Ecological and Cultural Significance of Burning Beargrass Habitat on the Olympic Peninsula, Washington

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ABSTRACT

To conserve or restore culturally significant plants, one must consider the important role that indigenous land management techniques have played in maintaining habitats of those species. Beargrass (*Xerophyllum tenax*) is a basketry plant used by Native Americans and is reportedly declining in traditional gathering sites. Many low-elevation beargrass sites on the Olympic Peninsula in Washington were maintained as savannas and wetland prairies through anthropogenic burning prior to European settlement. This study measures short-term (1 and 2 y) effects of reintroducing prescribed burning (both low and high severity) and manual clearing on beargrass growth and reproductive success—flowering, vegetative reproduction, and seedling establishment. High-severity fire led to a significant increase in beargrass seedling establishment and vegetative reproduction over two years but a decline in beargrass cover. Low-severity fire also decreased beargrass cover, but did not significantly affect shoot production or seedling establishment. In areas where vegetation and coarse woody debris were manually cleared, beargrass cover decreased, while shoot production and flowering increased. Neither low-severity fires nor clearing plots affected beargrass seedling establishment. Results indicate that fire is a useful tool for enhancing low-elevation beargrass populations in this region.

Keywords: basketry, beargrass (*Xerophyllum tenax*), prescribed fire, restoration, savanna

The persistence of many indigenous traditions is dependent upon the availability of culturally significant resources (Anderson 1996a, 1996b, 2005). Baskets made from local plants, for example, strengthen cultures by preserving traditions, reinforcing communities, and providing income (Shebitz and Kimmerer 2005). Indigenous basketmakers throughout the United States, however, have reported a decline in abundance and quality of basketry plant material at historic gathering sites. The absence of traditional burning over the past century is a potential cause for the decline in abundance of some basketry plant species, including deergrass (*Muhlenbergia rigens*) (Anderson 1996b), sweetgrass (*Anthoxanthum nitens* [=*Hierochloe odorata*]) (Shebitz and Kimmerer 2004), and beargrass (*Xerophyllum tenax*) (Hunter 1988, Rentz 2003, Shebitz 2005).

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To conserve or restore culturally significant plants, it is often necessary to restore land management techniques that formerly maintained habitats of those species. Prior to European contact, Native Americans played a complex role in shaping landscapes through activities such as burning and harvesting that were regular and long term (Lewis 1993, Stewart 2002, Anderson 2005). These practices often substantially altered the composition and structure of plant associations and influenced species distributions (Anderson 1996a, 2005). Management practices also led to the establishment of vegetation communities in locations where they would not otherwise have existed (Boyd 1999). For example, despite the high precipitation and low natural fire frequency in western Washington, prairies and savannas were dominant components of the Puget Sound Lowland landscape for millennia prior to European settlement due to anthropogenic management (Norton 1979, Lewis and Ferguson 1999, Peter and Shebitz 2006). Indeed, anthropogenic fire was the most important tool of environmental manipulation throughout the Pacific Northwest (Boyd 1999).

Anthropogenically Maintained Beargrass Habitat in the Pacific Northwest

Native Americans of the southeastern and western Olympic Peninsula of Washington have a long history of managing the landscape to ensure the availability and quality of desired plant species (Regan 1934, Norton 1979, Boyd 1999). Beargrass (Figure 1) is one example of a culturally significant plant commonly harvested from the Olympic Peninsula lowlands for use in basketry by tribes such as the Quinault and Skokomish (Gunther 1981, Nordquist and Nordquist 1983, James and Chubby 2002). Prior to Euro-American settlement, low-elevation sites where beargrass was gathered were maintained as savannas and wetland prairies through anthropogenic burning (Jones 1936, Peter and Shebitz 2006). Due to the exclusion of both anthropogenic and natural fires since the 1850s, most of these formerly open areas are now covered with dense Douglas-fir (*Pseudotsuga menziesii*) (Norton 1979, Boyd 1999, Peter and Shebitz 2006). Beargrass is believed to have declined in traditional gathering sites over the past few decades in part due to forest encroachment resulting from the absence of fire. In fact, a study by Shebitz and colleagues (2008) found a significant decline in beargrass abundance in the southeastern Olympic Peninsula since 1986.

Low-severity burns have long been used by Native Americans throughout the western United States to enhance the growth of beargrass and provide basketry material (Hunter 1988, LaLande and Pullen 1999, Rentz 2003, Shebitz 2005). In addition to Olympic Peninsula tribes, the Yurok, Karuk, Hupa, Chilula, Upland Takelma, and others burned beargrass periodically and then harvested leaves from the burned clumps one to three years later (LaLande and Pullen 1999, Rentz 2003, Anderson 2005). Historically, burns were low-severity, slow-moving surface fires (Lewis 1993, Hunter 1988, Rentz 2003) that burned old beargrass growth and up to 95 percent of living foliage (Hunter 1988). According to Skokomish oral tradition, on the southeastern Olympic Peninsula, lowland prairies were burned at 2–3 year intervals to maintain a diversity of edible and textile plants and to sustain suitable habitat with a relatively open canopy (Peter and Shebitz 2006). Beargrass generally was in the savanna-like periphery of

Figure 1. Beargrass (*Xerophyllum tenax*) growing in the Quinault site in June 2003 prior to treatments. Photo by Daniela Shebitz

prairies, occurring with some canopy cover. According to interviews with Skokomish and Quinault elders, this beargrass habitat was often not the target of the burns but was affected when a prescribed fire's boundaries were extended because of environmental conditions such as wind. The primary reason for the burns was generally to manage land for game such as blacktail deer (*Odocoileus hemionus columbianus*) and Roosevelt elk (*Cervus elaphus roosevelti*) in an otherwise forested landscape. A secondary reason was to maintain food plants such as camas (*Camassia quamash* var. *azurea*). Camas was traditionally a fundamental food crop for western Washington tribes (Gunther 1981), and maintaining an open habitat for camas was a primary reason for burning in the Pacific Northwest (Boyd 1999).

Beargrass Structure and Response to Fire and Management

Beargrass is an evergreen, perennial herb in the Melanthiaceae family (Liliales) (Rudall et al. 2000, Vance et al. 2004). Basal leaves grow close to the ground in large, dense clumps or tussocks (Cooke 1997, Higgins et al.

2004). The leaves can be up to 60 cm long and 5–10 mm wide at the base, gradually tapering to a narrow and stiff tip (Maule 1959, Henderson et al. 1989). Beargrass has a tuber-like woody rootstock with cord-like roots (Maule 1959). The species reproduces both vegetatively by rhizome and sexually by seed. The rhizomes are 1–2 cm thick (Hitchcock and Cronquist 1973), and each vegetative shoot arises from a basal meristem (Vance et al. 2004). From May to late summer, beargrass flower stalks grow to 150 cm tall and are topped with a club- or cone-shaped inflorescence of white flowers, each measuring approximately 0.13 cm across (Henderson et al. 1989, Cooke 1997, Munger 2003). The inflorescence may bear between 150 and 400 flowers (Vance et al. 2004).

The survival of individual plants through fire is determined by various life-history, anatomical, and physiological characteristics. The meristem is positioned within the leaf base and is the only part of the rhizome that produces new growth. This meristem is found near the interface of organic horizons and mineral soil and therefore is sensitive to intense fires (Bradley 1984). Basal beargrass leaves limit the transfer of heat to the meristem if the leaves are moist (Bradley 1984, Rentz 2003, Whelan 1995). Under dry conditions, however, leaves may increase the heat delivered to the base of the rosette (Bradley 1984). Tussocks may also accumulate leaf litter, which can continue to burn after a fire passes, thus increasing soil temperatures and causing mortality (Clark and Wilson 2001).

Beargrass basketweavers generally desire straight, long, and pliable leaves. Rentz (2003) found that beargrass in burned areas exhibited a reduction in support fibers along the adaxial and abaxial surfaces and secondary wall thickness in fibers, making leaves more pliable. It is believed that beargrass responds to the increased nutrient availability in burned areas. As nutrient properties return to their normal state over time, leaves become less pliable, reflecting the short-term nature of the changes (Rentz 2003). The postfire increase in available nutrients can stimulate leaf growth in situations where nutrients normally limit productivity (Whelan 1995), as is the case in much beargrass habitat (Henderson et al. 1989).

Clearcutting followed by broadcast burning in high-elevation sites has not been found to increase beargrass abundance. On the contrary, after a clearcut, logging, and slash burning experiment in old-growth Douglas-fir forests of the Cascades, beargrass did not recover pretreatment abundance after more than 20 years (Halpern and Spies 1995). Similarly, after clearcutting and broadcast burning in the grand fir/Oregon boxleaf (*Abies grandis/Paxistima myrsinites*) association, beargrass recovery was found to take up to 23 years (Crane 1990). Laursen (1984) attributes the negative response of beargrass to clearcutting and burning to its sensitivity to competition from understory shrub species.

Most research that has been conducted with beargrass was among high-elevation populations as opposed to the low-elevation populations involved with this study. A common finding is that high-elevation beargrass flowers bloom within the first year following fire (Kruckeberg 2003). Based on work on beargrass in the high elevations of Mount Rainier, Maule (1959) hypothesized that, in addition to canopy density, soil temperature was a limiting factor controlling flowering rates. Limited work has been done since these studies, however, to investigate the effects of fire on beargrass flowering rates.

Objectives and Predictions

This study was undertaken to determine if prescribed burning would help reverse the observed declines in the abundance of low-elevation beargrass. We predicted that burning would result in increased vegetative reproduction, growth, flowering rates, and seedling establishment. In addition, we compared the effects observed following fire with a manual treatment, since reintroducing burning might not be possible in some instances. Treatments were applied in the autumn, the same time of year that Native Americans historically burned the area (Peter and Shebitz 2006). By examining the effects of fire and manual clearing of vegetation separately, we can assess the extent to with competition from understory species is influencing beargrass growth and reproduction. We set up four experiments to examine different aspects of this hypothesis (Table 1).

Experiment 1 provided a case study on the effects of a large-scale, high-severity prescribed fire on existing beargrass. We predicted that the high-severity fire would increase the vegetative reproduction, growth, and flowering rates of existing beargrass individuals.

Experiment 2 determined the effect of high-severity fire on beargrass seedling establishment. We predicted that with equal amounts of seed applied to burned and unburned plots, there would be a higher seedling establishment rate in the burned plots.

Experiment 3 tested if low-severity burns will influence beargrass vegetative reproduction and, secondly, determined if the stimulus of fire or decreased competition influence beargrass reproduction and growth. We predicted that beargrass vegetative reproduction, growth, and flowering rates would increase in both low-severity burn plots and in plots manually cleared of vegetation.

Experiment 4 tested if low-severity fire or reduced competition would influence beargrass seedling establishment. We predicted that with equal amounts of seed applied to lightly burned, manually cleared, and control plots, the highest seedling establishment rate would be in lightly burned plots.

Table 1. A summary of the four experiments in this study, including objectives, methods, and key results.

	Experiment			
	1	2	3	4
Year Started	2003 (1 and 2 year data)	2003 (1 and 2 year data)	2004 (1 year data)	2004 (1 year data)
Site	Skokomish	Skokomish	Skokomish and Quinault	Skokomish and Quinault
Treatments	Burn and reference	Burn and reference Seeding and no seeding	Burn, manually clear, and reference	Burn, manually clear, and reference Seeding and no seeding
Fire severity	High	High	Low	Low
Beargrass Life Stage	Adult	Seedling	Adult	Seedling
Objectives	Effect on existing beargrass growth and reproduction	Effect on beargrass seedling establishment	Effect of fire and reduced competition on existing beargrass growth and reproduction	Effect of fire and reduced competition on beargrass seedling establishment.
Measurements	Beargrass: number of shoots (density), number of inflorescences All species: % cover	Number of beargrass seedlings	Beargrass: number of shoots (density), number of inflorescences; leaf length, plant height and widths of 5 plants All species: % cover Quinault: number of shoots on and off hummocks	Number of beargrass seedlings
Sample Size	5 replicates (*n* = 10)	3 replicates (*n* = 12)	6 replicates (*n* = 18)	5 replicates (*n* = 30)
Analysis Method	Mann-Whitney *U* test	Mann-Whitney *U* test	One-way ANOVA and Tukey HSD	Kruskal-Wallis test
Key Findings	1. Beargrass and shrub cover were significantly lower in burned plots (2nd year). 2. Forb cover and beargrass shoot density were higher in the burned plots.	Significantly higher seedling establishment in burned plots (both years).	Skokomish: Mean number of new beargrass shoots highest and shrub percent cover lowest in manually cleared plots. Quinault: No significant differences among treatments.	No significant differences among treatments at either site.

Study Site and Methods

Field Sites

We conducted the research at two sites on the Olympic Peninsula in Washington: the North Fork of the lower Skokomish River basin ("Skokomish"), in the eastern foothills of the Olympic Mountains (elev. 750 m), and the Quinault lowlands ("Quinault") of the western Peninsula (Figure 2). The Skokomish site was located on land that historically belonged to the Skokomish tribe and now is managed by the USDA Forest Service, Olympic National Forest (ONF). The site was located on a 13.35 ha savanna restoration project that was initiated by the Forest Service in 1995. Forest succession on the former Douglas-fir (*Pseudotsuga menziesii*) savanna and prairies has virtually eliminated obligate prairie species. Current vegetation is primarily coniferous forest of the western hemlock/salal/beargrass association (*Tsuga heterophylla*/*Gaultheria shallon*/*Xerophyllum tenax*). Western hemlock and Douglas-fir are the dominant overstory species, and the understory consists primarily of salal, cascade Oregon grape (*Mahonia nervosa*), sword fern (*Polystichum munitum*), bracken fern (*Pteridium aquilinum*) and beargrass (Henderson et al. 1989). Vascular species with the highest pretreatment percent cover in plots for all experiments at the Skokomish site were salal (50%), beargrass (12%) and bracken fern (11%). The primary tree species throughout the unit was Douglas-fir.

Soils in the southeastern Olympic Peninsula are typically low in nitrogen and potassium and are mostly shallow and derived from colluvium. The water-holding capacity of the soil is generally low (Henderson et al. 1989, Shebitz et al. 2008). The mean average temperature for the area around the Skokomish site is 10.4°C, with an average annual precipitation of 167 cm. In the first year of this experiment, from September 30, 2003, to September 30, 2004, the area around the Skokomish site received 170 cm of precipitation, and from September 30, 2004, to September 30, 2005, it received 142 cm (Weather Underground 2009).

In 2001, the designated restoration unit was thinned to a density represented in aerial photographs from 1929 (15 trees/ha); the image was taken before logging began in the area and depicts a thin forest with openings, suggesting that the area previously had a relatively open canopy (Peter and Shebitz 2006).

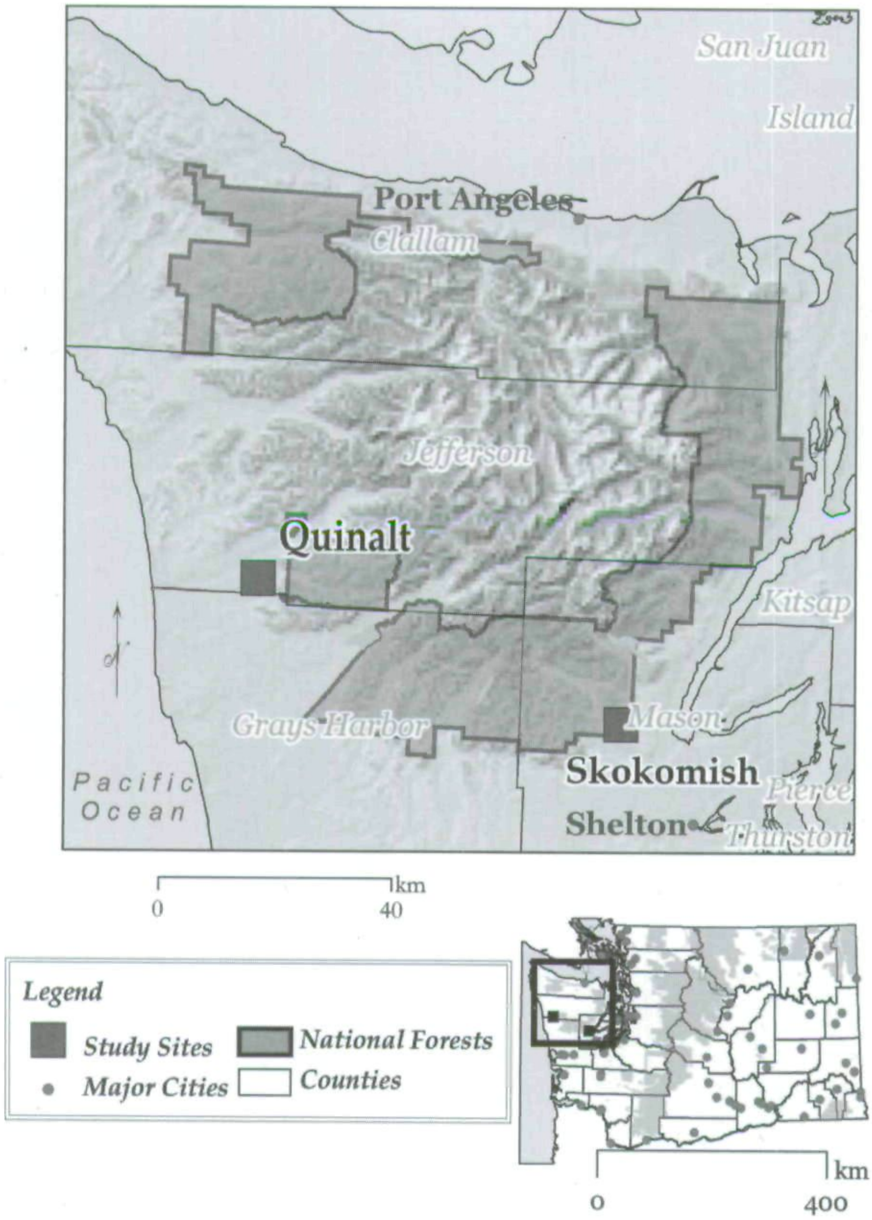


Figure 2. Study site locations on the Olympic Peninsula. The Quinault and Skokomish sites are indicated by gray squares. Map created by Dr. John Dobosiewicz, Kean University

Forest Service personnel burned the site on 30 September, 2003. Due to dry conditions, accumulated organic fuel, and the large size of the area burned, the fire was unexpectedly hot, exceeding 760°C at the ground surface. Almost all organic material in the soil was destroyed, exposing mineral soil throughout much of the site. Despite the high surface temperatures and resulting high-severity effects on vegetation, it was possible to determine where beargrass was located, as meristem remnants remained visible above ground.

The second study site, Quinault lowlands, was added in 2004, since representatives of the Quinault Indian Nation then expressed interest in restoring beargrass habitat and learning how to manage the existing populations. The site is located in an area where the Quinault tribe historically set fires to maintain prairie-like openings in the forests, particularly to sustain plant resources for food, medicine, and basketry (James and Chubby 2002). Prior to Euro-American settlement, the prairie-forest ecotone had a savanna-like structure and was beargrass habitat, according to an interview with a Quinault elder who hunted in the area (J.E. James, pers. comm.).

The Quinault site of the experiment is a bog laurel/Labrador tea/beargrass/sphagnum (*Kalmia microphylla*/*Ledum groenlandicum*/*Xerophyllum tenax*/*Sphagnum* spp.) bog (Kunze 1994, Kulzer et al. 2001) that was clearcut five years before the experiment's initiation. Vascular species with the highest average percent cover were salal (20%), beargrass (15%), and deer fern (*Blechnum spicant*, 8%). This site has a hummocky microtopography with water accumulating between the hummocks, even in the dry seasons. It was observed that some of the beargrass plants were located on the top of hummocks, while others were between the mounds. A unique feature of the site is the presence of camas, which indicated its historic management through fire.

The Quinault site has a sandstone bedrock layer and the most common soil texture is organic matter (Henderson et al. 1989, Shebitz et al. 2008). The elevation of the site is approximately 350 m. Its average annual temperature is 10.6°C, and it receives an average of 175 cm of precipitation a year (Henderson et al. 1989). During this study (from September 30, 2004, to September 30, 2005), the area received 140 cm of precipitation (WRCC 2009).

This temperate rainforest site experiences seasonal flooding and drought (Kunze 1994) and may have been maintained historically by fire (Kulzer et al. 2001). In particular, Native American burning was generally conducted in the late summer and early autumn when conditions were dry and there was not much standing water (Shebitz 2006). It was certainly more difficult to burn this wet site as often as the Skokomish site, but exact information regarding how frequently the Quinault site was burned is not available in the literature and is not recalled by living members of the Quinault Nation. Interviews with Quinault elders who hunt elk in the site indicate that it was burned until the early 1920s. Often the fires intended to maintain a camas prairie moved into the bog that served as beargrass habitat (James and Ellis, pers. comm.).

The area in which the experiment was established on the Quinault Reservation was clearcut in 1987, leaving the unit covered in slash. In 2002, plots for the low-severity fire experiments were established and the burning was conducted by the Quinault Fire Crew. The moist soil in the bog habitat assisted in limiting the severity of the fires.


Figure 3. Olympic National Forest Fire Crew treating the low-severity burn plots for Experiment 3 (September 30, 2004). Photo by Daniela Shebitz

The two field sites have remarkably different hydrology, soils, and plant diversity. While lightning fires were very rare events in the western Olympic Peninsula wetlands, in the Skokomish region, there were large stand-replacing fires that likely affected the study area in 1308, 1508, 1668, and 1701 (Henderson et al. 1989, Peter and Shebitz 2006). The natural fire regime in the dry Douglas-fir forests of the Puget Trough involved small, patchy fires at approximately 70- to 100-year fire return intervals (Agee 1993). Despite these differences, both of the sites were anthropogenically managed through fire. They were selected for this study because, first, they provide low-elevation beargrass habitat, and second, the indigenous people that had historically conducted burns in the area expressed interest in continuing to harvest and manage beargrass at these sites.

Field Methods

We investigated the effects of both a large-scale, high-severity burn and plot-scale, low-severity burns on beargrass vegetative reproduction, growth, flowering rates, and seedling establishment. Vegetative reproduction in this paper followed Keeley's (1981) definition: "any lateral spread with the potential for producing 'new individuals'." We evaluated vegetative reproduction by counting the number of new shoots formed on each beargrass individual. We also assessed growth responses to burning by measuring percent cover, crown area, height, and leaf length, and we calculated crown area as Area = $(W_1 \times W_2 \times 3.142)/4$, where W_1 is the widest diameter of foliar crown material and W_2 is the perpendicular diameter (Pendergrass et al. 1999). We measured plant height from the ground to the highest leaf, excluding flowering stalks. Leaf length was determined by averaging the length of the five longest leaves, since it is those leaves that would be used in basketry. Flowering rates were calculated as the number of inflorescences in each plot, and seedling establishment as the number of seedlings in each plot. Seedlings were easily recognizable, since they were farther than 1 m from an older beargrass plant, approximately 4 cm tall, and with leaves approximately 3 cm long. Vegetative shoots, on the other hand, were generally larger and arose from the meristematic region of an existing plant. Plant species lists were compiled at the field sites using nomenclature from the USDA PLANTS database (USDA NRCS 2005), and they are available from the first author.

There are differences in plot design and layout among the four experiments by virtue of logistical constraints in conducting the treatments. In the two experiments investigating vegetative reproduction, plots were established using a stratified random sampling technique. Plots were established randomly in areas that had a minimum of five percent of the plot occupied by beargrass. In the two experiments investigating seedling establishment, plots were established randomly in areas that had no beargrass but were no more than 10 m from beargrass and physically resembled the adjacent beargrass habitat.

Experiment 1—High-severity fire effects on beargrass vegetative reproduction. Through this case study, we established ten 5 m × 5 m plots with no canopy cover in the southwestern section of the thinned unit at the Skokomish site. Five plots were scheduled to be burned and were located randomly within the 13.35 ha burn unit. The other five were left as unburned reference plots and were located on a 50 m transect along the western border of the unit where they could be protected from fire.

We established plots in August 2003 and drew a map of the plots noting each beargrass location to assist in relocating individual plants. Within each 25 m^2 plot, we took measurements in each of 25 contiguous 1 m^2 quadrats before treatment. We measured the number of beargrass shoots, number of beargrass inflorescences, percent cover of each species in the plot, and total percent cover of vegetation in the plot for two years following the burn.

Experiment 2—High-severity fire effects on seedling establishment. To assess beargrass establishment rates from seed in relation to high-severity fire, we installed a total of twelve 3 m × 3 m plots. Three replicates were established for each of four seeding treatments at the Skokomish site. Treatment combinations included burn/seed, burn/no seed, unburned/seed, and unburned/no seed. None of the plots contained beargrass before the burn. Since the northwestern section of the burn unit did not have high beargrass cover before the burn, the burn/seed and burn/no seed plots were established randomly throughout this section, while the unburned/seed and unburned/no seed plots were just outside of the northwestern border of the burned area. As for all of the other studies, we estimated the percent cover of all species within each plot before the fire. Six of the plots were burned in the high-severity fire on September 30, 2003 (three of which had seeds applied and three of which did not have seeds applied). On October 2, 2003, we broadcast beargrass seeds at a rate of 93 seeds/m^2 over three of the burned plots and three of the unburned plots. The six remaining plots were left unseeded to determine if beargrass seedlings would become established without manually introducing seed. We recorded the number of surviving beargrass seedlings one and two years after the burn and estimated the percent cover of any other species that were present.

We harvested seeds from nine beargrass plants, all of which were located within 1 km of the experiment at a similar elevation and slope. Seeds were collected on 4 August 2003. A preliminary viability test was conducted by soaking beargrass seeds in water, resulting in approximately 70% imbibition. We then stored the seeds in a seed vault at 15°C and 20% humidity until the day before being broadcast in the plots.

Experiment 3—Stimulation of beargrass vegetative reproduction by low-severity fire. To assess the effects of low-severity fire, we installed six replicates of two treatments (burn and manual clearing) and a reference within the northeast section of the Skokomish site and at Quinault. Each replicate contained three 8 m × 8 m plots (1.5 m buffer on each side), with 1–10 m between replicate plots.

Treatments were applied on September 2004 (Figure 3). Fires were low severity and left most of the beargrass meristems visible aboveground after the burns. In plots manually cleared of vegetation, we used chainsaws, machetes, and weed whackers to remove all aboveground vegetation and coarse woody debris. Beargrass leaves were also cut, but the meristematic region was left intact to replicate the effects of a burn.

We took measurements in each of 25 contiguous 1 m^2 quadrats within each plot prior to treatment. Data collected before treatments and one year after the treatments included beargrass density, percent cover of all species, and the number of beargrass inflorescences. We randomly selected five beargrass plants in each plot for additional measurements: length of the five longest leaves, height to the highest leaf, widest diameter of foliar crown material (W_1), and the diameter perpendicular to this (W_2). We marked these five plants in each plot with an aluminum nail hammered into the ground at each of their bases. Each nail was spray-painted one of five colors so that the beargrass could be relocated and remeasured following treatment.

To help understand the effects of microtopography on beargrass survival, maps were created at each plot in Quinault, noting the extent of hummocks in each of the plot's 25 quadrats. The numbers of beargrass shoots on and off of hummocks were compared using a paired t-test to determine if beargrass survival was related to its location.

Experiment 4—Stimulation of beargrass seedling establishment and growth by low-severity fire. At both the Skokomish and Quinault sites, three treatments were compared: low-severity burning, manual clearing, and no manipulation (reference). We installed five replicate plots of each treatment at both sites in areas that had no beargrass. The plots were circular with a diameter of 1.5 m. Treatments were applied in September 2004. We sowed 60 seeds within the center 1 m^2 of each plot after the plots were treated.

We used seeds harvested from 15 beargrass plants within 1 km of the experiment at a similar elevation and slope. We collected the mature beargrass seeds in late June 2004 and stored them in a seed vault until used. Beargrass seedlings were counted one year after the treatments. Each beargrass seedling measured approximately 2–3 cm wide.

Data Analysis

We tested all data for normality and homogeneity of variances before comparing treatments with the reference plots. Analysis was conducted using SPSS 12.0 (SPSS, Chicago IL). Significance for all experiments was tested against $\alpha = 0.05$.

For Experiment 1, changes in beargrass percent cover and number of shoots and the percent cover of forbs, shrubs, and graminoids were analyzed by treatment by comparing their preburn measurements to those one and two years postburn. Experiment 2 compared the number of beargrass seedlings between the burn and reference plots, also one year and two years postburn. In both Experiments 1 and 2, we used a nonparametric analysis of a BACI (Before-After Control-Impact) design in the form of a Mann-Whitney U test to compare changes in the reference and burn plots, since there was an absence of normality and a lack of homogeneity in the variances. Mann-Whitney U tests are used to test only two groups, in this case the reference and burn plots. We also used the BACI tests to compare the number of beargrass seedlings between the burn and reference plots, also from one year and two years postburn.

Table 2. Results of Mann-Whitney *U* test of seedling establishment experiment conducted at Skokomish comparing burned and reference plots (Experiment 2). Number of beargrass (*Xerophyllum tenax*) seedlings was compared between treatments for one and two years after the burn; *p*-values are for comparisons between burned and reference plots.

	Burn		Reference		*p*-value
	Mean	SD	Mean	SD	
			One year postburn		
No. of beargrass seedlings	78.00	76.27	6.00	7.94	0.05
			Two years postburn		
	78.67	82.71	3.67	6.35	0.05

Figure 4. Beargrass recovering from the high-severity fire in the Skokomish site. Basal shoots from meristem in March, 2004 (six months after the fire). Photo by Daniela Shebitz

We performed a one-way ANOVA on changes in beargrass height, crown area, leaf length, and number of shoots for the five measured individual plants from each plot and for the plot's percent cover of beargrass, all forbs, shrubs, and graminoids in Experiment 3. For Experiments 3 and 4, data from the two sites were analyzed separately. Only data collected from the Skokomish site are provided here, since the Quinault site experienced widespread beargrass mortality and therefore did not yield usable data. Changes from preburn to one year postburn were used in the analysis, and Tukey HSD tests were performed as a post hoc test to compare the treatments. For Experiment 4, we compared beargrass seedling establishment among treatments using a Kruskal-Wallis ANOVA, since the data distribution was not normal and variances were not homogeneous. This test is the nonparametric equivalent of the one-way ANOVA and can be used to test any number of groups.

Results

Experiments 1 and 2—High-severity fire effects on beargrass vegetative reproduction and seedling establishment. Beargrass leaves began resprouting in late February 2004, within five months of the high-severity burn (Figure 4). One year after the burn, changes in percent cover were significantly lower in the burn plots for beargrass ($p = 0.01$) and shrubs ($p = 0.03$) than in reference plots (Figure 5). However, there was no significant difference between the burn and reference plots for changes in beargrass shoots, flowering rates, forb cover, or graminoid cover after one year. Two years after the burn, changes in percent cover of beargrass and shrubs were significantly lower in the burned plots than in the reference plots, but there was a significant increase in forb percent cover ($p = 0.01$) and beargrass shoot number ($p = 0.01$) in the burn plots compared to the reference plots (Figure 6a). The change in beargrass shoots was on average an increase of 28 individuals in burn plots, compared to a decrease of approximately 4 individuals in the reference plots. In the summer of 2005, there was an average of 85 beargrass shoots in the burned plots and an average of 35 beargrass shoots in the unburned plots. Beargrass cover was reduced by approximately 7% in burn plots compared to a slight increase of 1% in reference plots (Figure 6b).

Burned plots had significantly higher seedling establishment than the reference plots both years after the high-severity fire (Table 2, $p < 0.05$). The mean beargrass germination rate in the burn plots after one year was 9.2%, compared to 0.7% in the reference plots. Many of the beargrass seedlings continued to survive in the second year, yet the mean number of beargrass plants in the reference plots decreased by nearly half. No beargrass seedlings established in plots not sown with beargrass seed.

Experiments 3 and 4—Stimulation of beargrass vegetative reproduction and seedling establishment by low-severity fire. In Skokomish, the mean number of beargrass shoots increased significantly more ($p = 0.01$) in the manually cleared plots (39.17) than in the burned (7.17) or reference plots (9.17) (Figure 7a). The percent cover of beargrass was significantly lower in cleared ($p = 0.02$) and burned ($p = 0.01$) plots when compared to reference plots. Shrub cover decreased in the burned, cleared, and reference plots, by 34%, 26%, and 5%, respectively, with significant differences between the reference and burn plots ($p < 0.001$) and the reference and cleared plots ($p = 0.01$). There was no significant difference in the percent cover of all forbs between the burn, cleared, and reference plots ($p = 0.08$). In low-severity burn plots, percent cover of other herbaceous plants doubled in the second year postfire to a mean of over 50% and was more than three times that of the reference plots.

A total of 90 beargrass individuals were measured within the three treatments (burned, cleared, reference) in 2004; all of them persisted to the 2005 field season, regardless of treatment. Beargrass exposed to low-severity burns had significantly shorter leaves ($p = 0.03$) and lower stature ($p = 0.02$) (Figure 7b). There was slight evidence of a difference in crown area between the three treatments ($p = 0.06$). There were no significant differences between burned and cleared plots for changes in leaf lengths, height, and crown area. The flowering rate was significantly lower

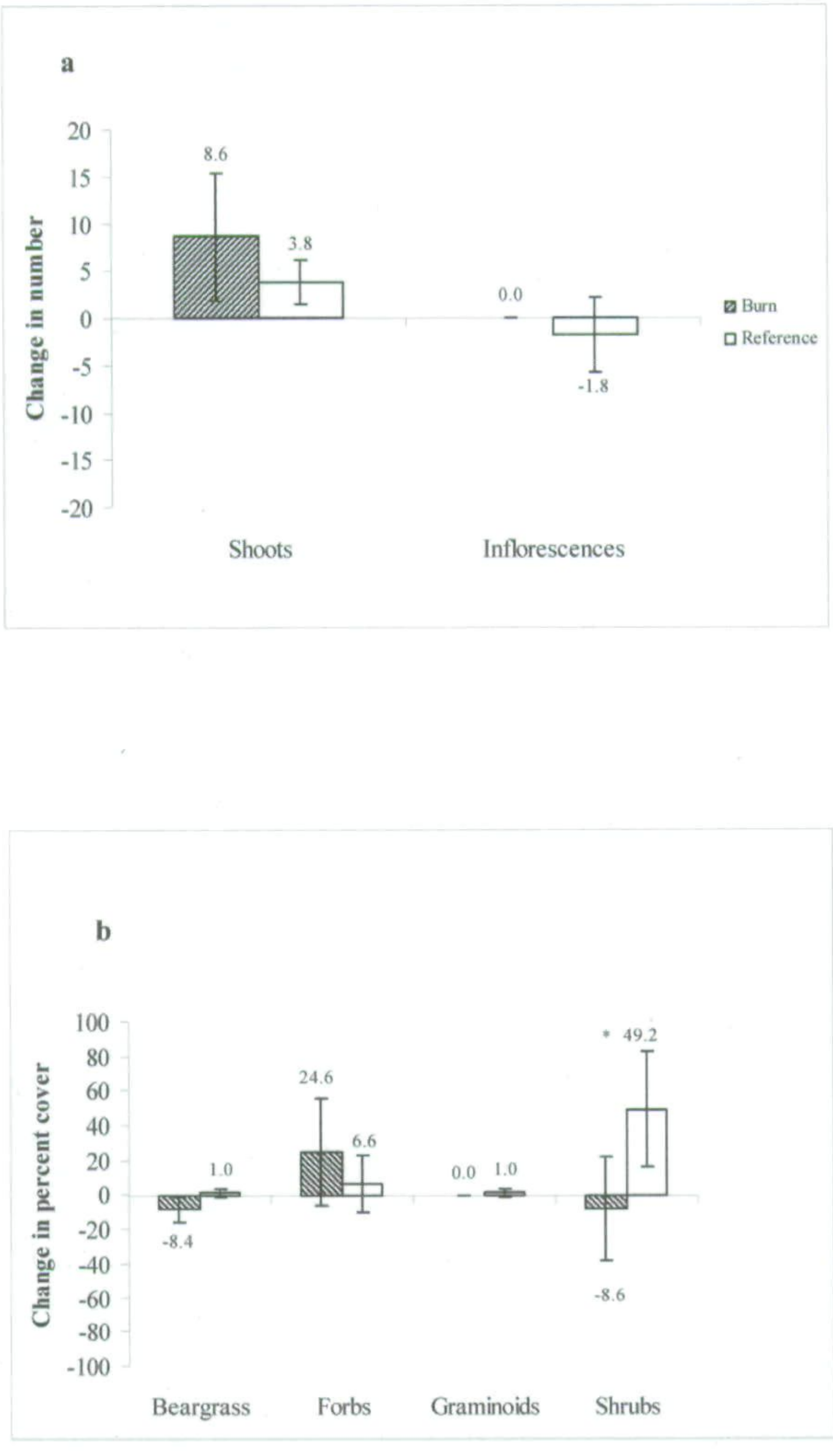


Figure 5. Changes in vegetation at the Skokomish site one year after initiation of study relative to initial observations in burned and reference plots: a) mean (± SD) number of shoots and inflorescences of beargrass (*Xerophyllum tenax*); b) mean (± SD) percent cover of beargrass, forbs, graminoids, and shrubs. An asterisk (*) indicates significant difference at $\alpha = 0.05$ using Mann-Whitney *U* test.

in burned plots than in reference plots ($p = 0.05$), but the number of beargrass flowers was not significantly different between cleared and reference plots ($p = 0.80$) (Figure 7c).

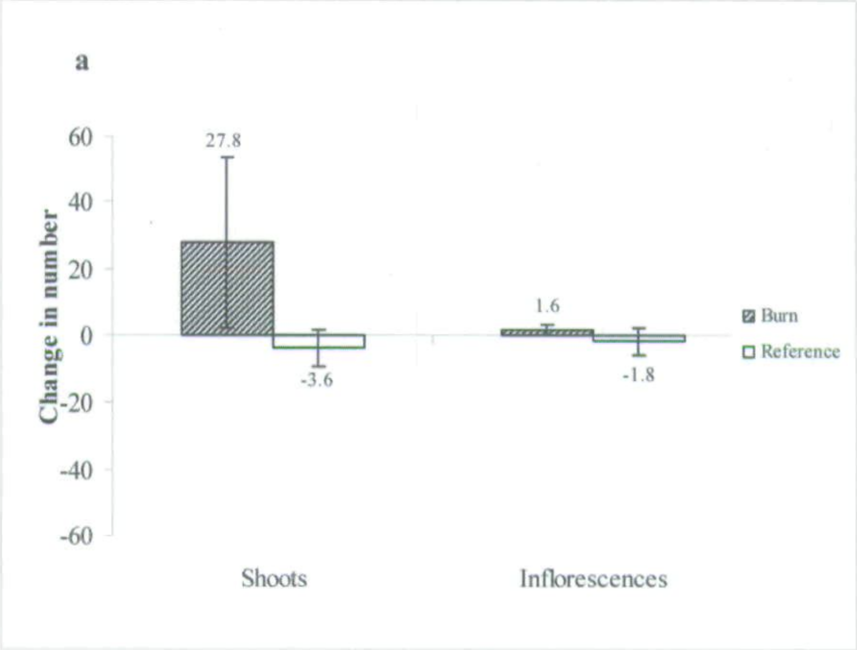
At the Quinault site, there was no significant difference between the treatments for changes in number of beargrass shoots or percent cover. In addition, there was no significant difference between treatments for changes in the percent cover of forbs, graminoids, or shrubs. The number of beargrass inflorescences, however, was significantly different between treatments ($p = 0.04$), higher in reference plots than burn plots ($p = 0.04$), and significantly higher in reference plots than cleared plots ($p = 0.05$). Prior to treatment installation, there was no significant difference between plots for flowering rate ($p = 0.40$). Although seeds were not broadcast in plots for Experiment 3, there were 13 seedlings in one of the burn plots at Quinault, and one seedling in a cleared plot one year after the treatments.

There was high beargrass mortality throughout Quinault. Of the 90 beargrass individuals measured in 2004, 57 (63%) were no longer alive during the 2005 field season. There was no significant difference in beargrass mortality between treatments ($p = 0.20$). Only live beargrass at Quinault was used in the analysis of individual plant measurements, and no significant difference was found between the treatments for changes in beargrass leaf length, height, or crown area. On average, 41 beargrass shoots were on hummocks in each plot compared to 13 not on hummocks ($p = 0.03$).

Neither the Skokomish nor Quinault treatments had a significantly different response of seedling establishment rates between the treatments. The litter layer remained intact after the low-severity burns.

Discussion

The research presented in Experiments 1 and 2 serves as a case study to investigate the effects of a high-severity fire on beargrass reproduction. The scope of inference was limited by the absence of replicate sites and the fact that the response data are limited to one or two years (for Experiments 1–2 and 3–4, respectively) following the treatments. In addition, the low number of plots for the experiments may explain why some observed differences were not statistically significant. While larger



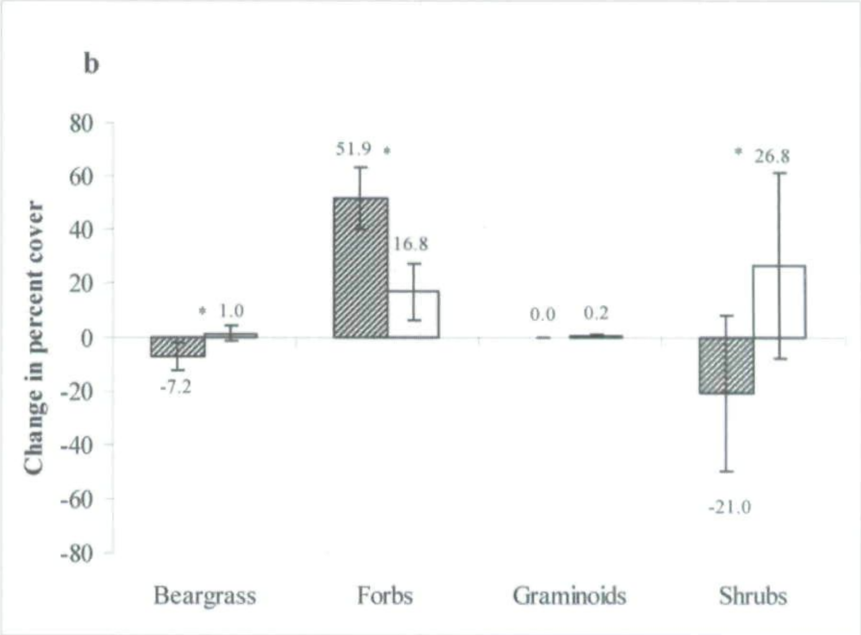


Figure 6. Changes in beargrass (*Xerophyllum tenax*) parameters and vegetative cover at the Skokomish site two years after initiation of Experiment 1 relative to initial observations in burned and reference plots: a) mean (± SD) number of shoots and inflorescences; and b) mean (± SD) percent cover of beargrass, forbs, graminoids, and shrubs. An asterisk (*) indicates significant difference at $\alpha = 0.05$ using Mann-Whitney *U* test.

sample sizes are always preferable to increase confidence in the results, in many cases, because of distribution of the communities studied, species rareness, or the need to take advantage of an opportunity such as a planned burn, smaller samples must be taken. In addition, it is important to note that conservation studies, by their nature, often have small sample sizes (Thomas 1997). Despite these limitations, the findings suggest trends that provide some understanding of the short-term consequences of reintroducing burning and limiting competition in low-elevation beargrass habitat.

As expected, burning reduced cover of beargrass for both the high- and low-severity fires. This decrease in percent cover resulted from the burning of leaves and damage sustained by some meristems, resulting in a lack of the cumulative annual growth (ca. 1%) that occurred in reference plots. The increase in shoot density two years after the high-severity burn supports findings from previous research conducted on high-elevation beargrass on Mount Rainier (Maule 1959). This study emphasized changes in quantity of beargrass resulting from the treatments. Basketmaking quality can be inferred through leaf length and pliability, but since the leaves will take a few years to recover from the burns, those that were in reference plots were inevitably longer than our postburn measurements.

Beargrass plants in low-severity burn plots had significantly lower leaf length, height, percent cover, and crown area than those in reference plots. Beargrass in cleared plots, however, was not significantly different from that in the reference plots. We believe that this finding suggests that the postfire recovery period for beargrass was longer than the recovery period after manual leaf cutting. The length of recovery for burned plants is dictated by the meristem damage; if severe, a plant may not recover. We suspect that beargrass leaf length, height, percent cover, and crown area will continue to be lower in the low-severity burn plots than in reference plots two years following the fire, based on the finding that two years after the high-severity fire, the percent cover of beargrass was still significantly lower than in reference plots.

Findings from Experiment 3 emphasize the importance of competition in beargrass vegetative reproduction. Research on high-elevation beargrass in Idaho found that the species can be very sensitive to competition from shrubs following disturbance (Laursen 1984). It is possible that at Skokomish, the decrease in shrubs as a result of fire was more influential than the

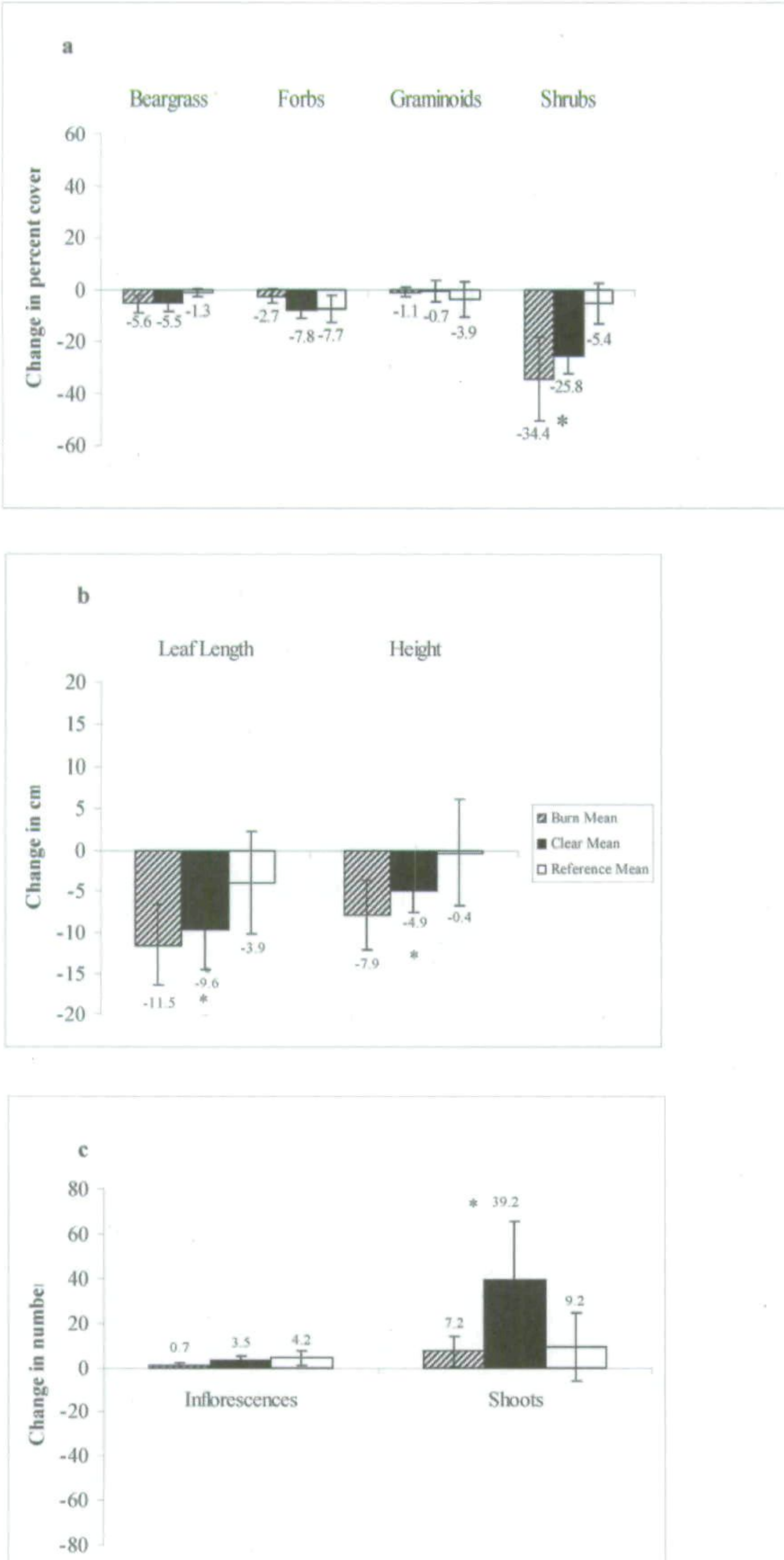


Figure 7. Changes in beargrass (*Xerophyllum tenax*) parameters and vegetative cover at the Skokomish site one year after initiation of Experiment 3 relative to initial observations in burned, cleared, and reference plots: a) mean (± SD) number of inflorescences and shoots; b) mean (± SD) percent cover of beargrass, forbs, graminoids, and shrubs; and c) mean (± SD) longest-leaf length and plant height for five individuals per plot. An asterisk (*) indicates significant difference at $\alpha = 0.05$ using ANOVA.

fire itself in the increase of beargrass. In Experiment 3, only manually cleared plots had a significant increase in beargrass shoots compared to the reference plots. A significant difference in beargrass vegetative reproduction may occur between the burn and reference plots after an additional year or two, yet much of the success of beargrass reproduction will depend on competition from shrubs.

Despite previous observations of high-elevation beargrass flowering within the first year following fire (Kruckeberg 2003), we found no significant effect of a high-severity burn on flowering one or two years after burning and a decline in flowering after a low-severity burn. Maule's (1950) emphasis on soil temperature in controlling flowering rates leads us to hypothesize that the relative increase in soil temperature is greater after a fire in subalpine prairies than in the warmer low elevations, and therefore has a greater effect on beargrass flowering. An alternative explanation for the low flowering rates of burned and cleared beargrass might involve the allocation of energy to growth, rather than to flowering, following the disturbance. Based on Experiment 1, however, it is possible that a significant difference between the burn and reference plots would occur two or three years after treatment.

Our research suggests that high-severity, but not low-severity, fires are effective at preparing safe sites for beargrass seedling establishment. Because of their small size (4 mm) and resulting small seedlings, beargrass seeds benefit from contact with mineral soil for germination and survival. No significant difference was found for beargrass seedling establishment between cleared or reference plots and the low-severity fire treatments, all of which left the litter layer intact. In a study of savanna shrub species, Miyanishi and Kellman (1986) also found that seed germination and seedling establishment were limited by the availability of safe sites with mineral soil exposed and litter removed through burning.

Plots that were not sown intentionally with beargrass seed had no seedlings either one or two years after the high-severity fire. The lack of seedlings in both the reference and burned plots of Experiment 2 indicate that the beargrass seed source was not naturally high enough for postfire recruitment at Skokomish. Since the plots were established in a section of the burn unit that did not have beargrass, the absence of seeds was not surprising. Determining whether germination from seed is an important component of fitness for beargrass is beyond the scope of this study.

The high beargrass mortality at Quinault following installation of Experiments 3 and 4 was likely due to the clearcut five years prior to the experiment and the resultant increase in competition with shrubs and other forbs. As found by other studies, beargrass in some clearcut sites decreases in percent cover and takes over 20 years to recover, primarily owing to competition with woody vegetation, if recovery occurs at all (Halpern and Spies 1995, Nelson and Halpern 2005). It is important to note that beargrass mortality was not significantly different between treatments and was high throughout the site. We believe that the most important factor influencing beargrass survival was its microtopographical position within the site. Microsites that are generally seasonally flooded and permanently saturated (Kulzer et al. 2001) characterize the bog. In 2005, these lower microsites were flooded throughout the year, including during the summer, despite the fact that 2004–2005 received less precipitation than average (WRCC 2005). Hummocks up to 1 m long occur throughout the site; they generally consist of well-drained soils, and are therefore typically drier. The higher number of beargrass shoots on hummocks compared with the number observed between hummocks supports the hypothesis that beargrass survival at this site was linked mostly to its occurrence in dry microsites in an otherwise flooded landscape. Perhaps as this landscape continues to transition to an open bog ecosystem, beargrass distribution will be further limited.

An interesting paradox is that, while historically burns set by indigenous people in the Pacific Northwest were likely low-severity surface fires, in this study it was the high-severity fire that resulted in an increase in beargrass vegetative reproduction and seedling establishment. Perhaps this finding is due to environmental changes, such as the accumulation of biomass and altered hydrology, that have occurred in the absence of fire. Over the past century, fire suppression has led to fuel increases and shifts in species distributions (Lepofsky et al. 2003, Ruppert 2003, Wrobleski and Kauffman 2003). We believe that if this study were to continue repetitive burns over the long term, our findings would reflect our original hypothesis that low-severity fire would yield the greatest increase in beargrass abundance.

Conclusion

We conclude that fire may maintain low-elevation beargrass populations. As a result of a high-severity fire, we observed increases in beargrass seed germination, seedling establishment, and shoot production. Manual clearing areas of vegetation and coarse woody debris may result in an increase in beargrass shoot number after only one year, but low-severity fires may not have a similar influence in the same length of time. One year after the low-severity fires, there was a trend toward increased biomass production, which may continue. Low-severity fires did not have the same effect on beargrass seedling establishment as high-severity fires, although we cannot rule out the role that site differences may have played in our experiments.

While this research suggests that fire can increase beargrass vegetative reproduction, the fires must be frequent enough to limit shrub and tree encroachment. To ensure that effects on beargrass reproduction and growth are long lasting, a fire-return interval of less than 20 years may be necessary for a savanna structure (Peter and Shebitz 2006). This frequency would likely limit competition and ensure enough time for long, basketmaking-quality leaves to develop.

To manage existing low-elevation beargrass, clearing the surrounding vegetation manually may be an effective short-term method to increase the number of shoots. However, this technique does not facilitate the establishment of new seedlings, which would be necessary to sustain populations over the long term. This study also involved manually clipping beargrass leaves as a form of management. Perhaps further research might investigate if the act of harvesting the leaves for basketmaking stimulates new shoots.

Over a large area of land, fire was beneficial to both existing beargrass and in preparing a site for new seedlings. If there is interest among land managers or tribal members to plant beargrass seeds, this study does show that increased seed germination will occur after a fire that exposes the mineral soil.

This research illustrates the importance of incorporating traditional land management techniques such as burning practices in ecosystem restoration. While methods focused on beargrass, this research serves as a case study for examining implications of reintroducing burning to a historically maintained vegetative community and resource. In addition to the species-specific effects of increasing beargrass vegetative reproduction and seeding establishment rates, restoring fire to anthropogenically maintained landscapes on the Olympic Peninsula has significant ecological and cultural implications. Reintroducing anthropogenic burning to maintain savannas and wetlands reestablishes an ecosystem component that is now absent from this area. Restoring low-elevation savannas and prairies through regular burning may support flora and fauna communities that have depended on these habitats for millennia.

Incorporating Native American land management into conservation and restoration efforts has great potential to provide an understanding of the past structure of an area, to restore the native biodiversity that characterized that system, and to strengthen cultural traditions dependent upon the land.

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References

Agee, J. K. 1993. *Fire Ecology of Pacific Northwest Forests*. Washington DC: Island Press.

Anderson, M.K. 1996a. Tending the wilderness. *Restoration & Management Notes* 14:154–165.

___. 1996b. The ethnobotany of deergrass, *Muhlenbergia rigens* (Poaceae): Its uses and fire management by California Indian Tribes. *Economic Botany* 50:409–422.

___. 2005. *Tending the Wild: Native American Knowledge and the Management of California's Natural Resources*. Berkeley: University of California Press.

Boyd, R., ed. 1999. *Indians, Fire, and the Land in the Pacific Northwest*. Corvallis: Oregon State University Press.

Bradley, A.F. 1984. Rhizome morphology, soil distribution, and the potential fire survival of eight woody understory species in western Montana. MA thesis, Colorado College.

Clark, D.L. and M.V. Wilson. 2001. Fire, mowing, and hand-removal of woody species in restoring a native wetland prairie in the Willamette Valley of Oregon. *Wetlands* 21:135–144.

Cooke, S.S. 1997. *A Field Guide to the Common Wetland Plants of Western Washington and Northwestern Oregon*. Seattle: Seattle Audubon Society.

Crane, M.F. 1990. *Xerophyllum tenax*. Fire Effects Information System. USDA Forest Service Rocky Mountain Research Station, Fire Sciences Laboratory. www.fs.fed.us/database/feis/plants/forb/xerten/all.html

Gunther, E. 1981. *Ethnobotany of Western Washington: The Knowledge and Use of Indigenous Plants by Native Americans*. Seattle: University of Washington Press.

Halpern, C.B. and T.A. Spies. 1995. Plant species diversity in natural and managed forest of the Pacific Northwest. *Ecological Applications* 5:913–934.

Henderson, J.A., D.H. Peter, R.D. Lesher and D.C. Shaw. 1989. Forested plant associations of the Olympic National Forest. USDA Forest Service Technical Paper R6-ECOL-TP-001-88.

Higgins, S., K. Blatner, B.K. Kerns and A. Worthington. 2004. Relationship between *Xerophyllum tenax* and canopy density in the southern Cascades of Washington. *Western Journal of Applied Forestry* 19:82–87.

Hitchcock, C.L. and A. Cronquist. 1973. *Flora of the Pacific Northwest: An Illustrated Manual*. Seattle: University of Washington Press.

Hunter, J.E. 1988. Prescribed burning for cultural resources. *Fire Management Notes* 49(2):8–9.

James, J.E. and L.A. Chubby. 2002. Quinault. Pages 99–116 *in* J. Wray (ed), *Native Peoples of the Olympic Peninsula: Who We Are*. Norman: University of Oklahoma Press.

Jones, G.N. 1936. *A Botanical Survey of the Olympic Peninsula, Washington*. Vol. 5 of Publications in Biology. Seattle: University of Washington.

Keeley, J.E. 1981. Reproductive cycles and fire regimes. Pages 231–277 *in* H.A. Mooney, T. Bonnicksen, N.L. Christensen, J. Lotan and W.Reiners (eds), Proceedings of the symposium on fire regimes and ecosystems properties. USDA Forest Service General Technical Report WO-26.

Kruckeberg, A. 2003. *Gardening with Native Plants of the Pacific Northwest*. Seattle: University of Washington Press.

Kulzer, L., S. Luchessa, S. Cooke, R. Errington and F. Weinmann. 2001. Characteristics of the low-elevation *Sphagnum*-dominated peatlands of western Washington: A community profile. Seattle: U.S. Environmental Protection Agency.

Kunze, L.M. 1994. Preliminary classification of native, low elevation, freshwater wetland vegetation in Western Washington. Olympia: Washington Department of Natural Resources.

LaLande, J. and R. Pullen. 1999. Burning for a 'fine and beautiful open country': Native uses of fire in southwestern Oregon. Pages 185–218 *in* R. Boyd (ed), *Indians, Fire and the Land in the Pacific Northwest*. Corvallis: Oregon State University Press.

Laursen, S.B. 1984. Predicting shrub community composition and structure following management disturbance in forest ecosystems of the Intermountain West. PhD dissertation, University of Idaho.

Lepofsky, D., E.K. Heyerdahl, K. Lertzman, D. Schaepe and B. Mierendorf. 2003. Historical meadow dynamics in southwest British Columbia: A multidisciplinary analysis. *Conservation Ecology* 7(3):5. www.consecol.org/vol7/iss3/art5

Lewis, H.T. 1993. Patterns of Indian burning in California: Ecology and ethnohistory. Pages 55–116 *in* T.C. Blackburn and K. Anderson (eds), *Before the Wilderness: Environmental Management by Native Californians*. Menlo Park CA: Ballena Press.

Lewis, H.T. and T.A. Ferguson. 1999. Yards, corridors, and mosaics: How to burn a boreal forest. Pages 164–184 *in* R. Boyd (ed), *Indians, Fire and the Land in the Pacific Northwest*. Corvallis: Oregon State University Press.

Maule, S.M. 1959. *Xerophyllum tenax*, squawgrass, its geographic distribution and its behaviour on Mount Rainier, Washington. *Madroño* 15:39–48.

Miyanishi, K. and M. Kellman 1986. The role of fire in recruitment of two neotropical savanna shrubs, *Miconia albicans* and *Clidemia sericea*. *Biotropica* 18: 224–230.

Munger, S.H. 2003. *Common to This Country: Botanical Discoveries of Lewis and Clark*. New York: Artisan Books.

Nelson, C.R. and C.B. Halpern. 2005. Edge-related responses of understory plants to aggregated retention harvest in the Pacific Northwest. *Ecological Applications* 15:196–209.

Nordquist, D.L. and G.E. Nordquist. 1983. *Twana Twined Basketry*. Ramona CA: Acoma Books.

Norton, H.H. 1979. The association between anthropogenic prairies and important food plants in western Washington. *Northwest Anthropological Research Notes* 13:175–200.

Pendergrass, K.L., P.M. Miller, J.B. Kauffman and T.N. Kaye. 1999. The role of prescribed burning in maintenance of an endangered plant species, *Lomatium bradshawii*. *Ecological Applications* 9: 1420–1429.

Peter, D. and D.J. Shebitz. 2006. Historic anthropogenically maintained beargrass savannas of the southeastern Olympic Peninsula. *Restoration Ecology* 14:605–615.

Regan, A. 1934. Plants used by the Hoh and Quileute Indians. *Transactions of the Kansas Academy of Science* 37:5–93.

Rentz, E. 2003. Effects of fire on plant anatomical structure in native Californian basketry materials. MS thesis, San Francisco State University.

Rudall, P.J., K.L. Stobart, W.P. Hong, J.G. Conran, C.A. Furness et al. 2000. Consider the lilies: Systematics of Liliales. Pages 347–357 *in* K.L. Wilson and D.A. Morrison (eds), *Monocots: Systematics and Evolution*. Collingwood, Australia: CSIRO Publishing.

Ruppert, D. 2003. Building partnerships between American Indian tribes and the National Park Service. *Ecological Restoration* 21:261–263.

Shebitz, D.J. 2005. Weaving traditional ecological knowledge into the restoration of basketry plants. *Journal of Ecological Anthropology* 9:51–68.

___. 2006. The historical role and current restoration applications of fire in maintaining beargrass (*Xerophyllum tenax* (Pursh) Nutt.) habitat on the Olympic Peninsula, Washington State. PhD dissertation, University of Washington, Seattle.

Shebitz, D.J. and R.W. Kimmerer. 2004. Population trends and ecological requirements of sweetgrass, *Hierochloe odorata* (L.) Beauv. *Journal of Ethnobiology* 24:93–111.

___. 2005. Re-establishing roots of a Mohawk community and a culturally significant plant: Sweetgrass. *Restoration Ecology* 13:257–264.

Shebitz, D.J., S.H. Reichard and W. Woubneh. 2008. Beargrass (*Xerophyllum tenax*) on the Olympic Peninsula, Washington: Autecology and population status. *Northwest Science* 82:128–140.

Stewart, O.C. 2002. The effects of burning of grasslands and forests by aborigines the world over. Pages 65–312 *in* H.T. Lewis and M.K. Anderson (eds), *Forgotten Fires: Native Americans and the Transient Wilderness*. Norman: University of Oklahoma Press.

Thomas, L. 1997. Retrospective power analysis. *Conservation Biology* 11:276–280.

USDA NRCS. 2005. The PLANTS Database (Accessed October 29). Baton Rouge LA: National Plant Data Center. plants.usda.gov

Vance, N.C., P. Bernhardt and R.M. Edens. 2004. Pollination and seed production in *Xerophyllum tenax* (Melanthiaceae) in the Cascade Range of central Oregon. *American Journal of Botany* 91:2060–2068.

Weather Underground. 2009. History for Shelton, WA: Thursday, September 23, 2004. www.wunderground.com/cgi-bin/findweather/getForecast?query=98584

Western Regional Climate Center (WRCC). 2009. Black Knob Washington. Western Regional Climate Center. www.wrcc.dri.edu/cgi-bin/rawMAIN.pl?waWBLA

Whelan, R.J. 1995. *The Ecology of Fire*. New York: Cambridge University Press.

Wrobleski, D.W. and J.B. Kauffman. 2003. Initial effects of prescribed fire on morphology, abundance, and phenology of forbs in big sagebrush communities in southeastern Oregon. *Restoration Ecology* 11:82–90.

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