

Monitoring Long-term Change in Coastal Prairies of the Marin Headlands

by Eric Wrubel¹ and Marie Denn² Photos courtesy of Eric Wrubel

Introduction

The Plant Community Monitoring Protocol of the San Francisco Bay Area Network of National Parks (SFAN) was established to track long-term change in vegetation composition and structure in four national park units located along the central California coast. Here we present the first three years of monitoring results in the coastal prairies of the Marin Headlands, colloquially referred to as bald hills prairie (Holland and Kiel 1995).

California's coastal prairies are celebrated for their biodiversity and endemism, yet have been significantly diminished in extent and quality by development and invasive species (Holland and Keil 1995, Stromberg et al. 2001, Jantz et al. 2007). Coastal prairie is defined here as grassland located in close proximity to the shoreline, with frequent exposure to maritime fog and salt spray, and typically situated on a terrace, bluff, or coastal headland in central or northern California (Sawyer et al. 2009). Competition from introduced annual grasses has been implicated in declined abundance of native species (Corbin et al. 2007).

The windswept Marin Headlands are located in southern Marin County in the Golden Gate National Recreation Area (GGNRA). It is a largely treeless peninsula of shrublands and grasslands bordered on the southwest by the Pacific Ocean and the Golden Gate, and the San Francisco Bay to the northeast. The bald hills prairies of the Marin Headlands are primarily situated on shallow soils derived from Franciscan chert. The chert of the Marin Headlands is high in silica and iron oxides, and is resistant to weathering (Elder 2001), forming red-colored balds on many of the ridgelines. These chert-derived soils appear to be less productive than many other Franciscan soil types since cover of bare, mineral soil and bedrock can be high and vegetation coverage is often sparse (Steers and Spaulding 2013).

The bald hills prairies are dominated by introduced annual grasses and forbs, but are notable for their richness of native perennial grasses and forbs, which are diagnostically present with over 10% relative cover. The dominant native perennial grasses are Idaho fescue (*Festuca idahoensis*) and purple needlegrass (*Stipa pulchra*), and the dominant

native perennial herb is coastal soap plant (*Chlorogalum pomeridianum* var. *divaricatum*). These grasslands are best described as an Idaho fescue-purple needlegrass association, a provisional association within the Idaho fescue alliance (Sawyer et al. 2009), intermediate between the purple needlegrass-dominated bald hills of the Central Coast Ranges, and Idaho fescue-dominated bald hills of the North Coast Ranges. Associated native grasses and herbaceous perennials include California oatgrass (*Danthonia californica*), Hall's bentgrass (*Agrostis hallii*), short-stem sedge (*Carex brevicaulis*), golden aster (*Heterotheca sessiliflora* ssp. *bolanderi*), and California plantain (*Plantago erecta*). Dominant non-native species include rattlesnake grass (*Briza maxima*), brome fescue (*Festuca bromoides*), slender wild oat (*Avena barbata*), and hairy cat's ear (*Hypochaeris radicata*).

The coastal prairies of the Marin Headlands provide vitally important habitat for wildlife, including critical habitat for the federally endangered mission blue butterfly (*Icaricia icarioides* ssp. *missionensis*). These grasslands also support high densities of Botta's pocket gopher (*Thomomys bottae*); gopher disturbance and herbivory have been shown to play a significant role in vegetation dynamics here and in other California grasslands (Steers and Spaulding 2013, Schiffman 2007).

Composition and structure of California grasslands fluctuates seasonally and annually at multiple temporal and spatial scales (Heady 1956, Hobbs and Mooney 1995, Jackson and Bartolome 2002). Species and functional groups (annuals, perennials, legumes, etc.) in California grasslands have long been known to respond differentially to weather patterns (Pitt and Heady 1978), which interact with numerous other factors such as fossorial mammal disturbance, grazing, and edaphic conditions. Temporal change in grasslands is also subject to hysteresis — lag effects from causal factors in previous years. Annual grasses and forbs especially vary widely in abundance and cover from year to year with annual variations in precipitation, with some years favoring grasses, some years favoring legumes such as clovers, and some years favoring filarees (*Erodium* spp.), or other tap-rooted forbs. The Plant Community Monitoring Protocol is designed to detect a 20% change in vegetation structure and composition metrics between sampling events. To understand the significance of long-term change in this and other grassland vegetation types, we need a better understanding of typical short-term variance in structure and composition. Therefore, we are currently monitoring these grasslands annually, and we may be able to lengthen the sampling interval to every two or more years in the future, when we can better predict the normal range of interannual change.

The first three years of bald hills prairie monitoring (2015, 2016, and 2017) coincided serendipitously with historic precipitation extremes,

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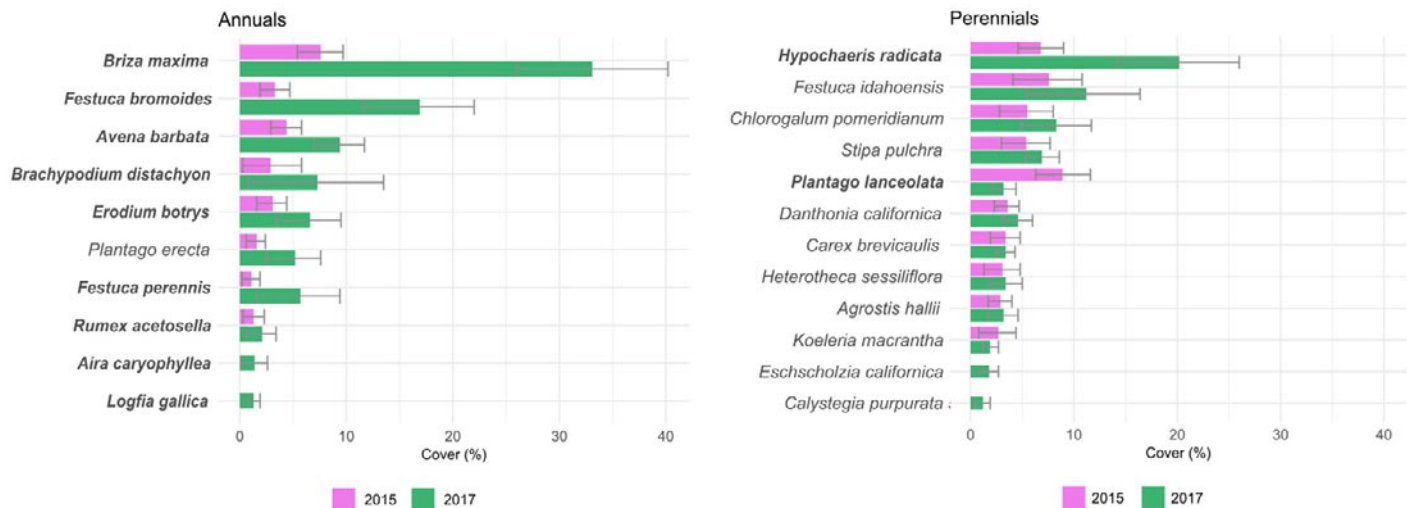


Figure 1. Percent cover of annual (left) and perennial (right) vascular plant taxa with 1% or greater cover found in 1-m² quadrats in 2015 (magenta bars), and 2017 (green bars). Non-native species names in bold. Error bars show bootstrapped 90% confidence intervals around the means.

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allowing us to compare vegetation response to the range of precipitation that may be expected in the future. The year 2015 marked the fourth consecutive year of record-breaking drought in California, perhaps the most severe period of drought in a millennium, due to accumulated moisture deficits caused by historically low precipitation and high temperatures (Griffin and Anchukaitis 2014). At the San Francisco Downtown weather station, less than 10 km from the Marin Headlands, the driest and hottest years on record were logged between 2013 and 2015 (NOAA 2015). This historic drought period was followed by above-average precipitation in calendar years 2016 and 2017.

Methods

In 2015, we established permanent monitoring plots in 12 randomly selected grassland stands in the Marin Headlands. One plot was established at the center point of each stand. These stands were all located on chert-derived soils, with Idaho fescue and purple needlegrass diagnostically present. The full methodology for site selection and plot establishment is described in the document *Plant Community Monitoring Protocol for the San Francisco Bay Area Network of National Parks* (Steers et al. 2016a, b). The protocol is designed to sample vegetation attributes across a wide variety of physiognomic types. Here we report on the sampling protocols for herbaceous plant richness and cover.

In each of the plots, we established three transects radiating from the center point. We measured vascular plant cover in three 1-m² quadrats, each placed at 4.57-m from the center point along one of the three transects. Within each quadrat we evaluated plant presence at 50 regularly dispersed points, by dropping a small-diameter pin flag vertically at each point. We recorded the name for each plant less than 3-m in height that was “hit” by the pin, identifying all vascular plants to the minimum taxonomic rank (species, subspecies, or varieties), the smallest taxonomic units recognized by the *Jepson Manual*

(Baldwin et al. 2012). Hereafter in this article we will refer to subspecies and varieties simply as “species” for the sake of simplicity. At points where no plants were observed, we recorded ground variables at the point, such as “litter” and “bare ground”. We estimated cover of each plant taxon encountered by first averaging the cover among the three 1-m² quadrats measured in each plot, then averaging cover for each taxon among all plots. For all these metrics we created bootstrapped 90% confidence intervals around the estimated mean (Efron 1981).

The area of a circle of radius 7.32-m from the plot center point was searched for all vascular plant taxa, and tallied to derive the metric species richness per 168.3-m². Species richness was then averaged among all plots to derive the mean. Richness within the 1-m² plots was also recorded, but is not reported here. We estimated diversity using iNEXT (Chao et al. 2016), to compute rarefaction and extrapolation sampling curves of Hill numbers for $q = 0$ (richness) and $q = 1$ (Shannon diversity) (Chao et al. 2014). The computation unit is the effective number of species, which is the idealized number of equally abundant species that would give the same proportional species abundance as a given dataset, in which all species may not be equally abundant.

Results

The cover of many species fluctuated markedly between 2015 and 2017. Figure 1 shows the mean cover of all vascular plants with 1% or greater cover within the 1-m² quadrats in the driest year (2015), and the wettest year (2017). Introduced annual grasses including rattlesnake grass and brome fescue, showed the greatest variance in cover between years, along with the introduced perennial forb, hairy cat’s ear. Mean cover of annual plants was about half that of perennials in 2015 (21.4%) and increased considerably to 101.5% by 2017,

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exceeding cover of perennials by about 25% (Figure 2, values are for plant taxa with >2% cover in the plots). Perennial cover also increased at a more modest rate from 54.4% in 2015 to 74.5% in 2017.

Figure 3 shows sample-based rarefaction and extrapolation diversity accumulation curves comparing results from 2015, 2016, and 2017 for species richness, and Shannon diversity in the 7.32-m radius plots (168-m²). These curves represent the cumulative number of effective species discovered by sampling 12 plots, and extrapolated to the number of species expected to be discovered with additional sampling of up to 30 plots. Observed (*Sobs*) species richness was significantly lower in the drought year of 2015 (*Sobs* = 80, 95% CI [72.4, 87.6]), than it was in 2016 (*Sobs* = 98, 95% CI [88.8, 107.2]) and 2017 (*Sobs* = 99, 95% CI [90.7, 107.3]), based on non-overlapping confidence intervals. Shannon diversity was also significantly lower in 2015 (*Sobs* = 56.7, 95% CI [52.8, 60.6]) than it was in 2016 (*Sobs* = 66.3, 95% CI [61.1, 71.6]), and 2017 (*Sobs* = 70.7, 95% CI [65.62, 75.2]). Richness and Shannon diversity were not significantly different between the two wetter years of 2016 and 2017, based on overlapping confidence intervals, though the confidence intervals for richness showed almost complete overlap, whereas the confidence intervals for Shannon diversity showed only partial overlap.

Conclusion

The grasslands in this study varied markedly in structure and composition in a short time frame between dry and wet weather years, supporting previous work in California grasslands (Heady 1956, Pitt and Heady 1978, Hobbs and Mooney 1995, Jackson and Bartolome 2002). Total vegetative cover increased with increased precipitation, driven mostly by increased cover of non-native annual grasses and forbs (Figures 1 and 4). Annual rattlesnake grass dominated the community in all years, but was far sparser during the drought of 2015, and increased dramatically in cover in 2016 and 2017 (Figure 1), accompanied by increases of two other non-native species: hairy cat's ear, a short-lived perennial, and brome fescue, an annual grass. The

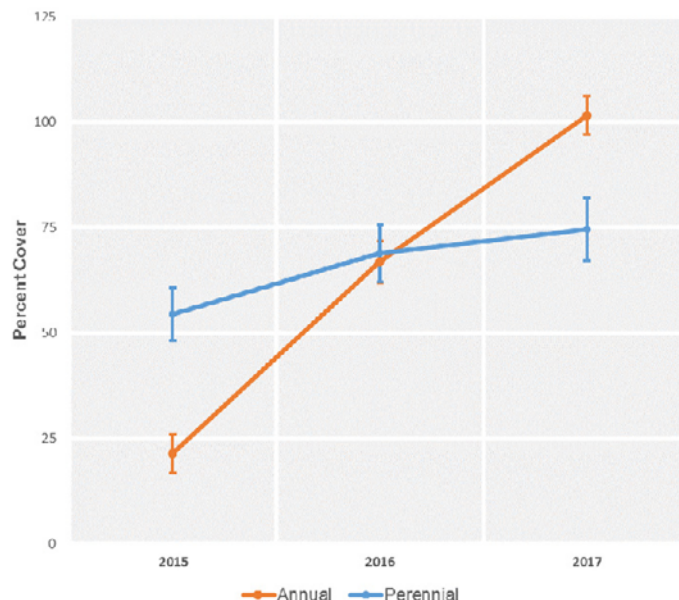


Figure 2. Mean percent cover of live annual and perennial plants with >2% cover found in 1m² quadrats in 2015, 2016 and 2017. Error bars show the standard error of the mean.

native perennial grasses, Idaho fescue and purple needlegrass, and the native perennial bulb, soap plant, increased only slightly in cover from 2015 to 2017. Cover of most species shifted significantly between 2015 and 2017 (Figure 12), although the frequency of most species in the plots remained relatively stable between years.

Annual and perennial plant cover both increased from lows in the 2015 drought year to highs in the wettest year of 2017 (Figure 2). Annual plant cover was much less than that of perennials in 2015, but well exceeded perennial cover by 2017. The bulk of the annual plant cover was contributed by non-native species, although the native annual California plantain contributed modest cover each year, reaching a high of 5.2% mean cover in 2017. Cover values for annual and perennial plant groups showed a high degree of variance between plots, as may be expected from an examination of plant guilds composed of many species.

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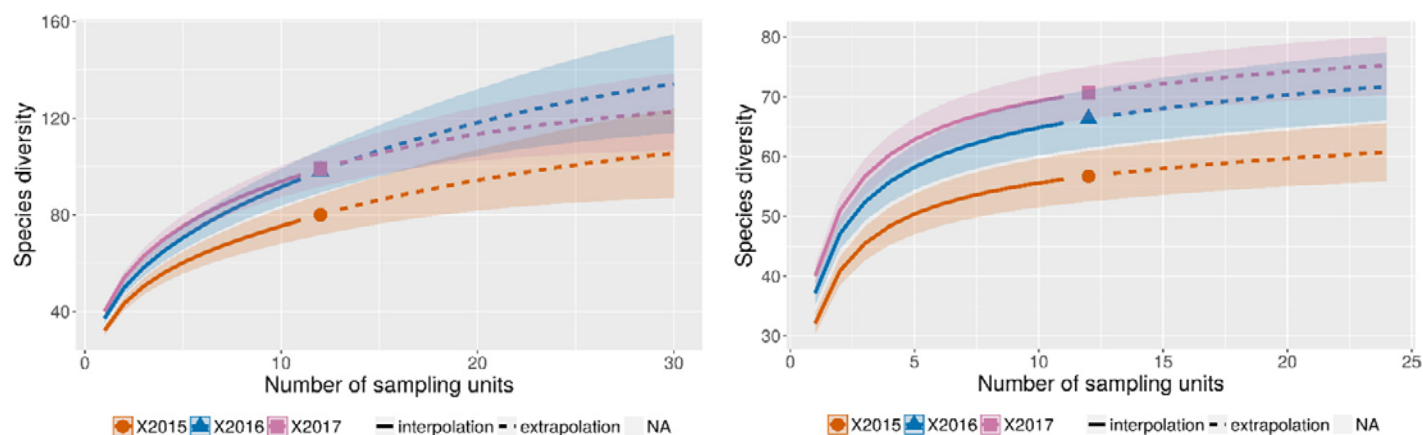


Figure 3. Rarefaction curves, showing effective species richness (left), and effective Shannon diversity (right) for the 7.32-m grassland plots sampled in 2015, 2016 and 2017. The curves show cumulative species diversity observed (*Sobs*) in the 12 plots sampled; interpolated below 12 samples, and extrapolated to an endpoint of 30 samples (95% confidence intervals shown in lighter colors, based on bootstrap methods with 200 replications).

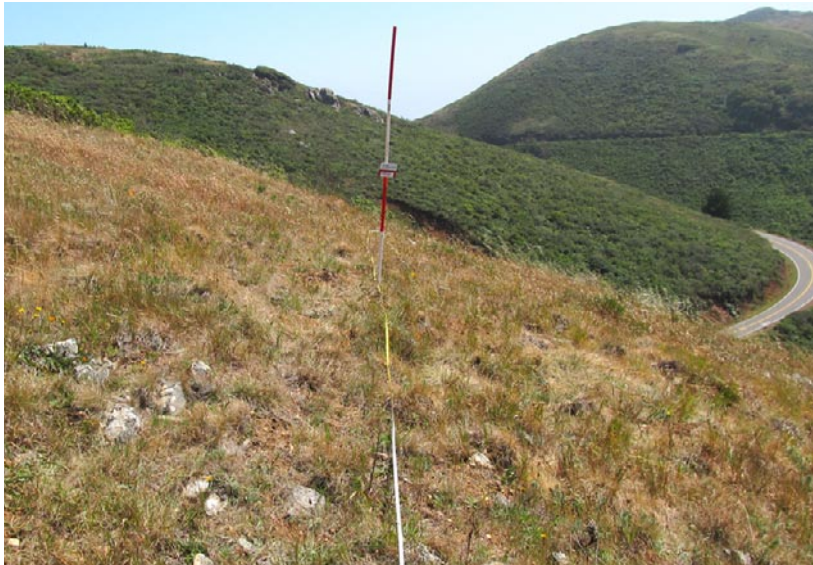


Figure 4. Time series of a coastal prairie monitoring transect. From the top: April 2015, during historic drought conditions, with sparse vegetative cover, bare ground and exposed bedrock; April, 2016, showing vegetation response to increased precipitation; April, 2017, showing dramatically increased vegetative cover in response to historically wet year, with little exposed bare ground or exposed bedrock.

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Alpha diversity, or site diversity in the 7.32-m radius plots, also varied significantly between dry and wet years. Effective species richness and diversity both increased from low points during the historic drought of 2015, with increased precipitation during 2016 and 2017 (Figure 4). This effect is likely due to numerous direct and indirect causal factors which could include, but are not limited to: favorable germination, seed production, and dispersal during wet years; dormancy of seeds or underground structures during drought, and dormancy release during wetter years; and plant-animal interactions. Increased cover of introduced annual grasses during wet years did not appear to have a negative effect on alpha diversity, which also showed a positive relationship with increased precipitation. However, a significant increase in diversity was seen between 2015 and 2016, the driest year and the first wet year, respectively, and diversity was not significantly different between the two wet years, 2016 and 2017, which could indicate that competitive interactions of the dominant annuals had a damping effect on overall diversity in the second consecutive wet year (see Corbin et al. 2007).

The results of this study support the hypothesis that the cover of annual species in bald hills prairies may be expected to fluctuate dramatically between wet and dry years, and that these fluctuations are quite variable by site. This study also showed that the cover of native perennial grasses and herbs also increased during wetter years, but more slowly. Moreover, we showed that alpha diversity in bald hills prairies of the Marin Headlands is subject to significant annual variation between wet and dry years, indicating that temporal flux in species richness must be accounted for in comparative studies of grassland biodiversity. We anticipate that longer-term annual sampling will better elucidate the vegetation responses to weather patterns and other environmental factors in these coastal prairies.

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