000, Kay 2001).

Most likely, aspen decline is the result of a combination of ungulate damage, fire suppression, and less than optimal climate conditions. The driving factors probably differ across the west and locally due to varying microclimates and patterns of herbivory. If land managers desire to maintain or restore aspen cover, there is little we can do about climatic conditions. However, we are capable of stimulating sucker production and limiting big game and livestock use of aspen in some areas.

Aspen Restoration

In light of the numerous benefits associated with aspen, land managers have used several techniques to restore declining stands. Management options include doing nothing, removal of existing aspen trees, removal of competing vegetation, prescribed burning, mechanical root stimulation and browse protection (Shepperd 2004).

Prescibed fire is generally considered a viable method for restoring aspen (Sheppard 2001). This technique meets all three requirements of the regeneration triangle; stimulation of suckering, improvement of growing conditions, and protection from ungulates by dispersion (Sheppard 2001). However, post-fire reports indicate burn treatments often have mixed results. In many cases, ungulate herbivory has prevented regeneration despite prolific suckering (Bartos and Mueggler 1981, Mueggler 1989, Bartos et al. 1994, Kay 2001, Shirley and Erickson 2001, Sheppard 2004). Despite an initial increase in sucker density after year 2, herbivory reduced sucker density to pre-burn levels of 10,000-20,000 suckers/ha after year 3 in Wyoming (Bartos and Mueggler 1981). After 12 years sucker density had declined to only 1,500 to 2,400 suckers/ha on that same site due to elk utilization (Bartos et al. 1994). In Oregon, prescribed fire stimulated extensive suckering but nearly all suckers were eliminated within 2 years by elk (Shirley and Erickson, 2001). A survey of 467 burned stands near Jackson Hole, WY indicated that many stands failed to regenerate despite natural and prescribed fire due to moderate to severe elk herbivory (Kay 2001). In Utah, post fire sucker density declined 59% after two growing seasons due to elk and cattle herbivory (Walker 1993). Based on these reports, some researchers have concluded that fire may actually hasten aspen decline when ungulate herbivory is moderate to severe (Bartos el al. 1994, White et al. 1998, Kay 2001).

Under some conditions, aspen stands have regenerated following prescribed fire and clear-cutting near Jackson Wyoming, despite the presence of elk (Gruell and Loope 1974, Kilpatrick and Abendroth 2001, Kay 2001, Kilpatrick et al. 2003). These studies acknowledge that regeneration was usually associated with lower levels of elk use due to topography, distance to feedgrounds or season patterns of use. In Yellowstone, Renkin and Despain (1996) found sucker heights and densities were similar inside and outside of exclosures following fire despite elk. Removal of conifer encroachment has also successfully restored aspen stands in California in the presence of native ungulates (Jones et al. 2005a).

Other reports suggest browse protection is all that is needed for aspen stands to regenerate. Following prescribed fire in Wyoming, suckers protected from browse were the same height on burned and unburned sites (Bartos and Mueggler 1981). Some unburned stands near Jackson Hole regenerated as well as burned stands in areas of little to no elk use (Kay 2001). These studies indicate that fire may not be necessary in some situations to promote aspen regeneration.

Treatment size may also be very important. Campbell and Bartos (2001) suggest burning at least 500-1000 acres for effective dispersal of ungulates. In an area of Wyoming with high elk use, a 500 acre burn was not enough to allow aspen to regenerate (Bartos and Mueggler 1981). Likewise, elk eliminated all suckers following a 20 acre burn in Oregon (Shirley and Erickson 2001). According to Kay (2001), 50% of the stands burned in a 1000 acre treatment were regenerating successfully. His results showed regeneration was less common in smaller burns, except when elk use was low.

Often, multiple methods must be used in conjunction to obtain the desired result. For example, it may be necessary to address browsing before a prescribed fire in order to protect suckers (Campbell and Bartos 2001). Therefore, it is important to evaluate site and stand characteristics before determining the most appropriate treatment (Kilpatrick and Abendroth 2001, Sheppard 2004). Something that is not addressed in these studies, that we feel may be very important, is the amount of aspen available in relation to elk density.

METHODS

Study Area

The Whitetail creek drainage is a small tributary of the Jefferson River in southwest Montana (Figures 1 and 2). This watershed receives approximately 25-30cm of precipitation annually, primarily as rain from April through June. The nearest weather station at Boulder, MT recorded 29.3 cm in 2006 and 26.1 cm in 2007 compared to the 30-year average of 29.6 cm. Soil textures range from excessively well-drained gravels to well drained sandy-loams (Habeck 1992). They are a mixture of course granitic sands originating from the Boulder Batholith and finer Elkhorn Mountain Volcanic material (Veseth and Montagne 1980). Primary habitat types within the drainage include *Festuca idahoensis*/*Pseudoroegneria spicatum* and *Artemisia tridentata vaseyana*/*F. idahoensis* below 1200-1500m and *Pseudotsuga menziesii/Juniperus communis* at higher elevations (Habeck 1992). Our study was conducted at approximately 1600m, in the transitional zone between the sagebrush-grassland and forest types.

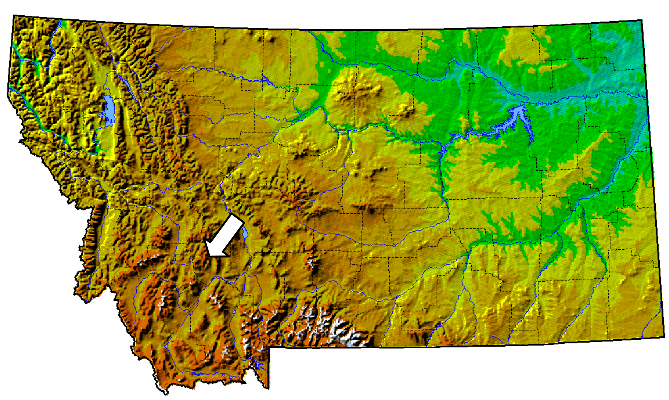

Figure 1: Location of the Whitetail Basin

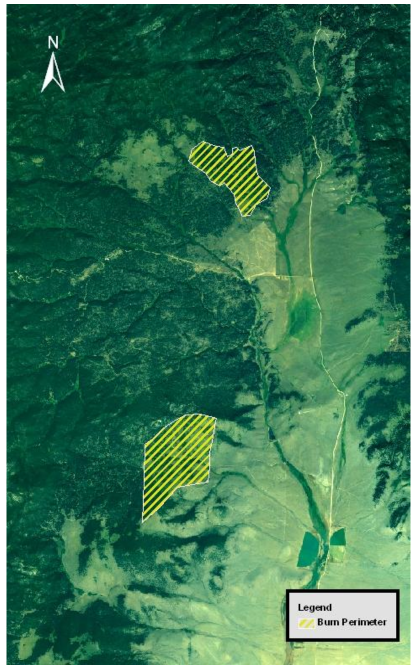

Figure 2: Aerial photograph of the Whitetail Basin with prescribed burn areas in Hay Canyon and Little Whitetail Creek indicated by hashed polygons.

Quaking aspen is the most abundant deciduous tree species in the Whitetail watershed. However, it is sparse, existing as small (< 5 acre) stands along meadow fringes, riparian areas and occasional scree and rock outcrops. Aspen comprises approximately 1% of forest cover although historical accounts suggest it was once much more prevalent (John Joy, US Forest Service, Ret., Personal Communication 2008). Scattered mature aspen and decaying trunks found in the understory of current conifer forest support this assertion. Douglas-fir and Rocky Mountain juniper encroachment is extensive throughout this area (Sindelar 1971, Habeck 1992). The absence of fire has allowed this invasion to persist (Sindelar 1971). Historic fire frequency is estimated at 14-25 years (Arno and Gruell 1986; Habeck 1992), yet most of the area has not burned in the last 80-100 years (Sinderlar 1971, Hessl and Graumlich 2002, Marlow 2006).

The study area includes two cattle grazing allotments, but the Hay Canyon allotment was rested in 2006 and 2007. Grazing occurs on a seasonal rotation, beginning in June and ending in late October. Year round elk density in the surrounding hunting district is 0.36 elk/km^2 (Montana Fish, Wildlife and Parks 2008). This is relatively low compared to other aspen studies (White et al. 1998). The area also supports smaller populations of mule deer, white-tailed deer and moose.

The Whitetail Watershed Restoration Project was a collaborative effort among federal, state, and private interests that used prescribed fire to restore vegetation and hydrologic function in the upper Whitetail basin. Five hundred and twenty-one acres (13%) of the Little Whitetail Creek drainage were burned in the fall of 2005. Approximately 500 acres (15%) of Hay Canyon were burned several months later in the spring of 2006. These burns were administered by the Bureau of Land Management and confined to BLM land. Burn intensity varied across the treatment areas, however there was not a significant decline in large trees or canopy cover in the riparian or upland areas (Tucker 2007). Restoration of decadent aspen stands was one of several goals of this treatment.

Study Approach

This study included four components. First, we surveyed stem size classes from a sample of unburned stands to determine whether or not regeneration was occurring naturally within the watershed. Second, we compared sucker density between these stands and a sub-sample of stands within the burned drainages to see if fire increased sucker density. To address our third and fourth objectives we constructed ungulate exclosures in a sample of five burned stands. Each set of exclosures had three treatment plots allowing us to compare big game and cattle impacts on sucker density and height three years after the fires.

Physical stand characteristics were recorded in each of the five burned stands to identify similarities and differences. Observations of slope, aspect, the presence of conifers (pre and post burn) and post-fire canopy cover were all recorded. Pre-burn aspen basal area was reconstructed following the burn by measuring live and fire-killed adult aspen stems. Soil texture, temperature, pH, and volumetric water content were also measured in 2007 to test for potential differences among protected and unprotected stands.

Regeneration

To determine whether aspen were regenerating naturally in the Whitetail basin we surveyed the stem height distribution of 40 random stands. These stands were selected by identifying unburned stands in the treated drainages and two adjacent drainages on aerial photos and then using a random number generator to pick our sample. Stands were then located in the field with a handheld Global Position System. One 2x30 m transect was established in a representative portion of each stand. All aspen stems were counted along these transects and classified in 4 size classes; suckers (< 1 m tall), saplings (1-2 m tall), regeneration stems (>2 m tall and < 5 cm dbh), and mature stems (>2 and > 5 m dbh. This classification was similar to a Forest Service aspen monitoring protocol (Jones et al. 2005a) and previous regeneration studies (Kay 1985, Kay 2001). To evaluate regeneration in aspen stands in the Whitetail Basin we followed the criteria described by Kay (1985).

Sucker Density

Mean sucker density was calculated by counting all aspen stems < 2 m in height (Bartos et al. 1991) along belt transects (2x30 m) in burned and unburned stands. The 40 unburned stands used in the previously described regeneration survey were also used for this portion of the study. Seventeen distinct aspen stands were then located in the burned drainages by surveying the entire area on foot. Stands were considered distinct if they were more than 30 m apart (Kay 1985), or they were separated by landforms such as ridges or large rock outcrops. Stands were considered burned if the trunks of mature trees were charred. Belt transects were established in a representative portion of each stand. While the burned sample was smaller than the unburned, it represented all available burned stands. More than 17 distinct stands existed in the burned area, but many were not affected by the fire.

A two sample t-test was used to determine whether sucker density differed between burned and unburned stands. The experimental unit for this comparison was the individual stand. Because this t-test assumes data is normally distributed the Shapiro-Wilk test was used to check for normality at an $P \leq 0.05$. The null hypothesis, that sucker density was normally distributed, was rejected for unburned stands (P=0.011). Therefore, transformation of both burned and unburned density was necessary to proceed with the t-test. The Box and Cox method indicated that a square root transformation was optimal for this data set. This agreed with previous studies that also used a square root transformation when comparing sucker density (Bartos et al. 1991, Jones et al. 2005a). Transforming the data also allowed us to use the t-test despite different sample sizes (Quinn and Keough 2002). Analysis was conducted using the R 2.5.1 statistical software (R Development Core Team, 2007). Differences in transformed density were considered significant at $p \leq 0.05$.

Herbivore Impact on Sucker Density

Three-part ungulate exclosures were constructed in five burned aspen stands in the treatment area. Two of the stands were located in the Little Whitetail burn and three were located in the Hay Canyon burn. The number of exclosures was limited due to the relatively small number of stands that burned and the potential for fencing impacts on wildlife and livestock movement. The exclosures each included one plot protected from all ungulate use (No Use), a plot protected from cattle (Big Game only) and a plot open to big game and cattle (Big Game and Cattle) (Figure 3). NU plots were surrounded by a 2.3 m fence consisting of two runs of woven wire (1 m) topped with two strands of barbed wire. The BGO plots were constructed adjacent to the NU plots. These were surrounded by a standard 4-strand barbed wire fence. These fences were raised prior to cattle arriving on the allotment, and dropped following their removal. The BGC plots were adjacent to the other plots and were open to big game and cattle use year round. Each plot measured 30 m x 30 m. Attempts were made to locate plots in portions of each stand that burned evenly and had similar canopy cover and topography.

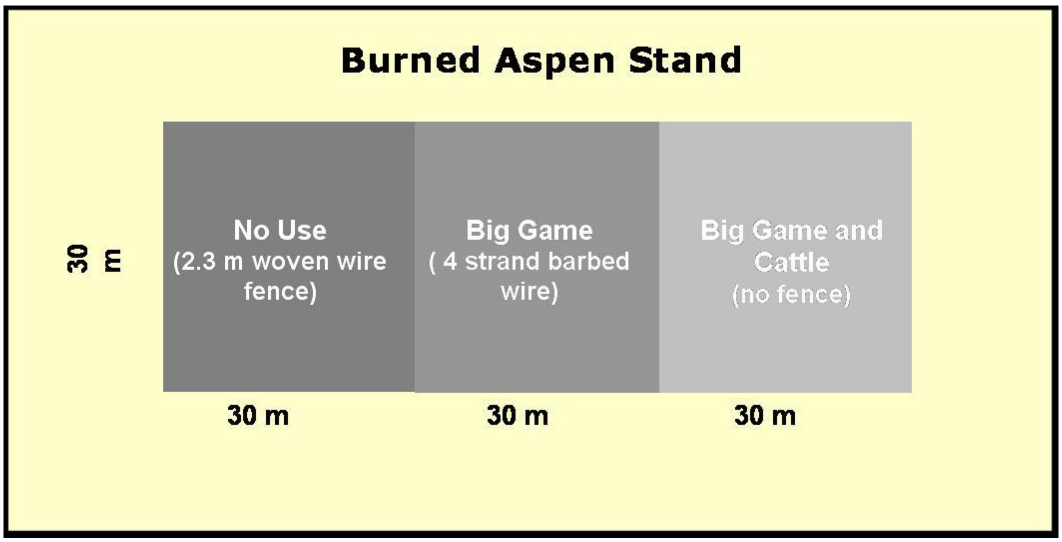

Figure 3: Three-part exclosure design for comparing ungulate utilization of aspen suckers. No Use and Big Game treatment plots were surrounded by fence to control ungulate use. The Big Game and Cattle plots were open to all herbivores.

Two transects (1x43 m) were located in each of the 15 treatment plots to quantify sucker density. All suckers (stems <2 m) were counted along these transects in the fall of 2007 and 2008. Because treatment plots and transects were established prior to suckering, some transects extended into areas with no aspen response. Therefore, density was calculated over the portion of each transect that was actually occupied by aspen suckers. Calculating density over the entire transect would lead to an underestimate of actual suckering response. While counting suckers, we also recorded the number of suckers that had some portion of the current year's growth removed by browsing. We calculated incidence of use by dividing the number of browsed suckers by the total number of suckers.

This study used a randomized complete block design (Quinn and Keough 2002) consisting of five blocks (burned aspen stands) with three treatments (No Use, Big Game Use, and Big Game and Cattle Use) in each block. The experimental unit for this design is the treatment plot. A two-factor analysis of variance (ANOVA) was used to analyze our data (Quinn and Keough 2002). In order to meet the assumption of ANOVA, sucker density was tested for normality using the Shapiro-Wilk test statistic at a predetermined $P \leq 0.05$. We did not reject the null that sucker density data was normally distributed (P=0.46), so no transformation was necessary. In conjunction with ANOVA, the Tukey multiple comparison method was used to determine where differences existed between stands and treatments. This analysis was conducted using the R 2.5.1 (R Development Core Team, 2007) software. Differences were considered significant at a $P \leq 0.05$.

Herbivore Impact on Sucker Height

Mean sucker height was measured in the same three-part exclosure plots described earlier in this report. These exclosures consisted of a No Use (NU) plots, a Big Game (BG) plot and a Big Game and Cattle (BGC) plot. Following the first frost in the fall of 2008, total height and annual growth were recorded for 35 randomly selected aspen suckers in each of the 15 plots. Suckers were selected by pacing two 43m transects in each plot following the *Nearest Plant Method* (USDI 1996). All sucker size classes were included in the sample. A sample adequacy test conducted on preliminary data indicated that a sample size of 35 was more than satisfactory given the variance in stem height (Elzinga et al. 1998).

This portion of the study also used a randomized complete block design (Quinn and Keough 2002) consisting of five blocks (burned aspen stands) with three treatments (No Use, Big Game Use, and Big Game and Cattle Use) in each block. The experimental unit for this design is the treatment plot. We used a two-factor analysis of variance to analyze our data (Quinn and Keough 2002). Because ANOVA assumes the data is normally distributed, sucker height and growth were tested for normality using the Shapiro-Wilk test statistic at $P \leq 0.05$. We failed to reject the null hypothesis of normality for either variable so transformation was not necessary. In conjunction with this ANOVA, the Tukey multiple comparison method was used to determine where differences existed between stands and treatments. This analysis was conducted using the R 2.5.1 software (R Development Core Team, 2007). Differences were considered significant at $P \leq 0.05$.

RESULTS AND DISCUSSION

Regeneration

This survey indicated that very few aspen stands were regenerating in the Whitetail basin. Out of the 40 stands we sampled, only one was successfully replacing the overstory (Kay 1985, 2001). In fact only four stands had any regeneration stems along the transects we sampled. Out of these stands, only one had more than a single regeneration stem.

We also compared our data to the risk assessment key proposed by Campbell and Bartos (2001). According to this key, aspen stands are at risk of decline if they have less than 1235 stems/ha that are 1.5-4.6 m (5-15 ft) tall. Trees of this height would be similar to our regeneration size class. There was not a single stand in our survey that had the suggested number regeneration stems. Again this confirmed that regeneration was not occurring in the Whitetail Basin.

While very few stems were growing tall enough to escape browsing, most stands were still producing a sufficient number of suckers to regenerate (Mueggler 1989). According to Mueggler, an established stand has the potential to regenerate naturally if it produced at least 2470 suckers/ha based on previous studies of aspen dynamics. Out of the 40 stands we sampled, 36 had more than 2470 suckers/ha and therefore had the potential to regenerate. However, Mueggler also stated that this level of suckering is irrelevant if herbivory is severe, or conifer encroachment is occurring. In our study area

Douglas-fir stems were present in most stands and, suckers showed signs of repeated browsing in every stand we sampled.

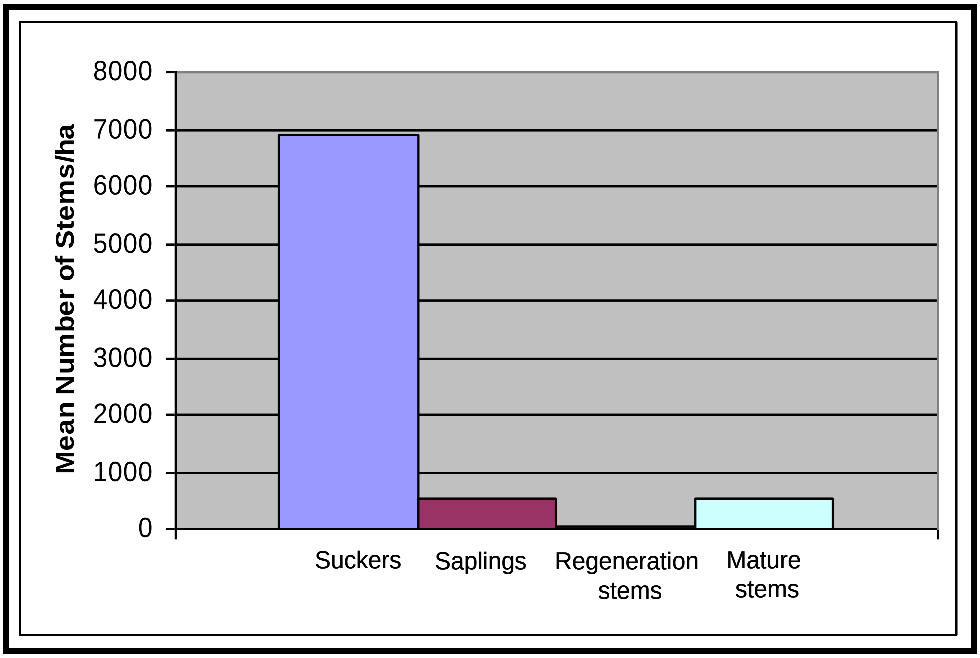

Figure 4: Distribution of mean aspen stem densities by height class in the Whitetail Basin. Sample included 40 random aspen stands. Size classes are as follows: Suckers (<1 m), Saplings (1-2 m), Regeneration stems (>2 m tall, < 5 cm dbh), and Mature stems (>2 m tall, >5 cm dbh)

The distribution of individual stems by size class from unburned stands (Fig. 4) demonstrated that very few suckers were able to grow to sapling size (1-2 m) and even fewer reached regeneration height (>2 m) . The impact of herbivores was evident in this lack of recruitment of suckers into larger size classes. We observed severely browsed, multi-stemmed suckers in every plot. While failure of aspen to grow into the overstory can sometimes be attributed to poor growing conditions (Renkin and Despain 1996), all our stands had decadent aspen in the overstory indicating the sites were capable of producing full size stems.

Sucker Density

There was a highly significant difference in transformed sucker density between unburned and burned stands (*P*=4.9e-07). Therefore, we rejected the null hypothesis that sucker density would not differ. Transformed sucker density was 85% greater three years after the burn than prior to treatment. While transformation was necessary for analysis, actual sucker density is more useful for making comparisons to previous studies (Table 1).

Table 1. Mean sucker density comparison between burned and unburned stands in the Whitetail Basin. T-test was performed on transformed density values. Differences are indicated by letters at $P \leq 0.05$.

		Transformed Density			
Site	**Sample size**	**Range (suckers/ha)**	**Mean (suckers/ha)**	**SD**	***P*-value**
Unburned	40	2600-13400	8200[a]	2600	
Burned	17	8700-21500	15200[b]	3800	0.00000049

		Actual Density		
Site	**Sample size**	**Range (suckers/ha)**	**Mean (suckers/ha)**	**SD**
Unburned	40	700-18000	6900	4300
Burned	17	7500-46000	24500	11700

Three years after the fires, mean sucker density (24500 suckers/ha) was well within the range previously reported following treatment (2,000-147,000 suckers/ha) (Walker 1993, Bartos and Mueggler 1981, Bartos el al. 1991, Romme et al. 1995, Renkin and Despain 1996). While many studies have confirmed that fire stimulates prolific suckering, initial sucker production often declines rapidly due to herbivory (Bartos and Mueggler 1981, Bartos et al. 1991, Bartos et al. 1994, Shirley and Erickson 2001). In many of these situations ungulate browsing has reduced sucker density to pre-burn levels or lower. In our study, actual mean sucker density was still 255% higher in burned than unburned stands after a similar time period to Bartos and Mueggler (1981) and Shirley and Erickson (2001). It appears increased sucker density throughout the burned areas was sufficient to disperse herbivore pressure and prevent sucker mortality by reducing the chance an individual sucker is browsed at all or browsed repeatedly.

The question then becomes whether current post-fire sucker density will eventually allow for stand regeneration. A model of aspen response to fire estimated 10-20,000 suckers/ha is needed to firmly reestablish a treated stand (Bartos et al. 1983). However, 10-20,000 suckers/ha were not enough to escape browse pressure in a Wyoming study (Bartos et al. 1994). Elk density was presumably much higher in the area studied by Bartos than in the Whitetail basin, so by this standard, overall mean sucker density after three years (24500 suckers/ha) would be sufficient for stand reestablishment. However, individual stand densities varied (7500-46000 suckers/ha) so some stands may not be able to escape browsing.

Although our study was not designed to compare individual stands, previous reports indicate differences in sucker production could be related to several factors. One possibility may be the intensity of the burn. Stands that burn more severely often produce fewer suckers (Bartos et al. 1991, Wang 2003). Intense burning reduces sucker production by killing sections of the root system (Schier and Campbell 1978). While we did not measure fire intensity, observations indicate that there was more scorched and bare ground in stands with fewer suckers. Genetic differences, soil conditions, carbohydrate reserves and other environmental conditions also believed to influence sucker production (Schier and Campbell 1978, Schier et al. 1985, Frey et al. 2003).

Herbivore Impact on Sucker Density

Mean sucker density was not significantly different among No Use, Big Game Only and Big Game and Cattle treatments after three growing seasons (Table 2). Density was different among the 5 burned stands (Table 3).

Table 2. Comparison of mean sucker density among herbivore treatments in burned portions of the Whitetail Basin. Differences were not significant at $P \leq 0.05$.

Treatment	**Sample size**	**Range (suckers/ha)**	**Mean (suckers/ha)**	**SD**	***P*-value**
No Use	5	5000-40500	29700	14100	
Big Game	5	3100-35500	21800	12300	
Big Game/Cattle	5	10400-46300	31000	13900	0.12

Table 3. Comparison of mean sucker density among burned aspen stands in the Hay Canyon (HC1-3) and Little Whitetail (LWT1 and 2) sub-drainages of the Whitetail Basin. Differences are indicated by letters at $P \leq 0.05$.

Stand	**Sample size**	**Range (suckers/ha)**	**Mean (suckers/ha)**	**SD**	***P*-value**
HC1	3	19200-39800	30800[a]	10600	
HC2	3	25600-40500	33900[a]	7600	
HC3	3	21700-33000	29200[a]	6500	
LWT1	3	29400-46300	37300[a]	8500	
LWT2	3	3100-10400	6200[b]	3800	0.002

We found that neither big game or the combination of big game and cattle, had a negative impact on post-burn sucker density in the Whitetail basin after three years. Although not significant, sucker density was actually highest in the areas used by both classes of ungulate. While observations confirmed that both elk and cattle were browsing aspen suckers, it appears that use was low enough to minimize mortality. The lack of difference between BG and BGC indicates that cattle had a lesser impact on sucker density to date. If big game and cattle impacts were additive, we would expect there to be a difference between big game and big game and cattle treatments. However, cattle only used three BGC plots for one year so further impacts may emerge after more time.

We did not expect sucker density to be similar among treatments based on earlier reports of herbivore impacts. As discussed previously, ungulate herbivory has caused rapid declines in sucker density in many situations. In a study very similar to ours, sucker density declined significantly in plots used by big game only and plots used by big game and cattle, while it increased in protected plots after a similar time period in Utah (Walker 1993). The decline in sucker density was significantly greater in areas used by both big game and cattle than it was in plots used by just big game or just cattle. They did not find a difference in sucker density between big game only and cattle only treatments. In a long-term project that compared three separate burned stands in Wyoming, sucker density was greatest in the protected stand, intermediate in a stand used only by big game and lowest in a stand used by big game and cattle (Kilpatrick 2003).

However, other studies agree with our findings. Sankey (2008) found no difference among areas with different assigned grazing levels in Montana’s Centennial

Valley. A study from Yellowstone National Park reported mixed results when sucker densities were highest inside some exclosures but highest outside of others 1 year after fire (Renkin and Despain 1996). This relationship continued for the next 5-7 years.

We did find a significant difference in sucker density among the five stands (*P*=.0004). Multiple comparison analysis indicated that four of the five stands had very similar densities. Of the stand characteristics we measured all five stands appeared very similar (Table 4). Therefore, we can only presume why the one stand lagged behind in sucker production. It may have been related to the season or severity of the burn. Fall burns generally produce fewer suckers than spring treatments due to higher burn severity (Bartos et al. 1991, Wang 2003). While the outlying stand (Little Whitetail 2) was burned in the fall, the adjacent stand (Little Whitetail 1) was also burned in the fall and produced an equivalent number of suckers to the spring burned stands. It is also possible the stand was too decadent to produce a large numbers of suckers. Dominant trees in the stand were more than 120 years old as determined by core samples and disks collected from a sample of adult stems. It was also evident that many of the surviving adult trees in the area had a fungal infection. It is possible this infection was present prior to the fires.

Table 4. Description of site conditions for burned stands in the Hay Canyon (HC) and Little Whitetail (LWT) sub-drainages of the Whitetail Basin. Soil temperature and water content from May are used because suckering would be most influenced by soil conditions at this time of year.

	Stand				
	HC 1	HC 2	HC 3	LWT 1	LWT 2
Season of burn	Spring	Spring	Spring	Fall	Fall
Soil texture	Sandy loam	Sandy Loam	Loamy Sand	Loamy Sand	Sandy Loam
Soil temperature at 10cm (mean for May 2007)	13.3 ^{o}C	14.4 ^{o}C	14 ^{o}C	13 ^{o}C	14.8 ^{o}C
Volumetric soil water content (May 2007 at 20cm)	0.1859	0.1245	0.2488	0.1963	0.265
Post-burn soil pH	6.7	6.9	6.5	6.4	6.5
Pre burn basal area (m^2/ha)	3.12	11.7	16.7	4.96	11.68
Post-burn canopy cover (%)	10-30	0-5	70-80	0	15-30

Herbivore Impact on Sucker Height

There was a significant difference (P= 0.003) in mean sucker height among NU, BG and BGC treatments (Table 5). Therefore we rejected our null hypothesis. Multiple comparison analysis indicated mean sucker height was different between NU and BG (P= 0.012) and BGC (P= 0.003) plots. However, there was not a significant difference in mean sucker height between BG and BGC treatment plots. There also was not a significant difference in height among the five stands.

Table 5. Comparison of mean sucker height among herbivore treatments in burned portions of the Whitetail Basin. Differences are indicated by letters at $P \leq 0.05$.

Treatment	**Sample size (plots)**	**Range (cm)**	**Mean (cm)**	**SD**	***P*-value**
No Use	5	99-124	108[a]	10	
Big Game	5	76-103	88[b]	11	
Big Game/Cattle	5	72-97	83[b]	10	0.003

Our results indicate big game browsing had a negative affect on mean sucker height. This agreed with studies from Utah and Wyoming that also found mean sucker height was lower in areas used by big game following treatment (Walker 1993, Kilpatrick 2003). It is logical that stem height would be lower in areas used by ungulate herbivores than protected sites. However, our findings differ from Renkin and Despain (1996) who found no difference in sucker height inside and outside of exclosures in Yellowstone National Park. Our results also contradict White et al. (1998) who state elk impacts on aspen will not be evident when elk density is less than 1 elk/km^2. This suggests that the aspen stand size may be more important than just the number of elk in a given location. In fact when aspen is sparse elk density may not matter at all as a few elk could potentially seek out all aspen stands and have a negative impact on regeneration.

Contrary to our expectations the combination of big game and cattle did not have a cumulative affect as found in these other studies (Walker 1993, Kilpatrick et al. 2003). If cattle impacts were equivalent to big game we would expect a significant difference in height between these treatments. The lack of difference in our plots suggests that cattle were having a minimal impact on sucker height in this area. However, cattle were present on only two of our BGC plots in 2006 and 2007, and all five plots in 2008. And while the difference was not significant, mean sucker height was 5 cm less in the BGC treatment plots. Therefore, continuation of this pattern may produce significant difference in height after a longer period of use.

While we do not have data to support it, our observations indicated cattle browsed very few suckers in Hay Canyon until most of the grass biomass was removed from the riparian areas. The cattle were then removed before they ate very many aspen stems. Even fewer suckers were browsed by cattle in the spring grazed Little Whitetail allotment. Low sucker use seemed to correspond with the presence of abundant palatable grass, and removal of the cattle while sufficient residual forage remained. These observations agreed with DeByle (1985) that, if cattle grazing is light to moderate, use of aspen will be as well. This suggests cattle can be effectively managed to minimize impacts on aspen regeneration.

While our data showed that big game and big game and cattle browsing impacted sucker height, we do not feel ungulate herbivory will prevent regeneration. The height distribution of individual stems indicates that suckers are successfully escaping herbivory outside of the exclosures and approaching regeneration height (Fig. 5). Furthermore, in 2008, we observed individual stems taller than 2 m (browse height) inside and outside of our exclosures. This would indicate successful stand regeneration by some definitions (Kay 2001, Jones et al. 2005a).

While our mean sucker height indicates these stands need several years to surpass browse height, by including first year suckers with other size classes our mean is much lower than if we only measured dominant stems. Despite this sampling method our mean heights were higher in all three treatments than those measured on four burns in Wyoming (Bartos et al. 1994), and similar treatment plots in Utah (Walker 1993). Mean heights were also greater than the median height found by Kilpatrick et al. (2003) (0.3-0.6m), in stands that appeared to be regenerating after a similar time period further supporting our conclusions that stands will likely be able to regenerate in burned portions of the Whitetail Basin.

Annual height increase for our treatments also suggests that these stands have the potential to regenerate. Kilpatrick et al. (2003) used an annual increase of 31cm as the objective for successfully regeneration based on recommendations from Dale Bartos. Annual height increase for NU, BG and BGC plots was 35, 33, and 28 cm respectively. While there was a statistically significant difference in height increase between NU and BGC (P=0.04), we do not feel it is enough to prevent regeneration in the BGC plots. This increase was only a few centimeters less than the recommendation of 31 cm. Furthermore, Kilpatrick et al. (2003), who presented this recommendation, reported an annual height increase of only 18.3 cm in successfully regenerating stands. Therefore, this further supports our assertion that ungulate herbivory does not appear to be preventing regeneration to date.

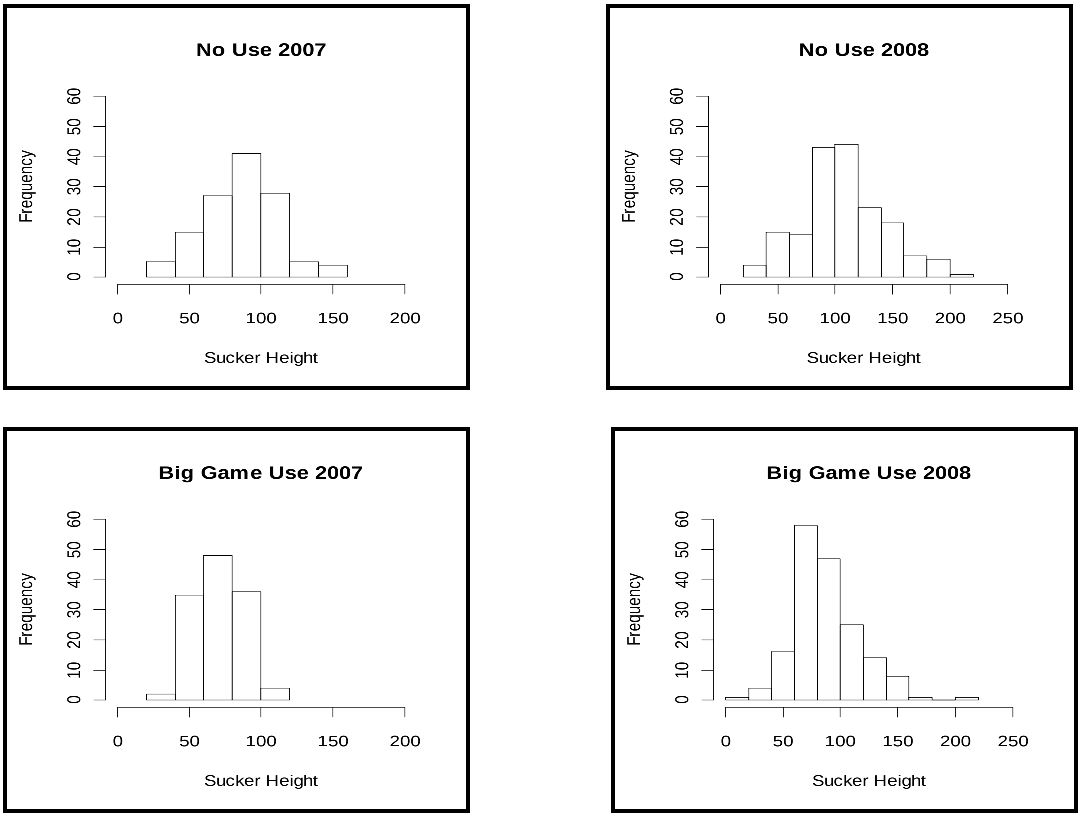

Figure 5. Histograms of individual stem heights by treatment and year in the burned portions of the Whitetail Basin. Recruitment of stems into larger size classes between 2007 and 2008 indicates escape from herbivory.

Summary

The regeneration survey of aspen in the Whitetail basin indicted most stands were not regenerating despite relatively low big game populations (0.36 elk/km^2) (White et al. 1998). The height distribution of stems sampled in this survey demonstrated that few suckers were growing beyond 1m, and even fewer were growing tall enough to escape browsing and replace mature stems. However, our comparison of these stands to 17 burned stands showed that prescribed fire had increased sucker density significantly. It is possible that the increase in suckers across the burn would be sufficient to disperse ungulate pressure and allow stems to escape herbivory.

We found no difference in density among no use, big game, and big game and cattle treatments. This suggests herbivory was not having a negative affect on sucker density as others studies have reported 2-3 years after fires. We also compared sucker height among these herbivore treatments. We found suckers were shorter in areas used by big game and big game and cattle than in full exclosures. However, we do not feel the difference would be enough to prevent unprotected stands from regenerating over the next several years. This assertion was supported by our observations of stems that had already surpassed browse height and sufficient annual height increases outside of exclosures after three years.

Overall it appears that prescribed fire has increased the likelihood that these stands will regenerate. After three years, sucker density, height and annual height increase are sufficient for regeneration according to previous reports. However, several more years are needed for a sufficient number of stems to grow beyond browse height to test this assumption. Therefore, we recommend that exclosures be kept in place and monitoring is continued annually to follow the transition of these stands from suckers to mature stems. Only then can we confidently argue that prescribed fire is effective for aspen restoration given the big game density and cattle pressure in this area.

Overall these findings indicate that ungulate pressure in our study is low enough that when fire produced higher sucker numbers ungulate herbivory was less negative. This supports the assertion that prescribed fire would be effective for regenerating aspen in the Whitetail basin. This suggests that in burned areas current big game densities and cattle stocking rates are acceptable to achieve aspen regeneration.

MANAGEMENT IMPLICATIONS

Managers are under increased pressure to restore ecological integrity on public lands. Particular attention is often paid to problems like aspen decline that have been caused in part by historic management practices. Like many natural resource issues, aspen restoration requires choices to be made among land uses. If healthy aspen stands (and the values associated with these communities) are a priority then we have to choose between a reduction in big game and livestock use, or a return to more natural fire frequencies to regenerate stands. While there is significant opposition to either solution, we feel this project demonstrates that fire can effectively restore aspen with minimal negative impacts on the surrounding community and no change in livestock/wildlife management goals.

We acknowledge that the implications for other management areas are limited due to the size of our study and the specific conditions found in the Whitetail basin. However, many parts of southwest Montana have similar elk densities, grazing programs and land use histories as the Whitetail basin. For aspen management in these areas we suggest the following:

- In the absence of fire, current big game densities and cattle stocking rates will limit aspen regeneration. To maintain or increase aspen cover active intervention is necessary.

- Treat a large enough area. Campbell and Bartos (2001) recommend 500-1000 acres. The Whitetail burn was approximately 1000 acres spread across 2 drainages. We feel this was sufficient to disperse big game and allow for variability in stand response.
- Consider mechanical harvest options. Previous reports have demonstrated harvest of aspen or encroaching conifers can be equally effective for aspen restoration. Cutting may also prevent root damage from severe burning and has potential economic benefits.
- If possible, tailor grazing rotations so important aspen sites are used early in the summer when other forage is abundant and highly palatable. Additional management may be needed for pastures grazed in late summer and fall when cattle are more likely to eat succulent aspen stems. Specifically, cattle need to be removed quickly once they start eating aspen.
- Finally, an effective management plan would likely treat large sections over multiple years, therefore diffusing ungulate use and maintaining a mosaic of aspen age classes and the values associated with different successional stages.

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