Three-Year Effects of Crown Removal by Clipping or Burning on Beargrass (Xerophyllum tenax) Size and Biomass

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Abstract

Beargrass (*Xerophyllum tenax*), an evergreen perennial herb of the northern Rocky Mountains, Pacific Northwest, and northern California, is used in Native American basketry and commercial floral greens. We studied beargrass size and biomass responses to crown removal by clipping or burning over three years in a coast Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) woodland with variable shrub cover in the southeastern Olympic Peninsula of Washington State. Clipping forested plants resulted in 28% mortality, mostly from smaller plants growing under 26% more total cover than the surviving plants; however, only 3% of completely crown-scorched open-grown plants died. Three years after treatment, crown width of surviving plants was only 61% of the pre-treatment size for clipped plants, compared to 88% for completely crown-scorched plants. Regression analyses indicated that the percentage of crown scorch accounted for only 16% and 27% of crown width and foliar height variation, respectively, one year post-burn, decreasing to 10% and 19% at three years post-burn. During the three years after burning, percentage flowering increased linearly to 64% of plants. Three years post-burn, foliar browse was higher on crown-scorched than on non-crown-scorched plants. Although shade tolerant, long-term survival of lowland beargrass is likely limited by combined competition from shrubs and trees. Stand density management is needed to maintain healthy, reproducing populations in the lowlands of western Washington.

Keywords: beargrass ecology, fire effects, shade tolerance, browse, flowering

Introduction

Beargrass (*Xerophyllum tenax* (Pursh) Nutt.) is an evergreen herb that grows primarily in summer-dry montane to subalpine meadows and forests of generally low productivity in the mountains of the Pacific Northwest, northern California, and the northern Rocky Mountains of western North America. It also grows in forests and bogs in the coastal lowlands of Washington State. Beargrass is not a true grass, but rather it is a member of the Melanthiaceae family within the order Liliales.

Beargrass forms perennial tussocks atop short, branching, 1- to 2-cm thick semi-woody rhizome offsets. Each offset produces a whorl of grass-like leaves up to 100 cm long and 2 to 6 mm wide (Vance et al. 2001, Hitchcock and Cronquist 2018). Collectively the leaf whorls from all the offsets produce the characteristic tussock. While beargrass continuously produces short new rhizomes, the plant maintains an overall single tussock appearance.

Beargrass leaves have xerophytic features, including a thickened cuticle, stomata only on the lower leaf surface, and a thick-walled epidermis (Rentz 2003). Each inflorescence arises from a separate leafy offset and is up to 1.5 m tall, including a flowering stalk with a terminal raceme of many small white flowers (Hitchcock et al. 1969). The flowering offset subsequently dies, but new offsets produced before flowering ensure long-lived persistence (Kruckeberg 1982, Hitchcock and Cronquist 2018). Flowering is mostly observed in open areas and is infrequent or absent under forest canopies (Maule 1959, Crane 1990, Vance et al. 2004, Peter et al. 2017, Antos et al. 2021). The primary roots are cordlike, up to 4 m long, with shorter, finer roots branching from these (Antos et al. 2021).

Beargrass invests heavily in tough, evergreen foliage, rhizomes, and an extensive root system that can occupy 50 times more area than the area of the crown (Antos et al. 2021). This helps in cop-

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ing with harsh, low-productivity habitats (Grime and Hunt 1975, Atkin et al. 1996, Atkinson et al. 2014), including droughty, infertile, and shady sites. Smith and Huston (1989) hypothesized that terrestrial plants cannot simultaneously use low levels of water and light efficiently, and yet this is typical of beargrass habitat. Belowground resources are often more limiting to plant survival and growth than light on shaded dry or infertile soils (Coomes and Grubb 2000). In this situation, shade-tolerant plants including beargrass often have thick, tough, long-lived leaves and a high allocation to roots, thereby achieving drought tolerance, but at the expense of rapid growth (Coomes and Grubb 2000).

Interest in beargrass management is increasing due to its cultural and commercial values. The tough, evergreen foliage of beargrass is useful in Native American basketry and in the floral trades industry. While native cultural uses date to prehistoric times, modern commercial harvest of beargrass began in the 1980s (Higgins et al. 2004, Lynch and McLain 2003) and today is a multi-million dollar industry (Schlosser and Blatner 1997, Charnley and Hummel 2011), yet little is known of the growing conditions under which harvest is sustainable. Our study contributes toward an understanding of sustainable harvesting but is not a "road map" to its achievement. Rather we provide a basic understanding of the effects of crown removal by clipping or burning on plant health under various levels of competition that will inform managers and guide future research.

Traditional and commercial beargrass harvesters require different foliar qualities satisfied by different growing conditions. Flat, long, straight, pliable leaves, low in pigment, but fibrous enough to withstand weaving are needed for basketry (Rentz 2003, Shebitz et al. 2009). Such foliage occurs in the first years after burning of partially shaded plants (Hunter 1988, Rentz 2003, Anderson 2005). It is thinner, less sclerified, and has more mesophyll than leaves from unburned plants, resulting in greater flexibility and elasticity (Rentz 2003). Rentz (2003) suggested that such leaves are grown from stored reserves to rapidly recoup lost photosynthetic capacity. In contrast, the commercial floral industry demands long, wide, firm, deep-green leaves (Schlosser and Blatner 1997), which come from large plants growing in shade (Thomas and Schumann 1993, Schlosser and Blatner 1997). Higgins et al. (2004) found that a moderately dense overstory (60% to 90% canopy cover) produced the best commercial-quality beargrass, but recent burns and clearcuts were not suitable.

Western Washington Native peoples burned landscapes to maintain culturally important plants (Rentz 2003, Wray and Anderson 2003). On the southeastern Olympic Peninsula, the Skokomish people burned beargrass savannas at 2- to 3-year intervals (Peter and Shebitz 2006) with low-severity surface fires (Rentz 2003, Shebitz et al. 2009). After a century of fire suppression, these savannas and woodlands have succeeded to coast Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*) and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) forests, resulting in declining beargrass populations (Jones 1936, Peter and Shebitz 2006).

Typical of rhizomatous species in which new rhizome production results in sustained renewal, beargrass is long lived (Harper 1977, Laursen 1984, Crane 1990, Vance et al 2004). However, little actual age data exist. Peter et al. (2017) estimated plant ages of 60 years or more based on forest stand history. Antos et al. (2021) documented mature plants living for 40 years and suggested beargrass may live for hundreds of years based on old-growth stand history.

Although mature beargrass is shade tolerant, flowering becomes infrequent in shade (Maule 1959, Crane 1990, Vance et al. 2004). During a 40-year study in old-growth forest, only occasional flowering stalks were observed (Zobel and Antos 2016, Antos et al. 2021). Peter et al. (2017) found no flowering or seedlings in areas with < 30% full sunlight. Also, shaded beargrass plants had lower crown density, smaller basal diameter, and fewer vegetative offshoots compared to plants in open areas (Peter et al. 2017). They suggested that plants in shaded areas, although of somewhat higher commercial quality, were likely to recover more slowly from foliar harvest, and

were less likely to be replaced after mortality suggesting that sustainable populations require at least partly open canopies (Peter et al. 2017).

Here we report on two new studies examining beargrass recovery from foliar removal conducted at the same site reported in Peter et al. (2017). We report on the effect of clipping foliage from plants growing under conifer shade with heavy salal (*Gaultheria shallon* Pursh) competition, thus testing the 2017 study's suggestion that foliar harvesting is harmful to shade-grown plants. We also examined how varying levels of competition affected the outcome of foliar clipping and whether some competing species or species groups were more important than others. In the second study, we examined the effects of crown scorch from prescribed burning on open-grown plants in terms of mortality, plant size, biomass, flowering, and browsing. We then compared mortality and plant size for plants that had sustained 100% crown loss from burning versus crown loss from clipping three years after treatments.

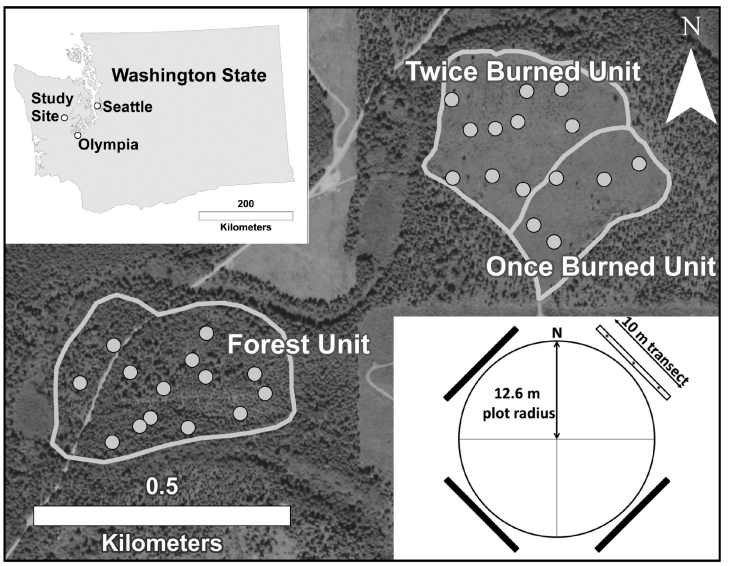

Figure 1. The three study units and associated plot locations (circles) for the beargrass crown recovery studies. The twice-burned and once-burned units were burned in 2003. The twice-burned unit was again burned in 2015 for the burn recovery study. The clipping recovery study utilized the forest and once-burned units. Also shown are the plot and transect configurations (lower right). Dots in the northeast transect indicate locations where photosynthetically active radiation (PAR) was measured in each transect at the center of 3.3-m transect segments.

Methods

Beargrass seedlings begin to produce short rhizomes after several years, and all our study plants had several to many rhizomes. The foliar tussock formed from all the rhizomes represents a genet due to its clonal origin, but in this paper we refer to it as a "plant," and the short ramets of which it is composed as "offsets."

Study Area

The studies took place on the Olympic Peninsula of Washington State, near the South Fork of the Skokomish River in the Olympic National Forest (Figure 1). Historically, this area had open woodland or savanna structure but became forested by the latter part of the 20th century (Peter and Shebitz 2006, Peter et al. 2017). Beargrass was an important plant in the historically open community and is still present today.

The clipping recovery study utilized an 11-ha formerly open woodland that is now a fully stocked Douglas-fir forest with a well-developed understory of salal (Figure 1, forest unit). In addition, this study used a nearby 5.1-ha area that was heavily thinned in 2001 and prescribed burned in 2003, but was previously otherwise like the forest unit (once-burned unit). The burn recovery study utilized a 13-ha area (Figure 1, twice burned unit) that was heavily thinned in 2001 and prescribed burned in 2003, but was previously otherwise like the forest unit. This unit was burned again in 2015 for this study. Before the 2015 prescribed burn, both burn units were largely open with a patchy shrub layer dominated by salal, a few scattered large Douglas-fir trees, and scattered understory conifers beginning to occupy the site.

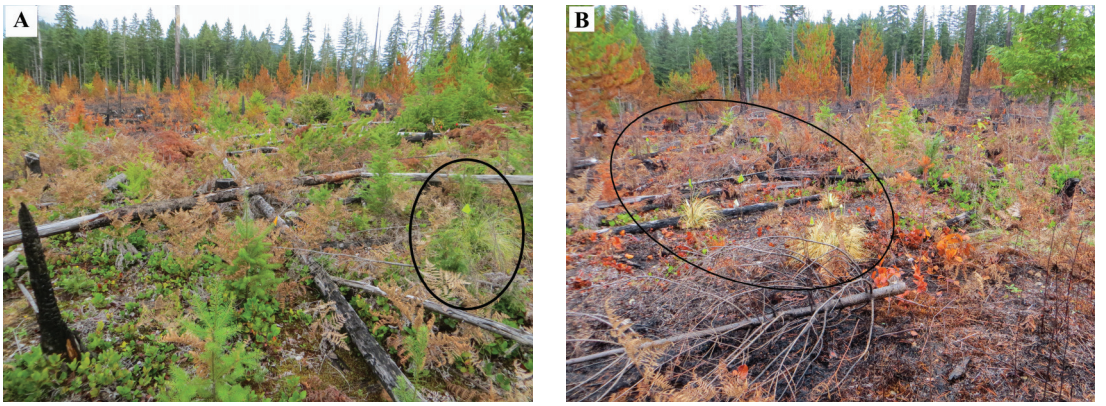

Figure 2. Two views showing the vegetation mosaic that resulted from the 2015 prescribed burn in the twice-burned unit. A: View across a plot with only 33% scorched vegetation. Char on the log and snag in the foreground resulted from the 2003 burn, and the browning of bracken fern was due to seasonal senescence. The yellow-flagged beargrass plant (circled) is a non-crown-scorched sample plant. B: View across a plot with 70% scorched vegetation. Note the yellow-flagged and crown-scorched beargrass plants (circled).

The soils at the study site are droughty, relatively infertile (Henderson et al. 1989, Peter and Shebitz 2006), and derived from glacial till and outwash (Thorson 1980, Soil Survey Staff USDA NRCS 2011). Flatter parts of the forest unit have the Duskpoint soil series, which is a very gravelly loam (sandy-skeletal, isotic, mesic Typic Hapludand). The once- and twice-burned units and the sloped parts of the forest unit have the somewhat excessively drained Bogachiel-Ishmael Complex soil series (Ishmael—medial-skeletal, ferrihydritic, or Bogachiel—isotic mesic Typic Udivitrands).

Annual precipitation is 2,260 mm, and the mean annual temperature is 10.5 °C (PRISM Climate Group 2019). The predominant plant community was classified as the western hemlock/salal/beargrass (*Tsuga heterophylla /Gaultheria shallon/ Xerophyllum tenax*) association (Henderson et al. 1989). In this association, salal is typically the dominant understory species (Henderson et al. 1989), and at our study site, its canopy coverage averaged 85% in the forest unit, 65% in the once-burned unit, and 53% in the twice-burned unit.

In 2002 to 2003, the once- and twice-burned units (Figure 1) were thinned to 15 mature trees ha^{-1} and then prescribed burned on 30 September 2003. A hotter-than-expected burn resulted in a residual density of 1 to 2 trees ha^{-1}. A second prescribed burn designed to control tree regeneration and shrub competition, and encourage beargrass growth and reproduction, was carried out on the 8.3-ha twice-burned unit on 23 September 2015. This burn was patchy with a few hot spots and some unburned areas (Figure 2). The remaining 5.1 ha were not burned, and we refer to this area as the once-burned unit (Figure 1).

Sample Plots

Both studies utilized a network of circular plots established for a previous study (Peter et al. 2017). Most of the plots were 0.05 ha in size and were located within each unit based on randomly assigned global positioning system coordinates. Two of the forest plots were 0.086 ha in area, and one forest and two twice-burned unit plots were 0.04 ha in area, originally installed by the USDA Forest Service Area Ecology Program for other purposes. As the experimental units are individual plants, the actual plot sizes are not relevant to this study. The plots were used to disperse the sample plants over the sample area while maintaining relevance of comparisons with the previous study (Peter et al. 2017).

The clipping recovery study was based on plants growing in 1-m × 10-m belt transects (Figure 1) associated with, but exterior to, the 14 forest unit and 5 once-burned unit circular plots just described

(Figure 1). These transects were permanently marked with rebar stakes at each corner. The burn recovery study used plants growing inside the boundaries of 10 twice-burned unit plots.

Clipping Recovery Study

Beargrass plants were measured in the order that they were encountered, beginning at the northernmost end of the transect associated with the northeast quadrat of the circular plot (Figure 1). Each transect was subdivided into three equal segments. In 2014, we measured the first 10 plants encountered in the transect. If 10 plants were not encountered in the first transect, we sampled additional one-third transect segments in the other transects, proceeding in a clockwise direction until a total of 10 plants had been located and measured.

We measured basal width (cm), crown width (cm), foliar height (cm), number of offsets, and crown biomass (nearest 0.1 g) on 227 plants. Basal width was measured as the plant width where it emerged from the ground. Basal and crown widths are presented as the average of two measurements taken at right angles. Foliar height was measured to the top of the tallest leaf, and the number of offsets was determined by counting the number of leaf whorls and confirmed after clipping. After measuring, the foliage was clipped, leaving leaf stubs less than 1 cm in length. Dead leaves were discarded, and the live foliage was dried to a stable weight at 60 °C and weighed.

In 2016, we returned to the site to remeasure the plants clipped in 2014. As the plants had not been individually marked, we depended on their position in the transect and previous measurements to identify them. For various reasons, we could not always be confident that we were matching a plant in 2016 to its measurements in 2014, and so plants that we were unsure of were rejected. We confidently relocated 112 plants, of which 31 were dead; therefore, we repeated the measurements on the 81 live plants and marked them with pin flags for subsequent measurements.

We randomly assigned each live plant to one of two groups to see how plants responded to one versus two clipping treatments. One group of 39 plants was remeasured, but not clipped a second time (the "clipped-once group"). The other group had 42 plants and was clipped a second time in 2016 (the "clipped-twice group"). At the end of the experiment in 2017, both groups were remeasured and clipped for biomass determination. We also relocated and measured 8 plants in the once-burned unit which had been measured and clipped in 2014. We did not statistically analyze this small dataset but present it for the insight it offers in interpreting the study.

As an indicator of competition level, we separately estimated the percentage of crown overlap from each species (including beargrass plants) with each sample beargrass plant in 2014. We then summed the species' covers by groups: 1) herbs, vines, and ferns, 2) conifers (equal to nearly all the overstory), and 3) all species (total cover group). As salal was the dominant shrub, we also separately report its overlap with beargrass plants.

In 2014, we indexed light competition with canopy cover estimates and photosynthetically active radiation measurements (PAR, 400 to 700 nm spectral band). The PAR measurement height (0.5 m) was typically above the salal canopy and measured overstory shade. Photosynthetically active radiation was measured at the middle of each transect segment with an AccuPAR ceptometer (Decagon Devices, Pullman, WA) in June or July, within 2 hours of solar noon on cloudless days. The instrument was held level and pointed south. A separate PAR sensor logged full-sun PAR in the twice-burned unit. Proportion of full PAR was obtained by dividing the sample location PAR by the full-sun PAR corresponding in time. Each transect segment PAR measurement was used for the plants in that segment.

Burn Recovery Study

In 2015, prior to the second prescribed burn, we measured all mature beargrass plants on the ten twice-burned unit plots for a total of 74 plants. As in the clipping study, we measured the foliar height, crown width, basal diameter, and offset number. These variables were remeasured in 2016 and 2018. We also recorded the number of inflorescences each plant produced in 2016, 2017,

and 2018. In 2018, we recorded whether foliar browse had occurred for each plant.

Eight days after the burn, we assessed the percentage of the crown that had been scorched by the fire. We defined scorched crown as any portion of the foliage that had died since our last measurement. Thus, it included dead leaves and portions of leaves, as well as portions that had been consumed by the fire. Percentage crown scorch was a visual estimate of the portion of the crown that was killed or consumed (estimated to the nearest 1%).

Description of the Prescribed Burn—The prescribed burn was conducted from 1400 to 1600 PDT on 23 September 2015. Air temperatures outside of the burn varied from 18 to 20 °C. The relative humidity during the burn was 50 to 52%, but rose to 63% at the end. Wind speed was 0 to 5 km hr^{-1} from the south. Fine fuel moisture was 7 to 8%. The mean volumetric soil water content measured at three locations on each plot just prior to the burn averaged 15.1% (standard deviation [SD] = 2.1). Soil moisture was measured with a Hydrosense sensor (Campbell Scientific, Logan, UT), which used standard laboratory calibrations to predict percentage volumetric water content in the range from air dry to saturation integrated over the 12-cm length of the probe.

During the burn, we recorded 2-cm-depth soil temperatures at two points on each plot using OM-EL-USB-TC single channel recorders (Omega, Norwalk, CT), programmed to record every 10 seconds with type K thermocouples. We also recorded temperatures at 50 and 100 cm height in the air, as well as on the soil surface, and at 2-cm soil depth at four locations that represented typical plant community/fuel types. For these measurements we used OM-CP-Quadtemp2000 recorders (Omega, Norwalk, CT) with four channels that recorded twice every second with type K thermocouples.

The maximum 100-cm air temperatures varied from 27.7 to 153.4 °C (mean = 74.5 °C, SD = 56.8) at the four selected locations. The 50-cm maximum air temperatures ranged from 21.1 to 146.1 C (mean = 64.0 °C, SD = 56.0). The soil surface maximum temperatures ranged from 13.1 to 35.5 °C (mean = 25.7 °C, SD = 10.6). Maximum 2-cm soil temperatures directly below these surficial measurements averaged 16.0 °C (SD = 2.8). Over all plots, the maximum 2-cm depth soil temperature measurement taken at three points on each plot averaged 26.5 °C (SD = 18.2), with a maximum of 94 °C and a minimum of 13 °C.

Statistical Analysis

All data analyses were conducted in SAS version 9.4 (SAS Institute, Inc. 2016) with a significance level (α) of 0.05. Mention of marginal significance refers to analyses resulting in $\alpha > 0.05$ but ≤ 0.1.

Clipping Recovery Study—As there was no difference in the treatment of plants prior to 2016, we combined data from the clipped-once and clipped-twice groups and used a *t* test (SAS Proc *t* test) to determine whether size and biomass in 2014 of the plants that died by 2016 were different from those that were still alive in 2016. Folded *F* tests indicated that the variances of the basal diameters and crown biomass for plants that lived or died was highly unequal ($P < 0.0001$), indicating that the Satterthwaite *t* test was more appropriate than the pooled *t* test for these analyses. However, the variances were equal in the case of crown width and foliar height, thus pooled *t* tests were used for these.

We also compared the overlapping cover of several species and species groups with beargrass plants that lived with those that died. As much of the data were skewed with "0" values, we used a nonparametric Kruskal-Wallis test (SAS Proc Npar1way). We analyzed the values of PAR associated with each plant in the same way.

Paired *t* tests were conducted on repeated plant measurements of the five plant size and biomass variables from different years (SAS Proc Means). For the clipped-twice group, we compared 2014 measurements with 2016 measurements when the plants were again clipped, then made a comparison of 2016 and 2017 measurements after the second clipping. We repeated these tests on the clipped-once group plants. Finally, we combined clipped-once group and clipped-twice group plants

for a paired *t* test comparison of measurements made in 2014 with those from 2016.

Burn Recovery Study—For the 34 surviving plants with 100% crown scorch, paired *t* tests were conducted on repeated measurements of basal diameter, crown width, foliar height, and the number of offsets from 2015 before the burn and 2018 data from three years after the burn (SAS Proc Means). We then calculated the three-year difference between these same 2015 and 2018 measurements and used independent group *t* tests to compare the differences with the corresponding three-year differences in measurements from the 37 clipped-once group plants. Folded *F* tests indicated that the variances of basal diameters, crown biomass, and numbers of offsets were unequal, indicating that the Satterthwaite *t* test was more appropriate for them, but the variances were equal for foliar height, so a pooled *t* test was used for it.

We used multiple linear regression (SAS Proc Reg) to quantify the effects of pre-burn 2015 plant dimensions and percentage crown scorch on the 2016 and 2018 post-burn dimensions. We present each analysis R^2 value as well as the partial r^2 values attributable to each 2015 plant dimension and the percentage crown scorch. We also used multiple regression for the analysis of the total number of inflorescences produced per plant in 2018 as a function of percentage crown scorch sustained in 2015, and either the 2015 number of offsets, basal diameter, crown width, or foliar height.

To determine if browsing on post-burned plants varied by scorch level, we used a *t* test to compare the percentage of 2015 crown scorch on plants browsed in 2018 to those not browsed. A folded *F* test indicated that variances of the two groups were not equal ($P = 0.0140$), indicating that the Satterthwaite *t* test was more appropriate than the pooled *t* test. We further analyzed the foliar browse data by comparing the proportions of plants browsed by three crown-scorch classes (0–1%, 25–95% and 100%) with Kruskal-Wallis tests. No plants were present in the gaps between classes. We also compared the percentage of plants in each class that flowered as well as the mean percentage of inflorescences on each plant that were browsed within each crown scorch class in the same way.

Results

Clipping Recovery Study

Thirty-one out of 112 plants (28%) died in the combined clipped once and twice groups by 2016. In 2014, the plants that died had a mean basal diameter of 2.5 cm (SD = 2.0), and for those that lived, mean basal diameter was 5.1 cm (SD = 4.1), which was a significant difference ($P < 0.0001$). Similarly, the crown widths of plants that died (47.0 cm, SD = 15.2) were significantly smaller than those that lived (60.4 cm, SD = 17.9, $P = 0.0003$). The 2014 crown biomass of plants that died averaged 8.9 g (SD = 14.4), but those that lived averaged 24.5 g (SD = 43.8), which was significantly different ($P = 0.0057$). Following the second clipping of the clipped-twice plants, two more plants (5% of the survivors) died by 2017. In the same interval, two plants (5% of the survivors) died in the clipped-once group.

Total cover and summed cover of herbs, vines, and ferns were significantly greater for beargrass plants that died than for those that lived (Table 1). Conifer cover (predominantly from overstory trees) and salal cover were marginally higher for plants that died. Photosynthetically active radiation was not significantly different between plants that died and those that lived.

The clipped-twice group basal diameters decreased significantly from 2014 to 2016 after crown removal and continued to shrink from 2016 to 2017 after a second crown removal (Table 2). For the clipped-once group, basal diameter shrinkage was marginally significant between 2014 and 2016, but the diameters remained essentially unchanged in 2017. The analysis of the combined clipped once and twice groups suggested significant shrinkage by 2016 (Table 2).

In all cases, crown widths were significantly smaller two years after crown clipping, and in the clipped-once group, were still smaller three years after a single clipping, although it appears that some recovery may have begun by then (Table 3). Two years after clipping, crown width

TABLE 1. Mean values (SD) of overlap in percentage canopy cover of competing taxa and photosynthetically active radiation (PAR; proportion of full sun at 0.5 m height) for beargrass plants that either lived or died after the 2014 clipping treatment. Crown overlap and plant mortality were assessed in 2014 and 2016, respectively. The Kruskal-Wallis (KW) *P* values are shown. These comparisons were based on 77 live plants and 35 dead plants.

	Salal	Conifers	Herbs/vines/ferns	Total cover	PAR
Live plants	59.0 (33.4)	96.5 (50.1)	6.7 (20.1)	168.6 (54.5)	0.19 (0.16)
Dead plants	68.9 (32.8)	111.4 (45.5)	8.2 (23.6)	194.8 (53.4)	0.15 (0.13)
KW *P*	0.0960	0.0948	0.0388	0.0234	0.2576

TABLE 2. Sample size and mean (SD) values for beargrass basal diameters (cm) before (2014) and 2 to 3 years after clipping (2016 to 2017). Also shown are the *P* values associated with paired *t* tests for the two means occurring to the left of the *P* value. Clipping treatment 1 = clipped-once group. Clipping treatment 2 = clipped-twice group.

Clipping treatments	*n*	2014	2016	*P*	*n*	2016	2017	*P*
1 (2016)	39	5.0 (4.0)	3.7 (4.6)	0.0643				
1 (2017)	37	4.9 (4.1)					3.5 (4.7)	0.0641
2	42	5.3 (4.3)	4.2 (4.1)	< 0.0001	40	4.5 (4.2)	3.7 (4.2)	0.0002
1 and 2	81	5.1 (4.1)	3.9 (4.3)	0.001				

TABLE 3. Sample size and mean (SD) measurements for beargrass crown widths (cm) before (2014) and 2 to 3 years after clipping (2016 to 2017). Also shown are the *P* values associated with paired *t* tests for the two means occurring to the left of the *P* value. Clipping treatment 1 = clipped-once group. Clipping treatment 2 = clipped-twice group.

Clipping treatments	*n*	2014	2016	*P*	*n*	2016	2017	*P*
1 (2016)	39	59.1 (15.7)	33.7 (18.3)	< 0.0001				
1 (2017)	37	59.2 (16.1)					36.1 (16.7)	< 0.0001
2	42	61.7 (19.8)	37.9 (12.6)	< 0.0001	40	38.9 (11.8)	26.0 (11.6)	< 0.0001
1 and 2	81	60.4 (17.9)	35.9 (15.6)	< 0.0001				

TABLE 4. Sample size and mean (SD) measurements for beargrass foliar heights (cm) before (2014) and 2 to 3 years after clipping (2016 to 2017). Also shown are the *P* values associated with paired *t* tests for the two means occurring to the left of the *P* value. Clipping treatment 1 = clipped-once group. Clipping treatment 2 = clipped-twice group.

Clipping treatments	*n*	2014	2016	*P*	*n*	2016	2017	*P*
1 (2016)	39	29.8 (6.4)	21.1 (9.1)	< 0.0001				
1 (2017)	37	29.7 (6.5)					22.1 (9.5)	< 0.0001
2	42	30.4 (9.0)	23.1 (7.0)	< 0.0001	40	23.8 (6.4)	16.0 (5.9)	< 0.0001
1 and 2	81	30.1 (7.8)	22.1 (8.1)	< 0.0001				

was 59% of what it was before clipping in the combined group. Two clippings two years apart resulted in a crown width only 42% of the initial pre-clipping size.

All comparisons indicated significant loss of foliar height two or three years after crown removal, although the clipped-once group data suggest some recovery began by year three (Table 4). A second clipping two years after the first caused a further loss of foliar height.

There were two marginally significant decreases in number of offsets (Table 5). These occurred in the clipped-once group between 2014 and 2016, and between the same two years for the combined groups, suggesting some offset loss was likely associated with clipping.

The clipped-twice group live-crown biomass decreased significantly ($P < 0.0001$), from 24.8 g in 2014 before clipping to 5.9 g in 2016. For the clipped-once group plants, the 2017 weight of

TABLE 5 Sample size and mean (SD) numbers of offsets for beargrass sample plants before (2014) and 2 to 3 years after clipping (2016 to 2017). Also shown are the *P* values associated with paired *t* tests for the two means occurring to the left of the *P* value. Clipping treatment 1 = clipped-once group. Clipping treatment 2 = clipped-twice group.

Clipping treatments	*n*	2014	2016	*P*	2017	*P*
1 (2016)	39	2.7 (3.7)	2.4 (3.4)	0.0832		
1 (2017)	37	2.6 (3.8)			2.5 (4.3)	0.5444
2	42	2.7 (2.6)	2.7 (2.6)	0.4119		
1 and 2	81	2.7 (3.2)	2.6 (3.0)	0.0573		

TABLE 6. Mean (SD) plant size and number of offsets for the 35 beargrass plants in the open, twice-burned unit having 100% crown scorch for pretreatment (2015) and year-three data (2018). Also shown is the *P* value for the paired *t* test comparison of the pre- and post-treatment data.

	Basal diameter (cm)	Crown width (cm)	Foliar height (cm)	Offset number
Pre-treatment (2015)	11.3 (7.3)	65.3 (20.9)	36.0 (9.3)	10.3 (10.3)
Post-treatment (2018)	11.4 (8.0)	57.7 (21.1)	27.6 (8.6)	10.4 (10.4)
P	0.9591	0.0002	< 0.0001	0.8200

10.6 g was significantly less ($P = 0.0011$) than their 2014 weight of 24.5 g.

The mean basal diameter of eight plants from the once-burned unit decreased from 12 to 11 cm, the crown width from 65 to 54 cm, and the foliar height from 35 to 32 cm after clipping. The number of offsets remained at 10.

Burn Recovery Study

We observed crown scorch levels from 0 to 100%. Nearly half had 100% crown scorch. During our crown scorch assessment, 8 days after the burn, we frequently observed green leaves or leaf bases near the center of plants, suggesting burn temperatures near the center were lower than near the edge of the crown.

Only one of 74 plants died after burning. That plant sustained 100% crown scorch. Its crown width and foliar height were 21.5 cm and 13 cm, respectively, which were the smallest values in the sample. It had only 3 offsets and a basal diameter of 3.5 cm (both among the nine smallest in the sample). The small crown size relative to its base suggests that this plant had poor vigor.

In comparing measurements made before the burn in 2015 and those from 100% crown-scorched plants in 2018, we found that crown width and foliar height were significantly smaller in 2018, but this was not the case for basal diameter and the number of offsets (Table 6). Three years post burn, the crown widths of plants with 100% crown scorch averaged 7.6 cm smaller than their pre-burn crown widths.

The regression analysis indicated that all 2015 plant measurements were positive predictors of post-burn measurements, but percentage crown scorch was a negative predictor. Initial plant size measurements were stronger predictors of post-burn plant size than was the percentage of crown scorch (Figure 3). Percentage crown scorch was a significant predictor only in the 2016 and 2018 crown width and foliar height models, and the 2016 offset count model. The crown scorch partial r^2 values for foliar height and crown width were larger in 2016 than in 2018, suggesting a declining effect of crown scorching.

In 2018, 24.7% of all plants had foliar browse. Among plants that were not browsed, there were similar numbers of previously crown-scorched (30) and non-crown-scorched (25) plants, but of the 18 plants that were browsed, all but one had experienced some level of crown scorching. Plants that were not browsed averaged 48.2% crown scorch (SD = 47.2), while plants that were browsed averaged 88.9% crown scorch (SD = 27.0), which was a significant difference ($P \leq 0.0001$). Over three crown-scorch classes, foliar browse increased linearly, with only 4% of non-crown-scorched and 40% of entirely crown-scorched plants having been browsed (Figure 4).

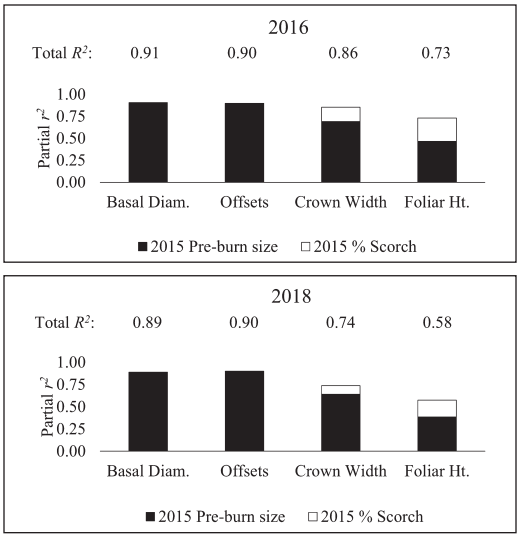

Figure 3. The R^2 and partial r^2 values for the regressions of 2016 (top) and 2018 (bottom) measurements of beargrass plants, as a function of the plant measurements in 2015 before the prescribed burn and the percentage of crown that was scorched by the fire. All models were highly significant ($P \leq 0.01$).

One year after the burn, only one non-scorched plant flowered with one inflorescence. In 2017, 26 plants bore 71 inflorescences, and in 2018, 47 plants bore 173 inflorescences. Thus, the number of plants flowering increased linearly for at least three years after the burn. Crown scorch on the 47 flowering plants in 2018 averaged 62%, and on the non-flowering plants it averaged 52%, which was not significantly different. The percentage of crown scorched flowering plants was 0% in year one, 36% in year two, and 64% in year three. In 2018, 84% of flower stalks borne on crown-scorched plants were browsed, compared to 77% on non-crown-scorched plants. There was no difference among three crown-scorch classes in the percentage of plants that flowered, but the amount of browse on the flowers was marginally different (Figure 4).

The number of inflorescences in 2018 was best predicted by the number of offsets in 2015 (before the burn) and the percentage of crown scorch:

$$\text{Inflorescences plant}^{-1} = 0.2901 \times \text{pre-burn offsets} + 0.1325 \times \% \text{ crown scorch} - 1.63647$$

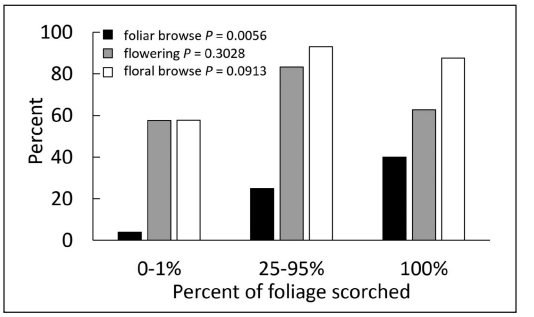

Figure 4. Browse and flowering of beargrass by three levels of crown scorch three years after the 2015 burn. Crown scorch is the percentage of each plant's foliage that was killed in the prescribed burn. Foliar browse is the percentage of plants browsed in each crown-scorch class (total = 18). Floral browse is the mean percentage of flower stalks browsed on each plant in each crown-scorch class (total = 138). Flowering is the percentage of plants in each crown-scorch class that were flowering. Crown-scorch class sample sizes: $n = 26$ for 0 to 1%, $n = 12$ for 25 to 95%, $n = 35$ for 100% groups.

The number of 2015 offsets explained 58% of the variation, and the percentage of crown scorch explained 3%. Models in which basal diameter ($R^2 = 0.47$), crown width ($R^2 = 0.47$), or foliar height ($R^2 = 0.22$) were substituted for number of offsets were significant but weaker. The amount of variation explained by the percentage of crown scorch (3%) was the same regardless of which size measurement was included in the model and it was always a positive relation.

Comparison of Clipped and Burned Plants

Three years after treatment, the forest unit clipped-once group plant crowns averaged 23.0 cm smaller than before clipping, which was significantly more loss than that for the 100% scorched twice-burned unit plants (7.6 cm) (Table 7). However, burned plant basal diameter remained essentially unchanged, and clipped plants lost 1.4 cm. Foliar height and offset number losses did not differ. Forest unit clipped-once group plants had 26% mortality by year three, and plants suffering 100% crown loss in the twice-burned unit had only 3% mortality. Plants in the once-burned unit were only remeasured two years after clipping, but at that time the plant crowns averaged 11 cm smaller, the basal diameters were 1 cm smaller,

TABLE 7. Comparison of the changes in beargrass plant size and number of offsets between pre-treatment and year three for plants with 100% crown scorch (from the twice-burned unit) versus once-clipped plants (from the forest unit). Shown are the mean (SD) values for the difference between years (2015 and 2018 for the twice-burned unit data, and 2014 and 2017 for the forest unit data).

	Basal diameter (cm)	Crown width (cm)	Foliar height (cm)	Offset number
Crown-scorched plants	0.0 (2.5)	-7.6 (11.0)	-8.4 (7.0)	0.1 (3.0)
Clipped plants	-1.4 (4.4)	-23.1 (15.8)	-7.5 (8.9)	-0.1 (1.0)
P	0.1	< 0.0001	0.6	0.7

the foliar height was 3 cm smaller, and there was no mortality.

Discussion

Clipping Recovery Study

There is a regional perception that beargrass has become less common due to tree and understory forest succession in formerly open areas that had been maintained by anthropogenic burning (Wray and Anderson 2003, Shebitz et al. 2008, Hummel et al. 2012). One such area is in lowland western Washington, where beargrass competes with dense salal under heavy conifer shade. Our findings suggest that combined competition from trees and shrubs results in physiological attrition that could lead to progressive loss of beargrass and that there is potential for beargrass harvesting to contribute to this loss. Thus, it seems reasonable that cumulative decline in beargrass plant vigor over the years since overstory crown closure in the 1950s (Peter et al. 2017) resulted in weak plants. Also, the high mortality that we observed, while precipitated by clipping, was likely due to their weakened condition.

Traditional and commercial harvesting of beargrass entails removing the longest inner offset leaves on partially shaded plants (Thomas and Schumann 1993, Anderson 2005, Rentz 2003, Shebitz 2005, Shebitz et al. 2009). In a simulation of traditional harvesting, removal of fewer than one-third of the inner offset leaves somewhat reduced offset survival (Hart-Fredeluces and Ticktin 2019). It has been assumed that properly harvested plants recover in 4 to 5 years, but no research confirms this (Higgins et al. 2004). However, commercial harvesting of too much foliage from plants is reportedly common (Shebitz 2005), with effects that may more closely approximate what we found with our methods. Nevertheless, clipping all the foliage, as we did, is a more drastic treatment than properly conducted traditional or commercial harvesting but is probably less damaging than complete crown scorch where heat may add to the damage. Three years after clipping, our surviving forest unit plant crown widths were only 61% of their original size, suggesting they were far from recovered.

The consequences of defoliation to a plant depend on its vigor. High mortality (28%), along with greater shrinkage of clipped forest unit plant crown width, foliar height, and basal diameter compared to open-grown clipped or crown-scorched plants, suggests forest unit low plant vigor, possibly due to an inability to accumulate sufficient carbohydrate reserves in this understory environment. Thus, plants growing in competitive, shaded stands have less recovery capacity than open-grown plants, with possible consequences for the shade-grown plants preferred by commercial harvesters.

Our analysis suggested that offset numbers did not increase, and some may have been lost after clipping. In any case, the basal diameter, which reflects the size of the rhizome mass at its top, shrank after clipping, perhaps partly due to offset loss. Hart-Fredeluces and Ticktin (2019) found that even with limited foliar harvesting, the probability of offset loss increased, but such foliar removal also stimulated new offsets to grow in healthy plants (Shebitz et al. 2008, Hart-Fredeluces and Ticktin 2019). This did not happen with our surviving forest unit plants within the first three years, and all offsets died along with the entire plant in 28% of our sample.

Most clipped plant mortality came from the smallest plants. Plant size is an important fitness component (Solbrig 1981), and small plants generally have higher mortality rates than larger plants (Harper 1977). Thus, beargrass plant size is an indicator of its ability to recover from injury and might be considered in foliar harvest guidelines.

High mortality was associated with higher total cover, suggesting these conditions decreased plant vigor. Laursen (1984) also found that overtopping cover had a negative effect on beargrass. However, shade tolerance of a species varies depending on belowground resources (Smith and Huston 1989, Coomes and Grubb 2000), as determined by environment and competition. Our site was droughty, infertile, and had high overstory and understory cover. When we measured PAR above the understory, we found no relation to mortality. Over a broader range of stand conditions, similarly measured PAR was important (Peter et al. 2017), but in the uniformly low light under our forest canopy (PAR = 0.15 to 0.19), understory shade and soil resources became critically important (Chapin et al. 1987). Trenching experiments have shown a large effect of root competition on forest understory cover (Lindh et al. 2003) and indicate that soil nitrogen and water may be more limiting than light (Riegel et al. 1992).

In examining the components of total cover, we found evidence for overstory and understory suppression of beargrass. Conifer cover (nearly all from overstory trees) was marginally higher over plants that died after clipping than over those that lived. Also, both the herbs, vines, and ferns group cover and the cover of salal were higher over plants that died than those that lived. These two groups constituted nearly all the understory.

Shebitz et al. (2009) and Laursen (1984) found that beargrass was sensitive to shrub competition, which was consistent with our findings and probably reflects primarily belowground competition. With 85% cover, salal was the dominant understory species, and beargrass that died grew in denser salal than did those that lived. Salal is a recognized competitor of tree seedlings and saplings, reducing their growth primarily through belowground competition, especially for soil nitrogen (Chang et al. 1996, Chang and Preston 2000, Mallik and Prescott 2001). Salal produces a canopy of evergreen leaves that is taller than beargrass and forms a dense mat of roots and rhizomes, so it likely competes with beargrass above- and belowground. Even in the understory, salal can have 11 m of rhizomes and 19 stems per m^2, and in the open it can have 178 m of rhizomes and 346 stems per m^2 (Huffman et al. 1994). As it rapidly responds to overstory canopy gap formation with rhizome growth of 3 to 5 m yr^{-1} (Huffman et al. 1994), it can likely occupy new canopy gaps faster than beargrass. Salal's uneven-aged structure, shade tolerance, and aerial stem longevity allow it to persist, shading out herbs and suppressing tree regeneration (Huffman et al. 1994). Thus, management of beargrass may require management of both the overstory and understory.

Beargrass roots extending 4 m from the plant allow it to exploit resources over a wide area, permitting it to survive changes to the competitive understory environment over long periods (Antos et al. 2021). This would be advantageous in most circumstances, but the superior height of salal coupled with its high rhizome density, homogeneity, and responsiveness would be especially difficult competition even for beargrass, particularly in the presence of overstory shade and tree root competition. Thus, the longevity of beargrass in understory environments may hinge on the composition of the understory. Clearly there are understories in which beargrass appears to have survived for hundreds of years (Peter et al. 2017, Antos et al. 2021), but there are also places where this may not be possible without stand density management.

Slow-growing species generally have lower mortality after defoliation than fast-growing species, as they recover from defoliation by mobilization of larger belowground pools of stored carbohydrates than those of faster growing species (Atkinson et al. 2014). Beargrass is a slow-growing species with considerable belowground investment, and we found it survived defoliation well in open environments. Thus, the high mortality of clipped forest unit plants suggests their stored carbohydrate reserves were critically depleted.

As beargrass plants in highly shaded situations produce few or no seedlings (Peter et al. 2017, Antos et al. 2021), any loss of plants has important long-term implications.

Partial and complete foliar removal can stimulate growth of new offsets (Shebitz et al. 2008, Hart-Fredeluces and Ticktin 2019), possibly from released dormant axillary buds. Axillary buds commonly replace growing shoots after the apex is damaged or removed, or when the shoot differentiates into an inflorescence (Stafstrom 1995). However, young leaves may have a greater influence on axillary bud dormancy than the shoot apex (Weiss and Shillo 1988, White et al. 1975, Hosokawa et al. 1990). Thus, removal of the young leaves may release axillary buds in sufficiently vigorous plants. As we removed all the foliage (young and old) in our clipping treatments, there is reason to expect that new offsets would be released to grow. However, we did not observe an increase in offsets in forest unit plants, and if anything, our data suggest a decrease up to three years post-treatment, pointing to a pre-treatment lack of vigor.

Light is the primary resource limiting flowering of understory plants (Chazdon 1991, Niesenbaum 1993, Cunningham 1997). We observed beargrass flowering almost exclusively in the once- and twice-burned units where there was little overstory and mostly larger plants with abundant offsets. Moreover, seedlings were often observed in these more open conditions, but never in the forest unit. Peter et al. (2017) suggested that reproduction in the forest unit may not have occurred since crown closure 60 years ago, and Antos et al. (2021) observed only minimal flowering and no seedling establishment in an old-growth forest over a 40-year period of observation. Thus, beargrass may not only be physiologically challenged in forest understories, but also fails to reproduce there. Without reproduction, continued decline in vigor, coupled with disease, age, browse, and stochastic processes such as tree fall and uprooting by windthrown tree root wads, may progressively thin the population until it dies out.

Burn Recovery Study

Beargrass in the southeastern Olympic Peninsula is likely a legacy of a long history of anthropogenic burning. Slow forest succession due to the droughty, infertile soils, and shade tolerance extended its survival under the developing forest overstory upon cessation of burning (Peter and Shebitz 2006). Because the shade tolerance of beargrass is limited, anthropogenic fire favors beargrass.

Fire benefits beargrass by reducing competition from shrubs and trees (Crane 1990), increasing light and soil temperature (Maule 1959), and releasing nutrients into the soil (Hart-Fredeluces and Ticktin 2019). The main adaptation of beargrass to fire is its ability to sprout new foliage from underground rhizomes (Crane 1990), which usually occurs by the next season (Rentz 2003, Shebitz et al. 2009, Hart-Fredeluces and Ticktin 2019). Rentz (2003) suggested that the dense tuft of green foliage above the offset meristems protects them from heat during low-intensity fires. We observed green leaves surviving near the center of crown-scorched plants, which supports this idea, but to our knowledge, no research confirms it. Other species similarly protected from heat include longleaf pine (*Pinus palustris* Mill.) (Wahlenberg 1946, Keeley 2012) and Australian grass trees (*Xanthorrhoea australis* R.Br) (Gill and Ingwersen 1976).

Our model showed that crown scorch barely decreased the number of offsets in surviving plants, and then only for one year post-burn. Shebitz et al. (2009) found no offset mortality from low-intensity burning, or even in the first year after more intense burning. Moreover, they found offsets increased in the second year, suggesting a delayed stimulation, but we did not see this in our study.

In a high-intensity wildfire, offset mortality was still just 6.4% (Hart-Fredeluces and Ticktin 2019), suggesting considerable heat resistance. The question arises as to whether weaker plants from denser low light forests would fare as well. The wildfire studied by Hart-Fredeluces and Ticktin (2019) burned in a forest with 78.5% canopy

closure, compared to our forest unit with 168% total (overlapping) cover over plants that lived and 198% over those that died. Even without fire, we lost 28% of plants from clipping, compared to just 3% in the twice-burned unit. As our forest unit plants appear to have suffered from shade and root competition and are growing in an area with high shrubby fuel loadings, we suggest they would suffer high mortality in a wildfire.

Although post-burn plant size was negatively related to crown scorch, the relation weakened after three years, suggesting some recovery was taking place. Foliar height was most affected, followed by crown width, but three years post-burn, foliar height and crown width were still significantly smaller than before the burn. Shebitz et al (2009) also documented decreased foliar height, crown area, leaf length, and percent cover after low-intensity burns. Interestingly, they found no such changes in open area plots where all vegetation, including the beargrass, was clipped to ground level, suggesting that fire damage was more harmful than clipping (Shebitz et al. 2009). Importantly, in this case both clipped and burned plants were open grown. When we compared open-grown twice-burned unit plants that had lost their crowns to scorch with shade-grown forest unit plants that had lost their crowns to clipping, we found significantly greater crown width loss in the clipped forest unit plants.

In 2002, our once- and twice-burned units were growing under similar overstory and understory conditions as were present in the forest unit. At that time, the stand was opened by overstory thinning followed by the first prescribed burn. The current basal diameters of these released and now open-grown beargrass plants were more than twice that of the forest unit plants, and the number of offsets was three times greater, suggesting they grew much faster than the forest unit plants after release. While the foliar heights and crown widths were similar, plants in the once- and twice-burned units had more than 4.5 times as much crown biomass as forest unit plants (Peter et al. 2017), suggesting that a superficial observation of crown size is a poor indicator of plant vigor. Crown regeneration of open-grown clipped or crown-scorched plants was stronger than that of similar sized clipped forest unit plants, reflecting their greater capacity to respond to injury.

Burning affected the frequency at which beargrass was browsed. Three years after our burn, 25% of the plants had foliar browse, but plants that had sustained crown scorch were clearly preferred. Perhaps the less sclerified foliage that grew back after scorch is more easily chewed and digested or is more nutritious. In any case, it seems that this could have severe consequences for recovery of shade-grown burned plants, but open-grown burned plants represent a manageable resource for wildlife.

Under some circumstances, fire can increase flowering (Maule 1959, Rentz 2003, Hart-Fredeluces and Ticktin 2019) and seedling establishment (Shebitz et al. 2009) 1 and 2 years post burn. However, post-fire flowering was neither immediate, nor dramatic after our burn, and the proportion of plants flowering in 2018 in three crown-scorch classes were not significantly different. Flowering increased linearly to year three when 64% of plants flowered, but whether this was due to the burn is not certain. Our regression analysis attributed only 3% of the variation in third-year flowering to crown scorch, but if another fire associated stimulus was a more direct cause of flowering (e.g., smoke or a nutrient pulse), crown scorch would be at best an indirect and possibly poor indicator. More important was the number of offsets. Flowers are produced from offsets, and large plants have more offsets. Hart-Fredeluces and Ticktin (2019) found that offsets increased after partial leaf removal and that flowering was more likely to occur with larger offsets. Shebitz et al. (2009) did not see an increase in flowering after burning but found that offsets increased two years after burning on plants that had lost their crowns in the high-intensity fire at our Skokomish site in 2003. Perhaps these offsets would have bloomed in the third year like ours did. Further studies, including non-burned controls, are needed to determine how or whether fire affects flowering in lowland beargrass.

Conclusions

Based on our research findings and the current body of research, we conclude that beargrass, although tolerant of shade, ultimately requires more open conditions to reproduce and thrive. Its shade response probably varies with environmental conditions, including understory competition. Our analysis indicated high mortality and slow recovery for highly shaded, clipped plants, especially with higher levels of salal competition, but when crowns were removed by fire from plants in an open environment, almost no mortality was observed, and recovery was rapid. These contrasting observations suggest that the competitive environment of the forest unit plants resulted in detrimental physiological attrition.

Beargrass clearly does well in full sun, but traditional and commercial uses require at least partial shade. Lowland beargrass flowers and grows with up to 50% overstory cover (Peter et al. 2017), but flowering, growth, and survival decrease with greater shade, and competition. Combined tree and shrub root competition seems as important as overstory shade, and release from shrub competition may be one of the more important fire effects. Research, including our own, has consistently shown benefits to beargrass from prescribed burning. Fire alters beargrass foliage in ways beneficial for wildlife browse and traditional uses. However, more information on the intensity and frequency of fire is needed to optimize these and other desirable outcomes.

The beargrass population in the once- and twice-burned units was all-aged with abundant seedlings (Peter et al. 2017). While there was a range of plant sizes in our forest unit, there were no seedlings, and smaller plants suffered the greatest mortality from clipping. Surviving clipped plants shrank, suggesting that beargrass plants may die incrementally, which could account for the presence of smaller plants in the absence of seedling establishment. This could happen, for example, if offsets died without replacement. This deserves further research, as it implies the need to open the stand and treat the understory to sustain the population.

Beargrass responds well to complete overstory removal (Peter et al. 2017, Hart-Fredeluches and Tictin 2019), as shown by our once- and twice-burned unit populations, which were released from conditions almost identical to our forest unit in 2003. Such open communities provide good beargrass habitat while supporting other associated heliophytic plants, but provide poor foliage for traditional or commercial purposes. We hypothesize that beargrass populations can be sustained for traditional and commercial uses by creating stand structures that include openings large enough to provide full sun for seedling production, with partly shaded edges that provide suitable conditions for basketry foliage. A woodland matrix of moderate density would provide shady conditions for commercial foliage. Ascertaining the optimal size, density, and placement of the openings and the optimal woodland matrix density would require further research but might in part depend on other uses of the woodland. This management regime would include periodic prescribed fire (e.g., every 5 to 10 years) to reduce competing understory cover and stimulate regrowth and flowering of beargrass. Guidelines for appropriate harvesting protocols are also needed to ensure that individual plant vigor is maintained or allowed to recover.

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