



VEGETATION RECOVERY AT POINT REYES NATIONAL SEASHORE TWO YEARS FOLLOWING THE 2020 WOODWARD FIRE

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EXECUTIVE SUMMARY

Fire can be both a positive and negative influence on native vegetation communities. Some species, such as Bishop Pine (*Pinus muricatus*), depend on fire for regeneration. Others, such as Douglas-fir (*Pseudotsuga menziesii*), are not reliant on fire for seedling establishment, but may survive and even germinate following low intensity wildfires. Increased fire severity and frequency are variables that influence the response of native vegetation to wildfire, potentially changing community species composition or even prompting conversion to other lifeforms (e.g., shrubland converting to grassland) or communities (e.g., Douglas-fir to Ceanothus scrub; Savage and Mast 2005, Syphard et al. 2018, Stevens-Rumann et al. 2018). In summer 2020, Point Reyes National Seashore (Seashore) experienced its largest fire in 25 years. The Woodward Fire burned 4,841 acres, primarily within the Phillip Burton Wilderness. Before this, the most recent large fire in the park was the October 1995 Vision Fire, which burned 12,354 acres. While about 2,000 acres on the western side of the Woodward Fire had previously burned in the Vision Fire, the majority of the affected landscape had not burned in at least 75 years. Fires of more than a few acres are relatively rare events at the Seashore, because rain and fog contribute to high moisture content in fuels. The Woodward Fire burned proportionately more forest and woodland communities and less shrubland than did the Vision Fire. Additionally, the Woodward Fire's effects were generally regarded as being more moderate, with fewer acres burning at a high severity.

To evaluate response of areas burned during the Woodward Fire, the Seashore implemented a field-based vegetation community recovery monitoring program in 2021-2022 that evaluated vegetation-related variables such as plant cover, species richness, and diversity, as well as seedling recruitment, basal regeneration, and plant damage (e.g., dead/damaged trees and shrubs). In 2022, approximately 35 permanent plots were established in the following communities: Hardwood Forest (8 plots), Douglas-fir (8 plots), Ceanothus (8 plots), Coastal Scrub (9 plots), and Bishop Pine (2 plots, not used in statistical analysis), with plots stratified to the extent possible to incorporate a range of fire frequency (e.g., once or twice burned) and fire severity based on BARC (Burned Area Reflectance Classification) ratings, which are derived from remote sensing.

In general, total vegetation cover two years following the Woodward Fire was quite high, even though the first two years of recovery coincided with a drought. Vegetation response differed more based on estimated burn severity (BARC) than habitat. Only Coastal Scrub included a relatively even number of plots in once- and twice-burned areas, but burn frequency did not appear to influence vegetation recovery in this habitat. Total and native plant cover and native species richness were higher in scorched/very lightly burned areas than moderately or highly burned areas. In contrast to findings of other studies (Crawford et al. 2001, Hunter et al. 2006, Alba et al. 2014), neither severity nor habitat affected variables related to non-native plant species, with the exception of thistle cover, which was elevated in areas with higher burn severity. The highest native species seedling recruitment was observed in Ceanothus plots followed by Hardwood Forest ones, whereas basal sprouting proved the dominant form of regeneration in Coastal Scrub plots.

Future monitoring efforts, both field- and GIS-based, will help to better inform about possible conversion to different vegetation communities or lifeforms.

INTRODUCTION

In summer 2020, Point Reyes National Seashore (Seashore) had its largest fire in 25 years. The Woodward Fire burned 4,841 acres*, primarily within the Phillip Burton Wilderness. The last large fire in the park was the October 1995 Vision Fire, which burned 12,354 acres. While about 2,000 acres on the western side of the Woodward Fire had previously burned in the Vision Fire (**Figures 1-2**), the majority of the affected landscape had not burned in at least 75 years. Fires of more than a few acres are relatively rare events at the Seashore. When the Vision Fire occurred, it was one of the largest wildfires recorded within the highly urbanized areas of the northern California coast, with more than 15% of the total park area burned (Forrestel et al. 2011). While the Vision and Woodward Fires have much in common in terms of some geographic overlap and general types of habitats impacted, there are differences, too. The Woodward Fire burned proportionately more forest and woodland and less shrubland than did the Vision Fire. Eighty-four percent of the area burned in the Vision Fire supported forest/woodland and shrubland (coastal scrub and ceanothus (*Ceanothus thyrsiflorus*)-dominated shrublands; **Figure 1; Figure 2**). Additionally, the Woodward Fire burned for almost a month, while the Vision Fire burned half its area in one day and was suppressed in five days. While the proportion of area that burned at high or moderate severity remained similar between the Vision and Woodward Fires remained somewhat similar (8% vs 9% and 14% vs 26%, respectively), the fact that the Vision Fire was

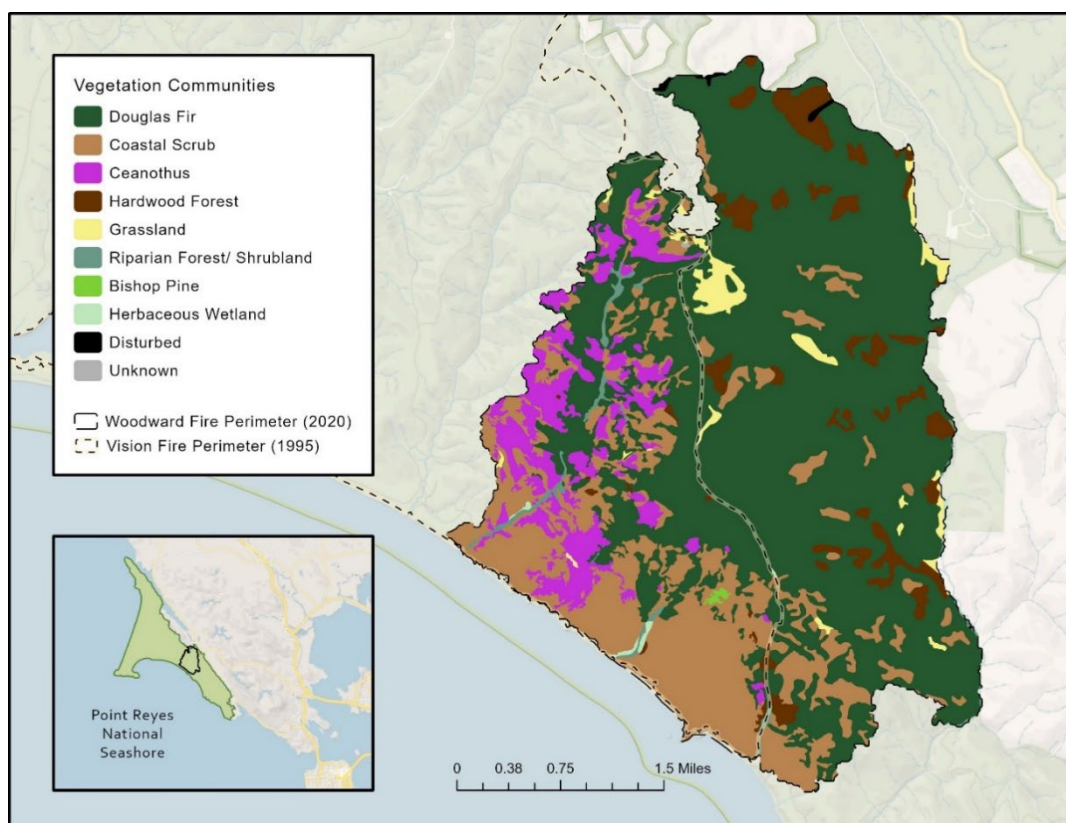


Figure 1. Vicinity and vegetation community map for Woodward Fire zone. The extent of overlap with the 1995 Vision Fire is also shown on the graphic. Vegetation communities represent post-Vision Fire communities mapped after 1995 (Forrestel et al. 2011).

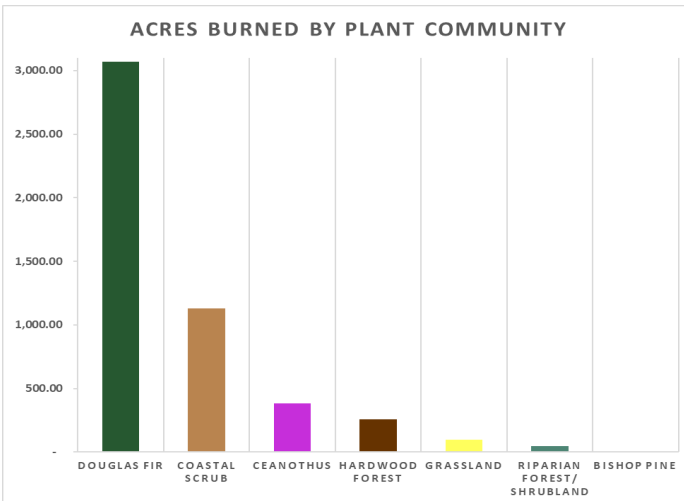


Figure 2. Distribution of burned acres in the Woodward Fire by broad plant communities where >1 acre/type burned.

almost twice the size translates into a larger total impact on the landscape. For the Woodward Fire, the remainder of the area burned at medium (37%) or low (29%) severity.

There are many factors that influence plant mortality and post fire recovery. Direct impacts of fires on vegetation such as plant mortality or scorch of tree canopies are considered first-order effects, while second-order effects related more to indirect effects of fire and are often more long-term and influenced by non-fire factors such as short- and long-term weather patterns or human activities. For example, the first winter following the

Vision Fire was a wet one, while the first year following the Woodward Fire occurred during a drought. These shorter-term climatic trends can alter vegetation responses following fire, including how fast both native and non-native invasive species re-establish with the burned area. These shorter-term climatic fluctuations in vegetation are juxtaposed against much longer-term climatic trends, which are also highly influential in shaping vegetation communities. Paleo-ecological studies conducted at Point Reyes point to longer-term climatic cycles determining in large part the vegetation community mosaic of forests, shrubland, and grassland present on the landscape at any particular point in time. Forests and coyotebrush (*Baccharis pilularis*)-dominated coastal scrub expanded during wetter geologic periods (late Pleistocene; mid-to-late Holocene) and contracted during drier ones (early Holocene), when pines (*Pinus* spp.), coastal sage (*Artemisia californica*)-dominated shrublands, and grasslands became more dominant (Rypins et al. 1989, Barron et al. 2003, Anderson et al. 2005, Anderson et al. 2013). With a shift back to a cooler climatic regime in the middle to late Holocene, more mesic vegetation established, including the Douglas-fir (*Pseudotsuga menziesii*)-mixed conifer-hardwood forests still present in Point Reyes today (Anderson et al. 2013), as well as coast redwood (*Sequoia sempervirens*) and tanoak (*Notholithocarpus densiflorus*), particularly near Wildcat Lake, Glenmire, and even Coast Trail Pond (Anderson 2005).

The role of fire in shaping vegetation communities at Point Reyes has intensified in more recent geologic times. Charcoal influx in sediment layers at Point Reyes predating 4,000 years before present (BP) was minimal, however, small amounts of charcoal were deposited consistently in sediments between 4,000 and 2,200 years BP, with a higher influx after 2,540 years BP (Anderson et al. 2013). Increasing charcoal deposition during this time period may be indicative of the use of fire by the prehistoric communities related to the Coast Miwok, which populated the area during this period (Anderson et al. 2013). Native peoples in California actively used burning to manage vegetation communities (Keeley 2005, Lorimer et al. 2009, Anderson et al. 2013), with Coastal Miwok believed to burn coastal prairie, scrub, foothill woodland, and grassland vegetation to promote abundance of certain species for harvesting (Duncan 1992). Charcoal influx increased substantially in the period between 2,200 and 1,000 years BP, when Douglas-fir, coast redwood, and tanoak reached their maximum expansion (Anderson et al. 2013). In fact, spikes in charcoal within sediments dated to this period strongly overlapped with increases in

pollen of either coast redwood or Douglas-fir (Anderson et al. 2013). Moreover, a significant decline by almost an order of magnitude in charcoal deposition around 1,000 years BP coincided with a decline in Douglas-fir pollen (Anderson et al. 2013). Anderson and his colleagues argue that the pollen and charcoal stratigraphies at Glenmire suggest a tight coupling of fire history and vegetation changes (Anderson et al. 2013), which was perhaps superimposed on a broader scale of vegetation changes associated with long-term climatic cycles.

Shifts in vegetation in Point Reyes imposed by climate and even fire were overshadowed by the unprecedented level of anthropogenic changes that accompanied Euro-American settlement, including introduction of livestock grazing and non-native plants, clearing and eventual logging of forests, and fire suppression (Duncan 1992, Anderson et al. 2013). The use of fire by Coast Miwok to manage vegetation decreased dramatically due to missionization of native peoples and efforts by the Spanish authorities to suppress fires and thereby “uproot this very harmful practice of setting fire to pasture lands” (Clar 1959). These widespread disturbances coincided with a steep decline in the mid-1800’s in conifer pollen, particularly Douglas-fir and coast redwood (Anderson 2005, Anderson et al. 2013), which at that point covered much of the east slope of the Peninsula along with Bishop pine (*Pinus muricata*; Livingston 1994 in Anderson et al. 2013). Coastal scrub and chaparral species declined, as well (Anderson 2005).

Despite efforts to suppress prescribed burning by native peoples, analysis of fire scars found that, between the 1820’s and the mid- 1900’s, mean fire intervals remained high in Point Reyes, with reconstructed mean surface fire intervals ranging from 7.7 to 8.5 years (Brown et al. 1999). Fires generally occurred late in the growing season or after growth had ceased for a year (Brown et al. 1999). These mean fire intervals (MFI’s) actually correspond well with those estimated for earlier periods (1450-1850) using fire scarring on coast redwoods on the Bolinas Ridge (8- 20 years; Finney 1990 in Brown et al. 1999) and may potentially approximate the pre-settlement surface fire regime on the Peninsula (Anderson et al. 2013). Most of these fires are believed to have anthropogenic origins (Brown et al. 1999). Since the 1850’s, however, MFI have risen exponentially, with return intervals in the last century averaging 37.6 years (1917-2020; Dawson 2021).

In considering interactions between fire and vegetation, it is definitely still important to consider the degree to which an area is departed from historic conditions. It is difficult to determine what the natural fire regime of coastal California would be absent intervention by humans, because this area has been subject to prescribed burning for millennia. Non-human ignition of wildfires in coastal California environments is believed to be infrequent, with lightning strike frequency estimated currently at approximately 4 strikes per century near Santa Cruz (Striplen 2014) or 3/100km²/year on the central coast (van Wagtendonk and Cayan 2008). However, despite the infrequency of natural fire, fire suppression in the past century has affected vegetation, because certain vegetative communities occurring in Point Reyes have evolved with fire, and certain species may depend on fire for their continued presence. Both Bishop pine and *Ceanothus* depend on fire for reproduction. As with many areas of the western U.S., for many decades, fire suppression activities shaped vegetation communities by favoring non-fire-dependent vegetation communities, with the spatial extent of fire-dependent communities potentially contracting over time due to lack of regeneration.

The Vision Fire reversed some of these successional trends. Some dramatic landscape-level shifts in vegetation types were documented after the large 1995 wildfire (Forrestel et al. 2011). *Ceanothus*-dominated maritime chaparral – which was a minor vegetation community component prior to the

Vision Fire -- expanded substantially (4,642%) in areas following the fire (Forrestel et al. 2011). Bishop pine communities also expanded noticeably (85%; Forrestel et al. 2011). Conversely, acreages of Douglas-fir, grassland, and coastal scrub dropped considerably, with reductions in extent ranging from 27% (coastal scrub) to 46% (Douglas-fir; Forrestel et al. 2011). Much of the new Bishop pine present post-fire was growing in areas that were formerly coastal scrub (Forrestel et al. 2011). Ceanothus also made significant incursions into areas of former coastal scrub, as well as of Douglas fir and Bishop pine forest (Forrestel et al. 2011; **Figure 3**). Evaluation of predictor variables associated with so-called “type conversion” – conversion of one type of habitat to another – varied for the fire-dependent species. For Bishop pine, distance from pre-fire Bishop pine stands and pre-fire vegetation type were significant predictors of post-fire Bishop pine presence or absence (Forrestel et al. 2011). In contrast, the probability of Ceanothus occurrence increased with higher burn severity and slope and decreased with higher elevation (Forrestel et al. 2011).

Since the Vision Fire occurred, the wildfire landscape in California – and even the world – has changed dramatically. Wildfires have become more frequent, larger, and more severe, fueled by the increasing unpredictable climatic conditions associated with climate change (US EPA 2021). According to the National Oceanic and Atmospheric Administration, between 1980 and 2020, the United States had 18 wildfire events that caused more than \$1 billion in damage: 15 of those have occurred since 2000 (NOAA 2021). Multiple studies have found that climate change has already led to an increase in wildfire season length, wildfire frequency, and burned area (USGCRP 2018, Westerling et al. 2006, Westerling 2016). The wildfire season has lengthened in many areas due to factors including warmer springs, longer summer dry seasons, and drier soils and vegetation (USGCRP 2018). Similarly, climate change threatens to further increase the frequency, extent, and severity of effects of fires through increased temperatures and drought (USGCRP 2017).

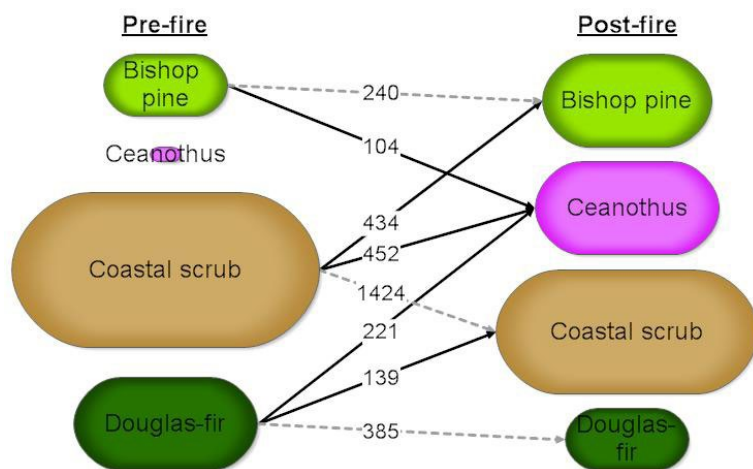


Figure 3. Transitions among major vegetation types following the 1995 Vision Fire. The size of each ellipse is proportional to the extent of that community within the study area. Solid arrows indicate transitions between vegetation communities, and dashed gray arrows indicate areas where the vegetation type remained the same following fire. Arrows are labeled with the number of hectares that either transitioned or stayed the same. A minimum 100 ha threshold was used for inclusion in the diagram. From Forrestel et al. (2011).

These climatic shifts have made vegetation recovery following wildfires even more unpredictable. One study that evaluated post-fire vegetation recovery in North American temperate forest ecosystems with Landsat time series analysis found that “post-fire climate anomalies” (unpredictable weather patterns) were the strongest predictors (30- 41%) of vegetation recovery, while pre-fire Normalized Burn Ratio (NBR) – essentially equivalent to pre-fire green vegetation cover – and burn severity were a distant

second (2- 6%; Bright et al. 2019). Numerous other studies conducted over the last several decades in habitats across the world that analyzed post-fire vegetation recovery also cited climate, drought, El Niño, and temperature as critical factors that impact post-fire vegetation recovery (Liu et al. 2016, Levin 2019, Kibler et al. 2019, Huang et al. 2020, Meneses 2021). A study conducted by a team at Colorado State University on 1,500 sites in the conifer forests of the U.S. Rocky Mountains that had experienced 52 wildfires between 1988 and 2011 found that the proportion of sites with no regrowth almost doubled after 2000, from 19% to 32%, coinciding with increasing temperatures and more droughts (Stevens-Rumann et al. 2018). On a regional landscape level, what this means is that, during drought or low-rainfall years, those sites that were often the driest to begin with due to aspect (south-facing) or elevation (lower) may have a greatly increased potential for more severe fires (Huang et al. 2020) and reduced chances of recovery (Hibbs et al. 2011; Potter 2016; Levin 2019; Kibler et al. 2019; Stevens-Rumann et al. 2018). Certain habitats or forest types appeared to recover more quickly than others, with Ponderosa pine forest (Bright et al. 2019) and chaparral (Kibler et al. 2019) among the slowest and shrubland (Meneses et al. 2021) and coast redwood (*Sequoia sempervirens*; Lazzeri-Aerts and Russell 2014) among the fastest to recover. Several studies noted that, for those habitats that appeared to be recovering, signs of recovery were apparent very quickly, often within 5 years (Potter 2016, Bright et al. 2019; Stevens-Rumann et al. 2018)

In recent decades, most researchers conducting post-fire vegetation recovery have favored use of Landsat imagery to evaluate pre- and post-fire changes in NBR. The NBR is an index designed to highlight burnt areas and assign severity ratings such as low, moderate, and high using both near infrared (NIR) and shortwave infrared (SWIR) wavelengths (United Nations (UN) 2021). NBR has also been used in post-fire to assess vegetation recovery, as healthy vegetation shows a very high reflectance in the NIR and low reflectance in the SWIR portion of the spectrum (UN 2021). A very large proportion of recent studies published on post-fire vegetation recovery – including many cited earlier – rely on this approach. However, generally NBR would seemingly be better to address whether some type of ground cover currently exists and less capable of determining whether that ground cover represents the habitat that existed in that area previously or a new type of habitat or the quality of that post-fire habitat. Another important note is that while Burn Area Reflectance Classification measures the apparent fire effects to vegetation, it does not accurately measure burn severity in soils, which impact vegetation indirectly for the long term. At least one study combined remote sensing analysis with collection of data on the ground, using both NBR and Green Vegetation (GV) fraction from spectral mixture analysis (Kibler et al. 2019). In chaparral areas burned by Santa Barbara's Zaca Fire in 2007, GV fractions increased rapidly in the four years after the fire, but the recovery of a dominant native plant species, *Ceanothus leucodermis*, lagged behind that of the other species (Kibler et al. 2019).

The results from the Kibler et al. (2019) study illustrate the challenges of trying to use remote sensing generated from space to track fine-scale changes in vegetation recovery post-fire. Whether or not some type of vegetation takes hold following fire is one question and may be pertinent about the potential for post-fire erosion. However, most of the interest from a vegetation ecology perspective is how fire changes – or not – vegetation communities that were present previously or that may be desired habitats. This may involve general changes in lifeform (grass, shrub, tree) or more specific changes related to vegetation communities (Douglas-fir forests, Ponderosa pine forests, oak woodland, coastal scrub). Some of the climate change-related shifts in fire ecology strongly relate to changes in life form or vegetation community. For example, an increase in the frequency of fire in southern California chaparral

habitats is spurring either a type conversion (into coastal sage scrub) or a life form conversion (into grassland), with areas having the most frequent fires typically converting to grassland (Syphard et al. 2018). Forests in the hottest, driest regions in the Rocky Mountains that were already at the edge of their climatic tolerance were most susceptible to lifeform conversion, converting to grassland or shrublands (Savage and Mast 2005, Stevens-Rumann et al. 2018).

Habitats along California's immediate coast may be buffered to some extent from the brunt of the climate change-associated warming predicted for inland areas of California due to the moderating influence of the ocean. While climate change models for California generally predict a drier and warmer climate (Polade et al. 2017), other models suggest that certain portions of coastal California come become cooler, windier, and wetter (Lebassi et al. 2009, Maloney et al. 2014, Pozo Buil et al. 2021), leaving a high degree of uncertainty about how coastal areas might be impacted. Cooler temperatures along the coast, coupled with warming of interior regions, would seemingly favor continuation of the fog "belt" that typically cloaks the coast during the summer when temperatures soar in inland areas. Fog blankets the Point Reyes coast about 30% of the time between June and October (Goodman 1982 *in* Ingraham and Matthews 1995), generating anywhere from 1.0 to 5.2 L site⁻¹ yr⁻¹ of additional moisture during what is otherwise California's driest period (June-Oct, Ingraham and Matthews 1995). Unfortunately, several studies have shown a decline in fog over the past century (Johnstone and Dawson 2010, O'Brien 2011) that is projected to continue (O'Brien 2011).

Cool temperatures, high winds, and fog are important drivers of vegetation community structure along California's coast. Perhaps due to these complex climatic interactions, ecosystems of coastal northern California have been documented to contain high levels of both alpha and beta diversity (Whittaker 1972, Barbour et al. 2007, Thorne et al. 2009, Kraft et al. 2010 *in* Forrestel et al. 2011). It is estimated that the Seashore supports over 900 species of flowering plants, representing approximately 15.3% of California's plant species. Of these more than 900 species, 48 plants have been classified as rare by the federal or state government or the California Native Plant Society (CNPS), with four (4) of them considered endemic to the Point Reyes Peninsula (NPS 2018, Howell et al. 2007). As a result of the high diversity of this system, the response of plant communities to fire may be more complex than what has been documented in other more homogenous vegetation types (Forrestel et al. 2011). The response might also differ from those reported in recent decades for more inland areas of California and the western U.S. due to the more complex climatic influences experienced along the coast, as well as differing from that of Vision Fire due to climatic shifts associated with climate change.

While it is evident that the effects of fire on coniferous and pine forests, chaparral, and shrublands have been relatively well-studied, Forrestel et al. (2011) pointed out that landscape-level effects of wildfires in California's coastal regions have received less research attention (Callaway and Davis 1993), although there have been a few more studies in recent years (Lazzeri-Aerts and Russell 2014, Potter 2016, Huang et al. 2020). Examining effects of the Woodward Fire on vegetation can help managers understand how fire effects the distribution of plant communities across the landscape in order to anticipate future changes, including adverse impacts, and potentially plan appropriate management. Park resources potentially affected by fire – and fire suppression activities – include sensitive and rare flora and fauna; natural ecosystem processes; invasive plant species; recreation; and park infrastructure. Undesirable changes in ecosystem processes, rare plant and wildlife species, invasive plants, cultural and archeological resources, aquatic ecosystems and water resources, soils and fuel loads could prompt the park to take management action, should park staff deem these changes to be a critical threat to

protection of preserving park resources and values unimpaired (NPS 2006). It should be noted that unlike many parks and federal lands in the Sierra Nevada mountains, the Seashore has not actively managed fuel loads at the landscape scale, with some minor exceptions that included small-scale prescribed burning. However, understanding vegetation recovery trajectories across burn severity units and under various habitat, environmental, topographic, and substrate conditions may inform responsible use of prescribed fire on this landscape in the future, and it may also inform where to prioritize management actions when future fires occur.

The Seashore developed a post-fire vegetation recovery monitoring plan in 2021-2022 in compliance with both NPS Management Policy (2006) and Fire Management Plan (FMP; 2007) directives to monitor changes following wildfires. The monitoring plan has two broad objectives. The first is to generally monitor changes in vegetation following the Woodward Fire. In keeping with NPS Management Policy, the emphasis is on detecting and understanding changes in park natural resources. This monitoring will inform both park staff and the general public on how wildfires can transform vegetation communities within the park. The second objective is to specifically monitor variables where, in accordance with NPS Management Policy, management actions might potentially be taken should “unacceptable adverse impacts to resources” be detected. Variables that would fall under this objective include establishment of non-native invasive weeds in both areas that burned and areas where suppression activities such as dozer lines and handlines were constructed. Most of the detailed surveying and monitoring of non-native, invasive plants are being conducted using a separate protocol funded by BAER/BAR funding (Hendrickson et al. 2021), but, to understand wildfire changes to the park landscape, vegetation recovery monitoring also needs to assess presence of non-native invasive species which fall into the category of second-order or indirect fire effects. Also, monitoring is also focusing on detection of rare plant species, with the idea that these might need to be protected from adverse effects associated with the fire.

The first level of monitoring will be a 10-year program to assess lifeform and type conversion of habitats within the Woodward Fire area, along with evaluating species composition and regeneration, as well as the effects of fire on vegetation. While many research-related vegetation recovery monitoring programs have relied on remote sensing to determine recovery trajectories, vegetation community mapping in the park conducted in both 1994 and 2018 utilized photointerpretation of aerial imagery coupled with ground-based accuracy assessment, with most communities mapped to at least an Alliance level and, in some cases, the more detailed Association level. Follow-up vegetation mapping conducted nine (9) years after Point Reyes’ Vision Fire by Forrestel et al. (2011) lumped some of these Alliances into broad categories, including Hardwood Forests, Bishop Pine forests, Douglas-fir forests, Ceanothus scrub, Coastal scrub, Riparian, Herbaceous Wetland, and Grassland. To determine lifeform and type conversion, PORE will reinitiate mapping of the Woodward Fire areas 5- to 7 years post-fire using the 2018 fine-scale vegetation community map as a base layer on high-resolution imagery either acquired specifically for the purpose or, more likely, from publicly available sources such as the County of Marin (MarinMap).

The second level or tier of monitoring is plot-based, because this type of information can generally not be reliably obtained from aerial imagery. Because there are no widely distributed pre-fire monitoring plots within the Woodward Fire area, we cannot directly assess changes in species composition or regeneration pre- and post-fire. However, we can potentially include control or reference from similar, nearby habitats that were not burned during the fire for comparison, although habitat diversity within Point Reyes makes it difficult often to find areas with identical management and disturbance histories

and environmental conditions. We are focusing our plot-based monitoring efforts on the following communities: Douglas-fir, Hardwood Forest, Coastal Scrub, and Ceanothus Scrub. Bishop pine was not formally included in statistical analyses, because very few mapped Bishop pine stands burned during the Woodward Fire: It is included in summary data tables and graphs, though. Plot-based monitoring is assessing three components: effect of fire on pre-existing vegetation; species composition of burned areas post-fire; and species regeneration post-fire. These variables were assessed in plots that incorporate different burn severity (Light-1/Medium-2 or Heavy-3/Extreme-4) and, when possible, burn frequency (single vs. double; **Figure 4**).

This report details results of plot-based monitoring from 2022, two years following the Woodward Fire.

MATERIALS & METHODS

PILOT MONITORING PROJECT: A pilot program using the Fire Monitoring Handbook (FMH) approach was initiated for the Woodward Fire area in 2021 with some modifications, including changing the plot size and shape for forested areas to circular plots with a 15-m. radius that utilized only two (2), not four (4), Brown's Fuel Lines transects. It also modified field interpretation of burn severity to mirror more closely the approach utilized by Santa Monica Mountains National Recreation Area (W. Rehlaender, NPS, *pers. comm.*). Of the 36 plots tentatively planned for eventual establishment, only nine (9) plots were installed as part of the pilot program due to staffing constraints and the time-consuming nature of traveling to plots and installing and reading them, with each plot taking on average 1 full day per plot (range = 0.5- to 1 day; Appendix A).

CURRENT MONITORING PROTOCOL: Following completion of the pilot project program, park staff had concerns about whether this approach would be too time-consuming to allow for effective replication, so, in winter 2021, park staff reevaluated the field-based monitoring approach to allow for a larger number of plots to be installed and monitored given the constraints on staff resources. The current field-based monitoring approach represents a hybrid of several methodologies, although it most closely parallels a rapid assessment (RA) protocol developed by California Department of Fish and Wildlife in conjunction with CNPS (CDFW-CNPS RA). The CDFW-CNPS protocol combines the Relevé and Rapid Assessment (RA) vegetation sampling protocol with post-fire vegetation severity assessment (CDFW-CNPS 2020, v. 8 November 30, 2020). The relevé approach involves demarcating a plot with a measuring tape, and each species in the plot is recorded along with its cover (CDFW-CNPS 2020). The rapid assessment (RA) approach is not based on a taped plot, but is based on a visually estimated, usually circular area within a representative portion of the entire stand, with up to 20 of the dominant or characteristic species and their cover values recorded (CDFW-CNPS 2020). RA's are typically performed in woody vegetation areas, and relevés are used in herbaceous vegetation (CDFW-CNPS 2020). The recommended radius for a RA ranges from 10 m for small stands or uniformly burned herbaceous communities to up to 30 m for larger stands of woody vegetation (CDFW-CNPS 2020).

Park staff carefully evaluated which variables would provide the best information regarding vegetation recovery, as well as those applicable to management priorities and which would provide the most information about changes in vegetation communities following the Woodward Fire. These variables were lumped into three general categories: 1) Effects of fire on pre-fire vegetation communities and substrate; 2) Species composition of post-fire vegetation communities; and 3) Species regeneration within post-fire vegetation communities. Other variables that are often included in plot-based fire

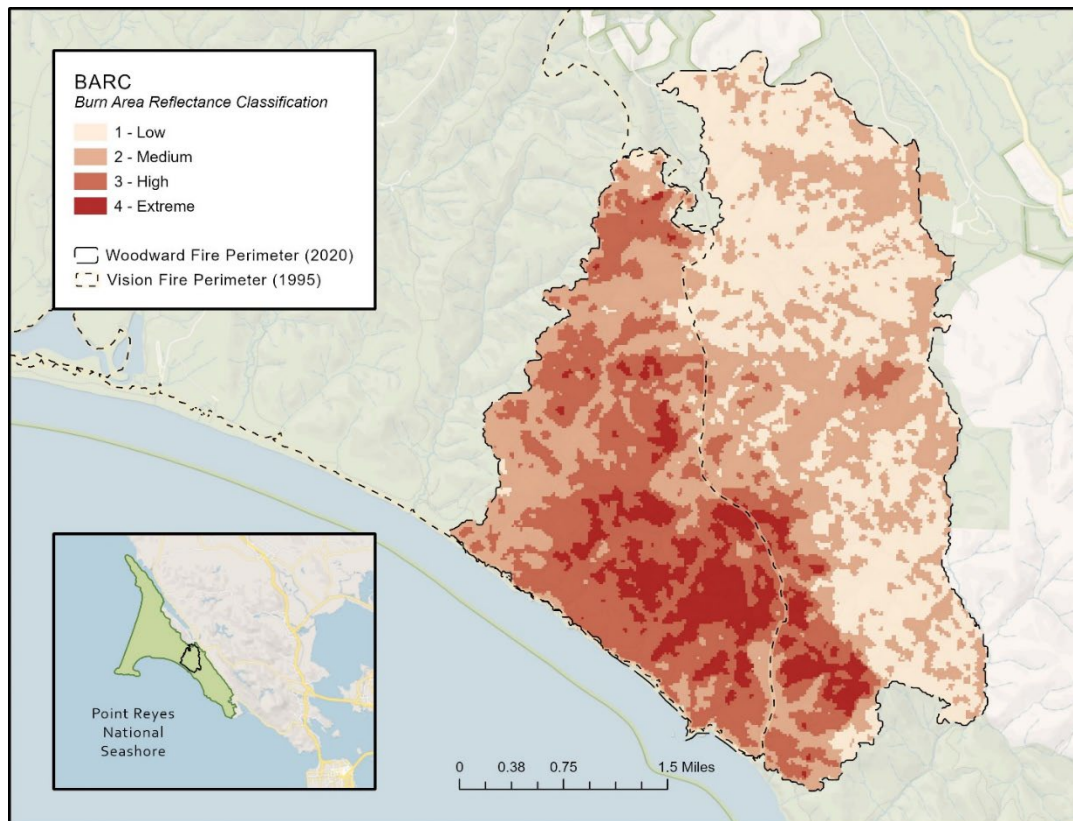


Figure 4. Burn severity ratings based on burned area reflectance classification (BARC). 1=Unburned/Scorched; 2=Low; 3=Moderate; and 4=High. More of the Moderate and High-rated fire intensity occurred in areas that were also burned during the 1995 Vision Fire.

monitoring programs were not incorporated, including those having to do with canopy height or strata height or Brown's fuel transects. As noted earlier, the park has not previously conducted fuels management, so the immediate applicability and value of collecting this data, which is typically quite time-consuming, was unclear.

PLOT SELECTION: In 2022, approximately 35 permanent plots were established in the following communities: Hardwood Forest (8 plots): Douglas-fir (8 plots); Ceanothus (8 plots); and Coastal Scrub (9 plots; Appendix A). Two additional plots were placed in areas that turned out to support Bishop pine. Those plots are shown in charts below but are not included in the analysis. Plots are demarcated with a flagged rebar in the center of the plot with the location documented using GPS. Factors that were used to determine their locations included: burn severity, burn frequency, accessibility, distance from disturbed areas and other plots, and slope. For plot selection, burn severity was based on the Burned Area Reflectance (BARC) derived from remote sensing using satellite imagery, with four categories assigned: Unburned, Low, Moderate, and High. It should be noted that, as BARC are derived from imagery, and not on-the-ground surveys, BARC ratings are not always accurate. At Point Reyes, areas classified as Unburned often included very lightly burned, as well as unburned. Some of these lightly burned areas may have incorporated areas that were back-burned as part of fire suppression.

It would require too many plots to replicate plots of every community at every fire severity level, and so we combined fire severity into two categories – Moderate/High and Unburned/Low – and used those to

stratify sampling efforts. We further stratified on burn frequency, where possible (i.e., burned once in Woodward Fire or burned twice in Vision and Woodward Fires). Certain communities in the Woodward Fire area only had one burn frequency level: for example, all of the Ceanothus scrub and Bishop pine had burned twice, while all of the Hardwood Forest only burned once. For communities in which both burn frequency classes occurred, plots were generally allotted equally for each burn frequency within each burn severity level (e.g., Moderate/High, Unburned/Low).

Potential locations for permanent plots were further stratified or filtered by a set of selection constraints. Vegetation communities were first identified by overlapping the 1994 vegetation map, adjusted in 2011 by Alison Forrestel to reflect changed conditions in the Vision Fire area, and the 2018 vegetation map. Sampling locations were then refined by focusing on areas within the selected vegetation community types that 1) have a slope < 45%, 2) are located > 10 m, but < 100 m, from trails, fire roads, dozer/hand lines, and 3) are more than 50 m from any known I&M Plant Community or research plot.

While not optimal from a statistical randomization standpoint, the decision to locate plots within areas along trails in the Woodward Fire area was made because the park needs to maximize the number of plots that can be monitored with limited staffing resources, and navigating to plots that are some distance from fire roads and trails can be considerably time-consuming, particularly when dense scrub and forest vegetation regrows. Fortunately, the trail network is well-balanced spatially across the Woodward Fire burn area. Plots were established outside of the disturbance zones associated with trails, with zones for singletrack and hand lines expected to be smaller than those associated with fire roads and dozer lines. Because spatial autocorrelation among forest plots has been found for distances to 200 m (Forrestel 2011), plots of the same vegetation community types, particularly forest ones, were located at least 200 m away from each other.

SAMPLING METHODOLOGY: As noted earlier, the revised methodology leans heavily on that developed by CNPS-CDFW for post-fire monitoring (CNPS-CDFW 2020), although there are some modifications. Sampling is generally conducted when the largest number of species have germinated and are usually identifiable, which is generally May to June, though it may be adjusted annually based on climatic and phenological conditions. In Year 2, many plants had already established, and even spring-flowering plants were already flowering in January 2022 due probably to the copious amount of fall and winter rainfall and some warm weeks, so monitoring was conducted starting in December 2021 through May 2022.

All of the plots are circular and have a 15-m radius. A transect tape is rolled out to the 15 m directly due north of the rebar. Monitoring staff walked around the circular plot and compiled a comprehensive list of all plant species present in the plot using the four- or six-letter plant code established by the USDA Natural Resources Conservation Service. Both information on Lifeform (i.e., is the species classified as a Tree, Shrub, Vine, Herb, or Grass?) and Growth Form (i.e., does the species occur in the Tree, Shrub, Herb, or Ground strata?) was collected (**Figure 5**). If a species occurs in multiple strata, its presence was recorded for each stratum. If a species was dead or damaged, but still present, it was also recorded as the species name with the initials (D) for “Dead” or (DAM) for “Damaged” appended to the species code. Bareground (BG) – or areas without vascular or non-vascular species – that present as mineral soils or soils with fine duff layer was also assessed. Areas with no vascular or non-vascular species that have a rocky substrate were characterized as “Rock,” and areas with downed logs or larger woody debris were

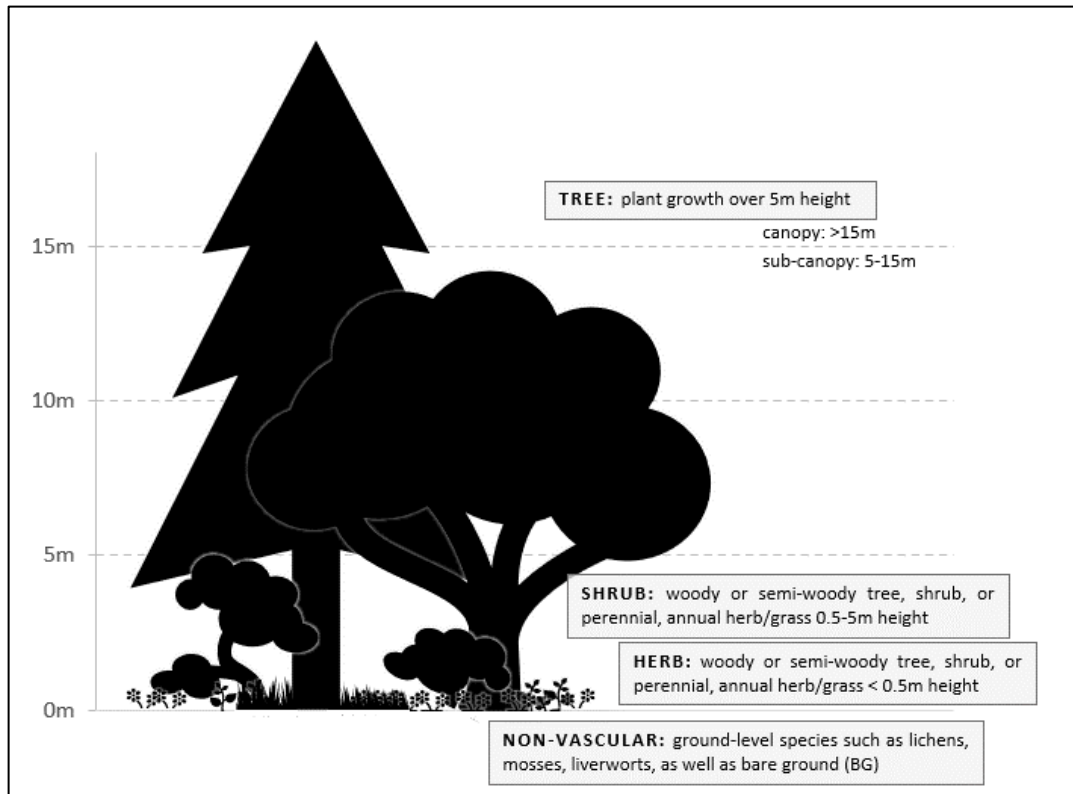


Figure 5. Vegetation strata as defined by Nature Serve.

classified as “LWD.” In the database, plants were classified as either native or non-native and were assigned a life form classification based on Jepson Manual Ver. 2 (Baldwin et al. 2012).

Once a comprehensive plant list has been generated, monitoring staff ocularly estimated percent cover for each species in each stratum. The final value is an average between the two observers’ estimates. For forested plots, overstory canopy was either assessed ocularly or by using a spherical densiometer. Staff also assessed regeneration by noting the presence of seedlings (Y/N) and resprouting individuals (Y/N).

In forested plots, an inner circle of 5 m was established around the plot center-point, and all trees within this smaller area or sub-plot were assessed or measured for the following characteristics: Alive (Undamaged)/Dead/Damaged, DBH, Tree Height, Maximum Scorch Height, Maximum Chair Height, and Regeneration via resprouting. A DBH tape was used to measure DBH for trees. A rangefinder was used to measure Tree Height and Maximum Scorch and Char Heights for each tree assessed. The scorch and char quantitative assessments derive from the CNPS-CDFW methodology.

In shrubland plots, an inner circle of 5 m was established around the plot center-point, and all shrubs within this smaller area will be assessed or measured for the following characteristics: Alive (undamaged)/Dead/Damaged, Dripline Diameter, Percentage of Shrub Scorched (for Damaged Shrubs), and Regeneration via resprouting. The diameter of the shrub was measured across its widest canopy diameter at canopy dripline or at the outmost edges of the shrub canopy, including both live and dead foliage for living shrubs.

In all plots, a 1-m belt transect was conducted to quantitatively assess regeneration. Starting at the plot center-point, a 1-m belt on the right side of the north-oriented transect line was conducted, and every seedling of trees (Douglas-fir, Bishop pine, California bay, etc.) or shrubs (Ceanothus, coyotebrush, red elderberry/*Sambucus racemosa*) was counted, and its height, measured. If the number of seedlings exceeded 25 of one species, measurements of average height of seedlings was taken at 0.5 m intervals.

Based on the CNPS-CDFW methodology, substrate and Vegetation Burn Severity Assessments are conducted in each of the four (4) quadrants of the 15-m plot. It should be noted that Substrate Assessment is often only assessed in the first year after fire, as it is difficult to evaluate burn severity and fire effects to litter, duff, logs, and soils once vegetation starts re-establishing. However, the impacts of fire to existing vegetation are often more persistent and can be monitored for a number of years following fire depending on how quickly some of the damaged vegetation begins to degrade and decompose. Unlike CNPS-CDFW, the park is directly using the Burn Severity Coding Matrix from the FMH (USDI NPS 2003), as it separates out qualitative assessment based on whether the vegetation community is Forest, Shrubland, or Grassland. If the burn severity of vegetation within a quadrant varies, then the assessment characterizes how much burn severity classification is present within each quadrant and then totals and averages the burn severity for the entire plot. Burn severity classifications are as follows: 1-Unburned; 2=Scorched; 3=Lightly Burned; 4=Moderately Burned; 5=Heavily Burned.

PHOTOPOINTS: Photopoints were installed at 61 locations along trails, dozer lines, and in one retardant zone between August 28, 2020, and January 29, 2021 (Appendix B). Locations were chosen to provide useful vistas for observational documentation of vegetation change. While some plots include mixed vegetation type or burn severity, the general distribution is: 5 Forb/grassland (all Low or Moderate burn severity); 35 Forest (25 Low to Moderate; 7 High severity; 3 either on dozer line or not within severity mapping); 8 Shrub/Forest (all High to Extreme severity); and 13 Shrub (5 Low/Moderate; 6 Moderate/High; 2 Extreme severity). Plots were photographed a second time March 16 – May 26, 2021, and again between March and July 2022.

STATISTICAL ANALYSIS: Information collected will be entered in Excel databases developed for each community type. From this, summary data tables were generated (Tables 2-3). We performed both multivariate and univariate statistical analyses in R (R Core Team 2021; version 4.1.3) using habitat (categorical) and BARC fire severity rating (continuous) as fixed effects. On-the-ground adjusted FMH severity rating (continuous variable) was employed, because it was considered potentially more accurate than the categorical BARC ratings derived from aerial imagery. As group dispersion between habitats was appropriately homogenous (evaluated using betadisper function), we conducted Permutational Multivariate Analysis of Variance (Permanova; vegan package; Okansen et al. 2022). For univariate analyses, assumptions for parametric statistics were evaluated (normality, homogeneity of variance) and, if data did not meet these assumptions, we analyzed using generalized linear model techniques employing other techniques for normalizing data (e.g., bestNormalize package; Peterson 2022). The final model was based on residual plots, Akaike information criteria (AIC) value, and data dispersion. When data did not meet parametric assumptions even with normalizing techniques, we conducted the non-parametric randomization approach Permutation (lmerPerm 2.1.0; Wheeler & Torchiano 2016). Post-hoc pairwise differences were assessed using Tukey HSD, with the emmeans package (Lenth et al. 2022) used for GLM analyses.

RESULTS

In general, total vegetation cover two years following the Woodward Fire was quite high, even though the first two years of recovery occurred during a drought. Total vegetation cover averaged 105.3 (± 15.7 (S.E.))% for Hardwood Forests and 126.3 (± 9.7)% for Coastal Scrub, with cover exceeding 100% due to layering of species within plots (**Table 1**). Vegetation cover, richness, and diversity variables differed based on estimated burn severity (BARC; $df=1$, $F=3.7$, $p=0.02$,) but not seemingly habitat ($df=3$, $F=1.6$, $p=0.13$; Permanova, method=Bray, 999 perm.; betadisper, $F=0.53$, $p=0.67$), although this model accounted for only a modest amount of the variation ($r^2=0.23$). As noted earlier, despite efforts to stratify by fire frequency, most of the habitats had either largely burned just once (e.g., Hardwood Forest, Douglas-fir) or twice (e.g., Ceanothus). Only Coastal Scrub had plots equally distributed between once- and twice-burned areas. Vegetation cover, richness, and diversity variables, however, showed no variation based on burn frequency in Coastal Scrub plots (Permanova, method=Bray, 999 perm., $df=1$, $F=0.56$, $p=0.79$, $r^2=0.06$; betadisper, $F=0.01$, $p=0.91$).

The response of native plant species appeared to be more highly influenced by estimated fire severity than habitat, with cover and species richness of native species as well as total vegetation cover generally inversely proportional to fire severity (**Figure 6**). For cover of native species, very lightly burned areas (BARC level 1) had higher percent native cover ($\bar{X}=127.6 \pm 11.3\%$) than highly burned areas (BARC level 4, $\bar{X}=78.9 \pm 13.0\%$; $t=-2.23$, $p=0.035$) and potentially high burn areas (BARC 2, $\bar{X}=87.8 \pm 9.9\%$; $t=-1.97$, $p=0.06$; GLM, $df=1$ ¹; Figure 6). While BARC ratings of 1 typically denote unburned areas, for the Woodward Fire areas, it included scorched and very lightly burned areas. A comparison of BARC and field derived adjusted FMH ratings are presented in Appendix C. It should be noted that lightly burned areas included areas where fire suppression activities such as back burning were performed to contain the fire. Native species richness also remained higher in lightly burned areas ($\bar{X}=23.0 \pm 2.9$ spp/plot) than in extreme ($\bar{X}=13.7 \pm 2.4$ /plot; $t=-2.75$, $p=0.01$) or moderate burn areas ($\bar{X}=15.9 \pm 1.5$ spp/plot; $t=-2.41$,

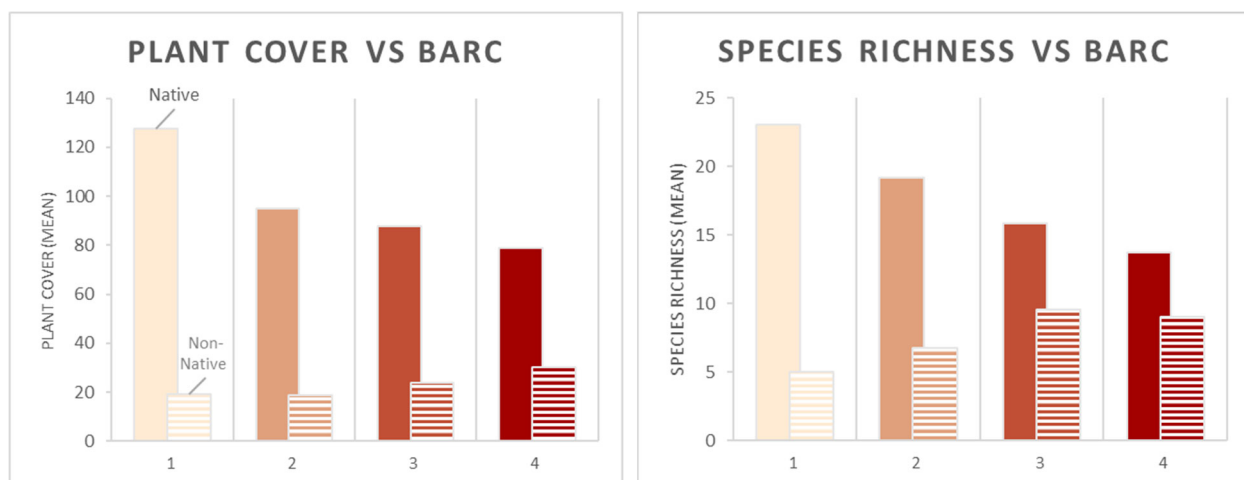
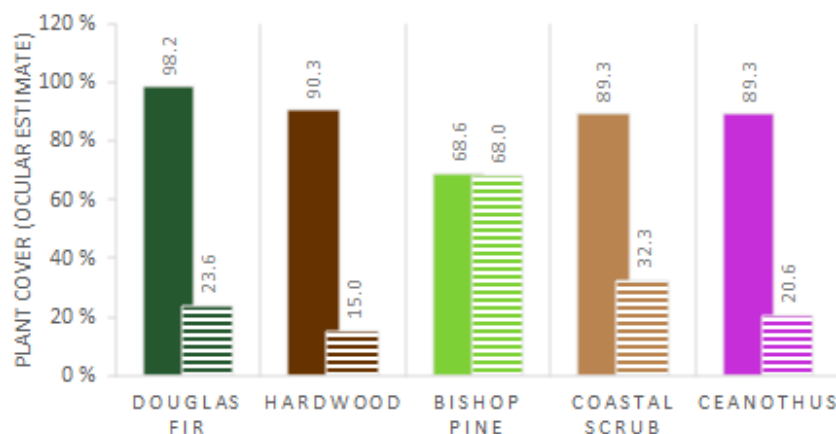


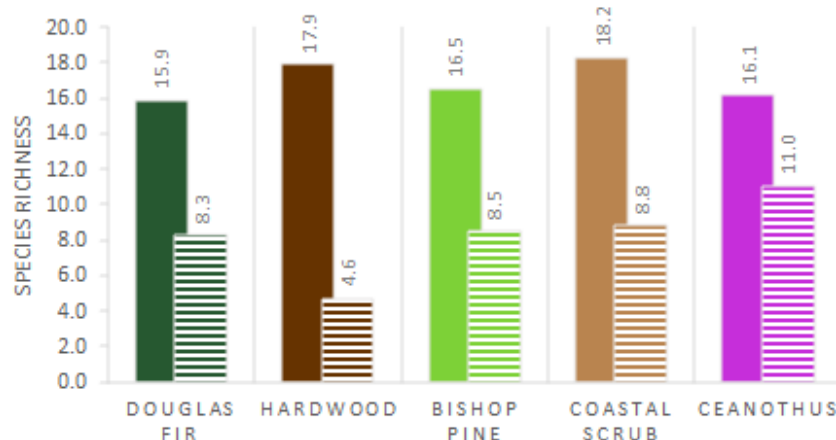
Figure 6: Mean plant cover (left) and species richness (right) of native (solid bars) and non-native (striped bars) cover at the four (4) BARC levels (an estimate of burn severity). Values are an average (mean) across all plots at that BARC level, regardless of habitat type.

¹ orderNorm-transformed

PLANT COVER VS PLANT COMMUNITY



SPECIES RICHNESS VS PLANT COMMUNITY



SPECIES DIVERSITY VS PLANT COMMUNITY

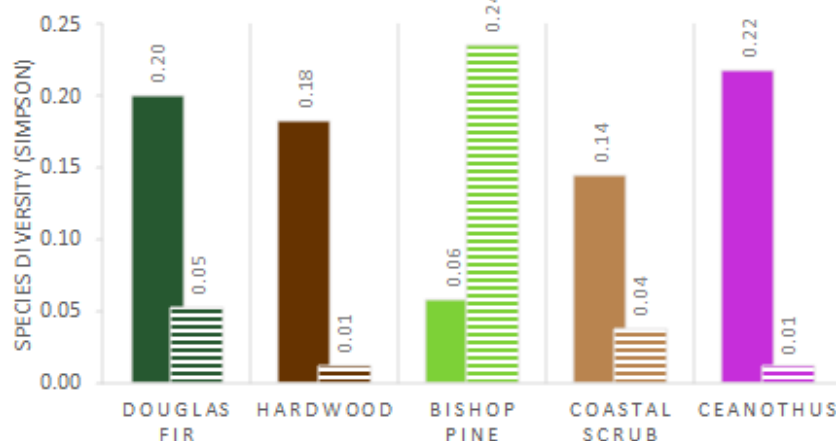


Figure 7a-c : Comparative measure of native (solid bars) and non-native (striped bars) species in the monitoring plots across all five vegetation communities. For full data and description of measurements, refer to Table 2.

$p=0.024$; GLM, $df=1^2$; Figure 6). Habitat appeared to have no influence on native plant species cover and richness (all $p>0.34$; Table 2; **Figures 7a-7b**). Neither fire severity nor habitat appeared to control species diversity, either native or non-native (all $p>0.20$; Table 2; **Figure 7c**).



Figure 8. Dense stand of bush lupine along Coast Trail in 2022

In general, total vegetation cover by year two following the fire seemed to be highest in scorched/very lightly burned areas relative to lightly burned ($t=1.9$, $p=0.07$); moderately burned ($t=1.93$, $p=0.065$), and highly burned ($t=1.96$, $p=0.06$) ones (GLM, $df=1^1$). This relationship is evident in some of the time-series photos taken between 2020/2021 and 2022 (Appendix B). Based on these photos, forested areas that were highly burned (e.g., Sky PP16 25°) appear to support less plant cover than those that were more lightly burned, even two years following the fire (e.g., HT PPT03 280° Appendix B). Conversely, the Coastal Scrub and Ceanothus plots appear to rebound more quickly, at least in terms of establishment by herbs (e.g., CT PP1 130°; Appendix B). Within some of the scrub

plots, rapid vegetation recovery appeared to be spearheaded by extensive establishment – or re-establishment -- of bush lupine (*Lupinus arboreus*), a species native to California, but believed to not be native to areas north of San Francisco (e.g., Sky PP22 355°; Appendix B). Dense stands of bush lupine colonized many of the Coastal Scrub areas, including along the Coast Trail (**Figure 8**). Bush lupine cover ranged from 14 (± 8.5)% in the few Bishop Pine stands to 15.2 (± 21.4)% in Coastal Scrub plots, with the remaining habitats having less than 5.5% cover.

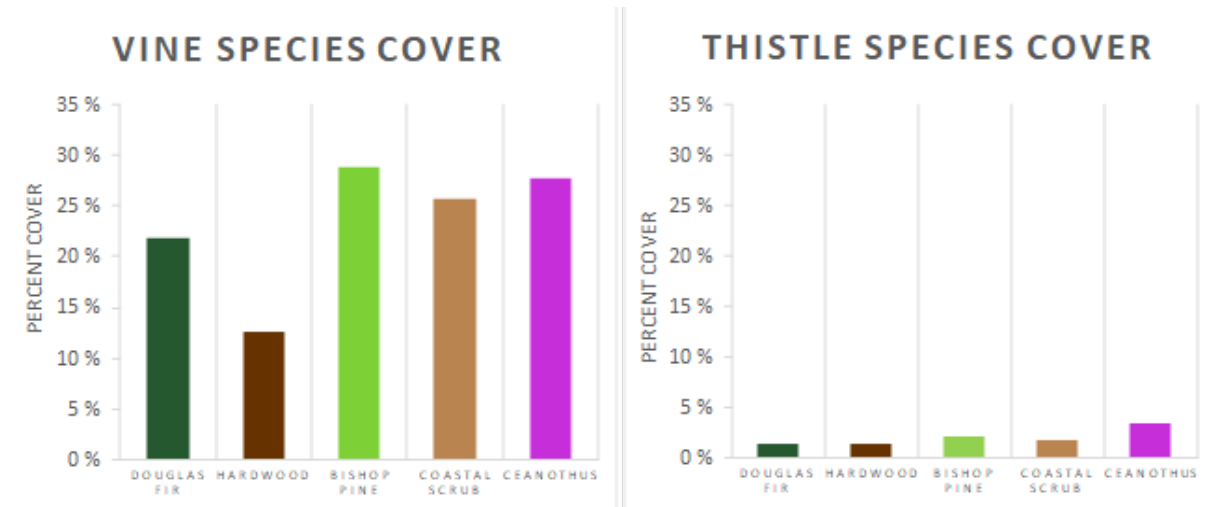


Figure 8. Burned Ceanothus stand colonized by morning glory (*Calystegia purpurata*)

Burned areas were also extensively colonized by vine lifeform species in 2022 (**Figure 9a**). While differences between fire severity and habitats were not statistically significant (all $p>0.22$), vine species cover ranged between 12.6 ($\pm X$)% in Hardwood Forest plots and 28.9 (± 2.9)% in Bishop pine plots, with other habitats averaging higher than 20% cover (Table 2; e.g., WVT PP3 180, Appendix B). Vine taxa included native species such as California blackberry (*Rubus ursinus*), wild cucumber (*Marah fabaceus*), morning-glory (*Calystegia purpurata*), and bedstraw (*Galium* spp.), as well as non-native ones such as bindweed (*Convolvulus arvensis*; **Figure 10**). Only five (5) to seven (7) of the 35 plots did not support California blackberry or wild cucumber, respectively. While the Seashore

did not monitor vegetation recovery as extensively in 2021, park staff removing weeds in the burned area attest to California blackberry being often among the first species to establish post-fire (M. Golding, NPS, *pers. comm.*) It is likely that these early seral stage species will dwindle over time as they are outcompeted.

² Box-Cox transformed



Figures 9a-b: Percent cover, as determined by ocular estimate, of vine (left) and thistle (right) species.

Non-native cover within plots ranged from a low of 15.0 (± 12.7)% in Hardwood Forest to a high of 68.0 (± 27.7)% in Bishop pine, with remaining habitats averaging around 20% (Table 2). Non-native plant species cover did not show a strong relationship with either estimated fire severity or habitats, perhaps due to high variability between plots. However, the percent cover of thistles -- which includes non-native thistle species such as milk thistle (*Silybum marianum*), Italian thistle (*Carduus pynoccephalus*), bull thistle (*Cirsium vulgare*), and tocolate (*Centaurea melitensis*) – differed significantly between burn levels (**Figure 11**). Thistle cover for moderately burned areas ($\bar{X}=2.7 \pm 0.3\%$; GLM, $df=1$, $t=2.85$, $p=0.01^1$; Fig. 9) averaged higher than very lightly burned areas ($\bar{X}=0.3 \pm 0.8\%$; Tukey HSD, adj. $p=0.01$).

Cover of thistles also varied by habitat, with cover in Ceanothus plots ($\bar{X}=3.4 \pm 0.8\%$) exceeding that of



Figure 11. Dense patch of milk thistle (*Silybum marianum*)

Douglas-fir plots ($\bar{X}=1.4 \pm 0.5\%$; GLM, $df=3$, $t=-2.41$, $p=0.02^1$; Table 2; **Figures 9b & 12**). Non-native species richness also differed between Ceanothus plots ($\bar{X}=11.0 \pm 1.3$ spp/plot) and Hardwood Forest plots ($\bar{X}=4.6 \pm 0.7$ spp/plot; $t=-4.50$, $p=0.0001$) and potentially Douglas-fir plots ($\bar{X}=8.3 \pm 1.3$ spp/plot; $t=-1.92$, $p=0.07$; GLM, $df=3^3$; Table 2; **Figure 7b**). Hardwood Forest supported fewer non-native species than either Douglas-fir (Tukey HSD, adj. $p=0.03$) or Coastal Scrub ($\bar{X}=8.8 \pm 1.0$ spp/plot; Tukey HSD, adj. $p=0.003$; Table 2; Figure 7b).

Ceanothus plots differed from Coastal Scrub counterparts in having a lower percent cover of damaged vegetation ($\bar{X}=16.1 \pm 8.2\%$ vs $\bar{X}=40.0 \pm 3.3\%$; GLM, $df=3$, $t=2.67$, $p=0.01^1$; Table 2). Cover of damaged plants in Hardwood Forest plots ($\bar{X}=29.6 \pm 6.1\%$) also differed significantly from Coastal Scrub (Tukey HSD, adj. $p=0.003$) and Douglas-fir ($\bar{X}=36.8 \pm 15.4\%$; Tukey HSD, adj. $p=0.03$; Table 2). Results from data collected within damage assessment subplots supported this finding, with density

³ Standardized inverse hyperbolic sine-transformed

of damaged vegetation significantly higher in Coastal Scrub ($\bar{X}=0.27 \pm 0.10$ indiv/m²) than in Ceanothus ($\bar{X}=0.01 \pm 0.01$ indiv/m²; Tukey HSD, adj. $p=0.01$), Douglas-fir ($\bar{X}=0.02 \pm 0.01$ indiv/m²; Tukey HSD, adj. $p=0.02$), and Hardwood Forest ($\bar{X}=0.05 \pm 0.02$ indiv/m²; Tukey HSD, adj. $p=0.04$; ImPerm, df=3, $p=0.001$; Table 3). While Table 3 would suggest that Coastal Scrub and Ceanothus might have had a higher density of dead individuals than either of the forest habitats, these differences were marginally significant (ImPerm, df=3, $p=0.06$; Tukey HSD, all adj. $p > 0.10$). There was also no relationship between density of damaged, dead, and resprouting individuals and fire severity (all $p > 0.74$).



Figure 12. Ceanothus plot colonized by abundant thistles.



Figure 13. Resprouting individuals in highly burned Coastal Scrub area.

Habitat type strongly influenced density of resprouting individuals (ImPerm, df=3, $p=0.001$), with density significantly elevated in Coastal Scrub ($\bar{X}=0.16 \pm 0.04$ indiv/m²) relative to Ceanothus ($\bar{X}=0.002 \pm 0.002$ indiv/m²; Tukey HSD, adj. $p=0.0009$), Douglas-fir ($\bar{X}=0.01 \pm 0.01$ indiv/m²; Tukey HSD, adj. $p=0.0009$), and Hardwood Forest ($\bar{X}=0.04 \pm 0.02$ indiv/m²; Tukey HSD, adj. $p=0.04$). The dominant species in most of the Coastal Scrub plots, coyotebrush, depended entirely on resprouting for recolonization (**Figure 13**). Hardwood tree species such as California bay (*Umbellularia californica*) not only resprouted, but produced seedlings (**Figures 14a & 14b**), while coniferous species such as Douglas-fir and Bishop pine relied strictly on seedling recruitment. Many of the understory and subcanopy species in forested environments – California huckleberry (*Vaccinium ovatum*), tanoak, coffeeberry (*Frangula californica*), and hazelnut (*Corylus cornuta*), also relied primarily on resprouting, although some also recruited from the seedbank (e.g., red elderberry).

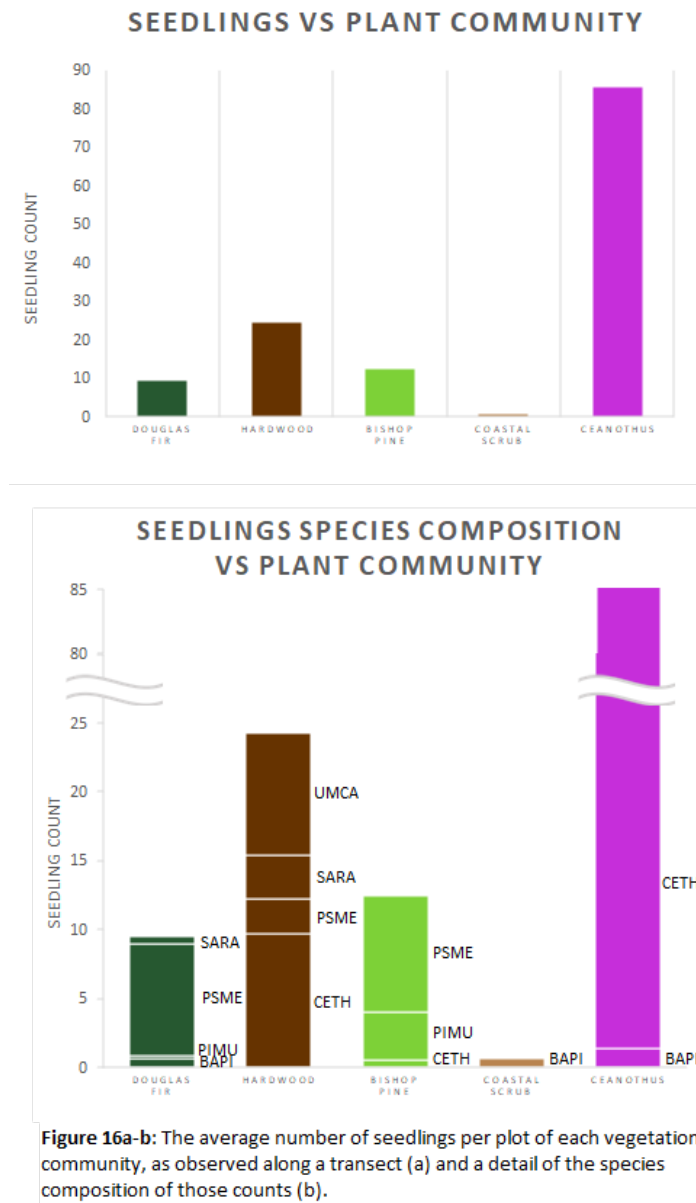
Seedling density peaked in Ceanothus plots ($\bar{X}=5.71 \pm 3.30$ /m²), which had much higher seedling recruitment than Coastal Scrub plots ($\bar{X}=0.05 \pm 0.02$ /m²; $t=-2.68$, $p=0.01$; GLM, df=3⁴; **Figure 15**). Hardwood Forests plots also supported higher density of seedlings ($\bar{X}=1.6 \pm 0.6$ /m²) than Coastal Scrub (Tukey HSD, adj. $p=0.04$). Total counts of seedlings in each habitat type are displayed in **Figure 16a**, while **Figure 16b** shows the species composition of seedlings in each habitat type. It should be noted that Ceanothus seedlings were very common in the Hardwood Forest plots (8), while Douglas-fir seedlings were very common in the few Bishop pine plots (2) monitored (Figure 16b). Cover of



Figure 15. Ceanothus stand with dense seedling recruitment

⁴ log-transformed data

Ceanothus seedlings was highest, not surprisingly, in Ceanothus habitats ($\bar{X}=20.6 \pm 30.7\%$), but Ceanothus seedling cover in Hardwood Forests and Bishop Pine plots averaged 7.4 ($\pm 19.3\%$) and 5.5 ($\pm 6.4\%$), respectively. The large standard error for these data suggests that Ceanothus establishment within both Ceanothus and non-Ceanothus plots is highly variable, probably due to factors not captured by this study.



DISCUSSION

Despite two years of drought following the Woodward Fire, most of the habitats impacted appear to be rebounding in terms of becoming revegetated, and a high percentage of the species and cover that reestablished are native. In fact, in most cases, cover of native plants surpassed that of non-native plants by a factor of three (3) or more. The only exception was for the few Bishop pine plots, where native and non-native were roughly equivalent (Table 2). The exceptional recovery of native plants in all habitats may ultimately be linked to several factors. First, the Woodward Fire unlike the Vision Fire was not characterized as an extreme or severe fire. Our results suggest that fire severity was inversely related to native plant cover and species richness, and thus the more moderated nature of this wildfire may have promoted widespread recovery of native plant species within all habitats. Other studies have found a strong positive relationship between fire severity and abundance of non-native plants or weeds (Crawford et al. 2001, Hunter et al. 2006, Alba et al. 2014). While high variability in non-native cover and species richness between our plots may have lessened our ability to discern the same linkage between fire severity and non-natives, the results point to higher native plant cover in areas that burned less severely. Many of the lightly burned areas comprised areas where fire crews used suppression tactics such as burn out operations to ignite surface fuels within the containment lines of a wildfire to consume fuel between the control lines and the wildfire's edge. This may have helped to promote recovery of native species.

A meta-analysis that used 268 studies from the U.S. and Australia that evaluated the effect of wildfire and prescribed fire on community-level responses of native and non-native species reached somewhat similar conclusions regarding the relationship between native vegetation recovery trajectories and fire severity (Alba et al. 2014). Over intermediate to long time scales, wildfires promoted non-natives both in terms of species composition and plant performance, while having no effect on native species composition and significantly hindering native plant performance (Alba et al. 2014). Because wildfires are typically larger, more variable, and potentially more intense than prescribed fires (Graham et al. 2004), establishment and/or spread of non-native invasives – particularly those who leverage disturbance to gain a foothold – may be favored over natives (Alba et al. 2014). Invasive species often possess traits that enable them to capitalize on large-scale disturbance events, including rapid growth and reproduction, and a greater ability to respond favorably to environmental fluctuations (Alba et al. 2014). Furthermore, while some invasive species do seem to act as pioneers or primary successional species that eventually give way to others through vegetation succession, studies included in the meta-analysis pointed to non-native invasives often persisting in the landscape even more than 10 years after wildfires (Alba et al. 2014).

The relationship between wildfire and invasives did show variation within habitat types, but temperate forests, including semi-arid coniferous forests in the western U.S., were among those most impacted by spread of invasives after wildfires (Alba et al. 2014). Fire suppression, past logging, drought, and increasing temperatures in this region (Westerling et al 2006) have led to replacement of relatively frequent, low-severity surface fires by high-severity, stand-replacing crown fires (Savage and Mast 2005). As noted in the Introduction, within western portions of Marin County, MFI climbed from a range of 7.7-8.5 years in the 1820's through the mid-1900's to 37.6 years in the last century (Brown et al. 1999, Dawson 2021).

Fire suppression and prolonged intervals between fires create forest-health issues such as higher fuel loads and loss of fire-dependent habitats like Bishop pine to senescence of older stands, but an increase

in fire frequency with much shorter MFI's can also generate negative impacts on vegetation communities. Within California chaparral shrublands, sites that burned again 3 years following the last wildfire had much higher non-native plant cover and lower plant diversity than those that had burned 24 years previously (Keeley and Brennan 2012). At sites where fires occurred four years apart, annual species represented the dominant vegetation after the second fire, with non-natives annuals being the most common (Keeley and Brennan 2012). Native woody species regenerated well after the first fire, but declined after the second, with one obligate seeding shrub extirpated from two sites after repeat fires (Keeley and Brennan 2012). Similar results were reported by researchers evaluating the effect of fire frequency on Mediterranean forests and shrublands in Chile: mean native species richness – if not cover -- declined with increasing fire frequency (Smith-Ramirez et al. 2021). The authors concluded that native vegetation in Chile could rebound to pre-fire conditions after one or two fires, but three consecutive fires reduced cover, richness, and abundance even 20 years after the fire (Smith and Ramirez 2021).

As discussed earlier, roughly half of the Woodward Fire area also burned during the Vision Fire, but we were largely unable to analyze fire frequency due to the fact that some habitats were not equally distributed between once- and twice-burned areas, with forested habitats mostly burned once and Ceanothus habitats always burned twice. Coastal Scrub plots were roughly divided between once and twice burned: they showed no differences in vegetation cover, richness, and diversity variables based on fire frequency. Sample size, however, was admittedly small, so it's difficult to draw any definitive conclusions. In addition, the effects of repeat burning might have been dampened by the fact that the Vision Fire occurred 25 years prior to the Woodward Fire, a much-longer return interval than any of the studies cited above.

Despite positive signs of rapid revegetation and recovery by native species, there is still room for concern. Results of the meta-analysis study are a sobering reminder that non-native invasive plant species are poised to take advantage of any and all disturbance events and, once established, may be difficult to oust. The second-tier fire effects will continue to reveal themselves over the long term. The Seashore started actively managing jubata (*Cortaderia jubata*) in 1988, and by 1995, all known jubata grass occurrences had been removed from the Seashore (Hamingson 2020). However, after the Vision Fire in 1995, a large, previously undetected jubata grass population was found in the wilderness, upwind of the exposed burn area (Hamingson 2020). Aided by wind dispersal, jubata soon expanded within the Vision Fire footprint and, by 2016, occurred across 3,025 acres of the Seashore (Hamingson 2020). By 2020, the NPS had invested more than \$200,000 to treat 278 of these patches, but difficulty in accessing many of the patches prevented full treatment (Hamingson 2020).

Not surprisingly, jubata has again proved to be one of the biggest management challenges since the Woodward Fire. Seventy-two historic jubata points occurred in the Woodward Fire footprint (Hendrickson 2022). In 2021, park staff documented 1,543 mature plants, some of which may have been present previously, but had not been recorded (Hendrickson 2022). The park treated slightly more than half of the resprouts before they set seed in 2021, but in 2022, approximately 400 seedlings were found in areas downwind of plants that hadn't been treated the previous year (Hendrickson 2022). With each passing year, jubata treatment becomes more difficult due to issues with accessing off-trail areas supporting dense vegetation (R. Hendrickson, NPS, *pers. comm.*). Ironically, one positive sign of vegetation recovery – high total plant cover – can be viewed as an obstacle in removal of jubata and other weeds.



Figure 17. Ox-eye daisy along the Coast Trail in burned Coastal Scrub

Other non-native invasive species that existed within the burned area previously include veldt grass (*Ehrharta erecta*), tocalote, ox-eye daisy (*Leucanthemum vulgare*), infertile capeweed (*Arctotheca prostrata*), onion grass (*Romulea rosea*), Scotch broom (*Cystisus scoparius*), and Eucalyptus (*Eucalyptus globulus*; **Figure 17**). Ruderal species such as thistles are often quite prevalent after fires (Rew and Johnson 2010), as is cutleaf burnweed (*Senecio glomeratus*). While somewhat patchy in spatial extent, dense thistle stands have encroached upon quite a few habitats in the Woodward Fire area, with establishment more prevalent in highly burned areas, along the dozer lines, and in certain types of habitats such as Ceanothus.

The dilemma that land managers are faced with is whether these species pose a high enough risk to invest considerable resources in eradication. Many non-native plant species decline naturally during later successional stages (Rew and Johnson 2010). During the Vision Fire, park staff and volunteers devoted a significant amount of time in manually removing cutleaf burnweed, while staff whittled away at the thistle population: later, park staff lamented that this effort could have been better spent elsewhere, as these species would probably have naturally dwindled in abundance as native vegetation communities started to recover (NPS 1997, L. Allen, former NPS, *pers. comm.*). Ironically, according to one researcher who conducted a post-fire vegetation study on the Inverness Ridge, efforts to remove cutleaf burnweed along trails “seemed only to increase its presence” (Holzman 1995). However, by year 5, cutleaf burnweed had largely disappeared from the study area (Holzman 1995). While these observations are promising, given the results of the Alba et al. (2014) meta-analysis study, caution must be advised in dismissing spread of any non-native species as inconsequential or naturally reversible.

A case in point is bush lupine. Abundance of this species -- which, again, is native to California, but supposedly not to Marin County -- exploded after the fire, with vast monocultures forming in many shrubland or open forest areas, particularly along the Coast Trail. While post-wildfire issues with bush lupine are not as well documented, at least one study claimed that lupine only germinated after intense fires (Whelan 1995 in Holzman 1995). Bush lupine has proven to be a tenacious issue within New Zealand dunes and non-native portions of its range in California coastal prairie and dunes (Pickart et al. 1998, Maron and Jeffries 1999, Hetherington and Wilson 2019). As a nitrogen (N)-fixer, bush lupine derives 60% to 70% of its total N from atmospheric sources (Bentley and Johnson 1991) and builds N pools in normally N-limited systems to twice that of lupine-free areas (Maron and Jeffries 1999). With leaf drop or lupine death, high levels of N become available to plant colonists, particularly weedy annuals such as ripgut grass (*Bromus diandrus*), that return nutrients to the soil upon death, creating a positive feedback loop for nitrophilous weeds only disrupted by re-establishment of bush lupine from seed (Maron and Connors 1996, Maron and Jeffries 1999).

Information on this species’ spread following the Vision Fire is scant. The transect-based vegetation study conducted on the Inverness Ridge did report extensive colonization by bush lupine, noting that percent cover jumped by more than 200 percent between years 1 and 2 post-fire (Holzman 1995). Excluding the Bayview transects, percent cover of lupine in all Inverness Ridge sites averaged 20 percent, with cover lowest in less intensely burned areas (Holzman 1995). Exposure did not appear to

influence bush lupine abundance, as it was fairly evenly distributed between protected and exposed sites at the end of the second season (19.3% and 22.3% cover, respectively; Holzman 1995). A second study in coastal scrub areas burned during the Vision Fire also detected bush lupine and correlated its abundance with burn intensity (A. David, SFSU, unpub. data). While formal monitoring on the Inverness Ridge was discontinued after year 2, informal surveys during year 5 found that lupine had virtually disappeared from the site (Holzman 1995). While formal vegetation monitoring for the Woodward Fire was not conducted in 2023, evidence of bush lupine die-back was already evident three years after the Woodward Fire (R. Hendrickson, NPS, *pers. comm.*; **Figure 18**).



Figure 18. Dead bush lupine shrubs in 2023

Following the Vision Fire, a large portion of the vegetative cover on the Inverness Ridge after the second year (roughly 85 percent) was composed of three nitrogen-fixing plants, specifically *Ceanothus*, bush lupine, and lotus (*Lotus* spp.; Holzman 1995). While both lupine and lotus are legumes, *Ceanothus* is a member of the Ericaceae family. Legumes rely on soil bacteria called *Rhizobia* to fix nitrogen, while non-legumes such as *Ceanothus* associated with a different soil bacteria, *Frankia*, which are often found in nodules along the plant roots (Delwiche et al. 1965, Story 1974, Kummerow et al. 1978). Other non-legume genera are also able to fix nitrogen, including

manzanita (*Arctostaphylos*), mountain mahogany (*Cercocarpus*), and even alder (*Alnus*; Story 1974). *Ceanothus* seeds are stimulated to germinate by the heat of fire and provide nitrogen-fixing qualities to the soil (Daubenmire 1968 in Holzman 1995, Quick and Quick 1961, Biswell 1989, Schoenherr 1992 in Holzman 1995). As Holzman (1995) noted, this species establishment pattern “is typical of early successional stages and indicates a healthy process.” These species traits may also explain why *Ceanothus* was able to gain a foothold in both Hardwood Forests, to a much lesser extent, Bishop pine stands through seedling establishment. Whether this will eventually result in a type conversion remains to be seen.

One other interesting finding from our study was the prevalence of vine species among all habitats, with California blackberry, wild cucumber, morning glory, and field bindweed being the most common. As discussed earlier, with the exception of Hardwood Forest (12.6%), cover of vine species ranged between 21.9% (Douglas-fir) to 28.9% (Bishop pine). Holzman (1995) referenced California blackberry as a species of significant cover two years after the fire, occurring in 12% of the monitored areas, with percent cover increasing 22% to 33% between year 1 and 2 in the more successfully colonized sites. Wild cucumber apparently boomed immediately after the fire, but had largely disappeared by year 2 (Holzman 1995). Few studies have specifically addressed response of vine species to wildfires. In the tropical forests of the Amazon, large-stem-diameter vine species dwindled dramatically even in lightly burned forests (76%), but in more intensely burned areas, vines with a smaller stem diameter (<2.5 cm) formed a dense mat on remaining trees, with species often characterized by so-called aggressive, pioneering species that included Curcubitaceae, of which California’s wild cucumber is a member (Cochrane and Schulze 1999). The climbing habit of vines enables these species to compete with non-climbing vegetation for resources such as light, nutrients, and water (Paul and Yavitt 2011). Damaged and dead trees and shrubs provide

ample substrate for these “scramblers,” as they have been called, to flourish after wildfires. It remains to be seen whether these species will persist once trees and shrubs have started to recover more.

With only two years having elapsed since the Woodward Fire, it is too soon to predict whether this fire will result in type or lifeform conversions. However, the seedling data does provide some insight into possible future conversion trajectories. Ceanothus and, to a much lesser extent, Douglas-fir were actively recruiting into Hardwood Forest. Some of these potential changes may not be realized over the long-term (i.e., Hardwood Forest conversion to Ceanothus), just as they weren’t following the Vision Fire (Forrestel et al. 2011). However, the incursion of Douglas-fir into Hardwood Forest is interesting in the context of strong concern being expressed in recent decades about continued Douglas-fir expansion and encroachment into grasslands, shrublands, chaparral, and oak woodlands (Biswell 1954, McBride and Heady 1968, McBride 1974, Williams et al. 1987, Dunne and Parker 1999, Russell and McBride 2003, Keeley 2005, Kennedy and Sousa 2006, Zavaleta and Kettley 2006, Hsu et al. 2011, Startin 2022). Locally, many have expressed unease regarding the seemingly aggressive expansion by Douglas-fir on the Inverness Ridge and adjacent Bolinas Ridge over the past century (Brown et al. 1999, Anderson et al. 2013, Startin 2022).

In addition to cessation of logging and agriculture-related clearing, researchers also point to suppression of wildfires and the prescribed burning practices of the Coast Miwok as reasons for Douglas-fir expansion, with the assumption that Douglas-fir is averse to burning (Dunne and Parker 1999, Brown et al. 1999, Russell and McBride 2003). As noted earlier, results of paleoecological and charcoal studies within Point Reyes suggest that fire and regeneration of Douglas-fir might have been tightly coupled prior to human settlement (Anderson 2005, Anderson et al. 2013). While some older studies claimed that Douglas fir only prospers under a “no fire” scenario (Agee and Huff 1980 *in* Gallagher et al. 2021, Ryan et al. 1988, Keane et al. 1990), recent studies point to Douglas-fir being fire-tolerant (Mooney and Dawson 2016, UCANR 2022) and showing an intermediate level of tolerance for wildfires, with response seemingly influenced by fire severity (Welch et al. 2016). Several studies released in the last 10 years that evaluated species regeneration post-wildfire over broad geographic scales (e.g., North American Mediterranean climate zone) have found Douglas-fir to be among the most abundant regenerating species, with regeneration particularly favored under moderate-intensity fires (Welch et al. 2016, Povak et al. 2020). In the coastal ranges of California, Douglas-fir tends to be less of an early seral species than in other areas such that other species tend to colonize first (A. Forrestel, NPS, *pers. comm.*). Ultimately, Douglas-fir may possess a competitive advantage by exhibiting an extremely adaptable or plastic response to fire – regenerating following both less severe wildfires and also during periods of no fire by infilling underneath the canopy of other tree species (Welch et al. 2016).

Results of a recent historical vegetation mapping project that evaluated vegetation community changes between the 1940’s and 2018 found that the spatial extent of Douglas-fir as well as Hardwood Forests was relatively stable between the early 1940’s and early 1960’s, but jumped dramatically between the 1960’s and late 1990’s (~1,086 and 996 acres, respectively; Parsons et al. 2023). Bishop pine forest cover jumped 59% due largely to extensive establishment following the Vision Fire (Parsons et al. 2023). Between the 1990’s and 2018, all three forest types continued to expand, albeit at a much slower rate of growth (Parsons et al. 2023). However, at least over the past 20 years, gains by Hardwood Forest have outpaced those of Douglas-fir by four-fold (Parsons et al. 2023). Support for these findings comes from a separate analysis conducted by GGNPC on changes in Lifeform between the early 1990’s (which does not include changes resulting from the Vision Fire) and 2018, which showed that acreage of

coniferous forest (Douglas-fir and Bishop pine) climbing slightly (2%) during this period, while Hardwood Forest grew by 64% (Tukman Geospatial 2023). Viewed from a broader lifeform perspective, between the 40's and 2018, mixed herbaceous/shrublands and native forests grew by 40% and 37%, respectively, while herbaceous communities shrunk by 40% (Parsons et al. 2023). Shrub-dominated habitats contracted by 8% (Parsons et al. 2023). In contrast, a recent historical aerial imagery study conducted for the adjacent Bolinas Ridge found that between 1985 and 2010 herbaceous plant communities and shrubland shrunk by 62% and 51%, respectively, while forests increased by 307% (Startin 2022).

As Holzman (1995) concluded following the Vision Fire, vegetation in the Seashore has demonstrated its evolutionary resilience to fire. The fast vegetative regrowth after the 1995 Mt. Vision and the 2020 Woodward Fires characterized by a strong native component attests to this resiliency. However, it is still too early to determine what role non-native invasives might play in influencing this recovery and whether this fire will result in habitat conversion as did the Vision Fire. Vigilant re-treatment of invasive non-natives will be key to ensuring return of a healthy ecosystem. Managers also need to better understand exactly which factors drive invasibility of habitats following fire -- why do some stands/habitats have more thistles and other weeds than others? -- and the long-term threats posed by some of the pioneering or early successional species such as bush lupine. Future monitoring, including aerial imagery analysis of change at the community level, will help us to better understand the role of wildfire at Point Reyes in controlling vegetation landscape dynamics.

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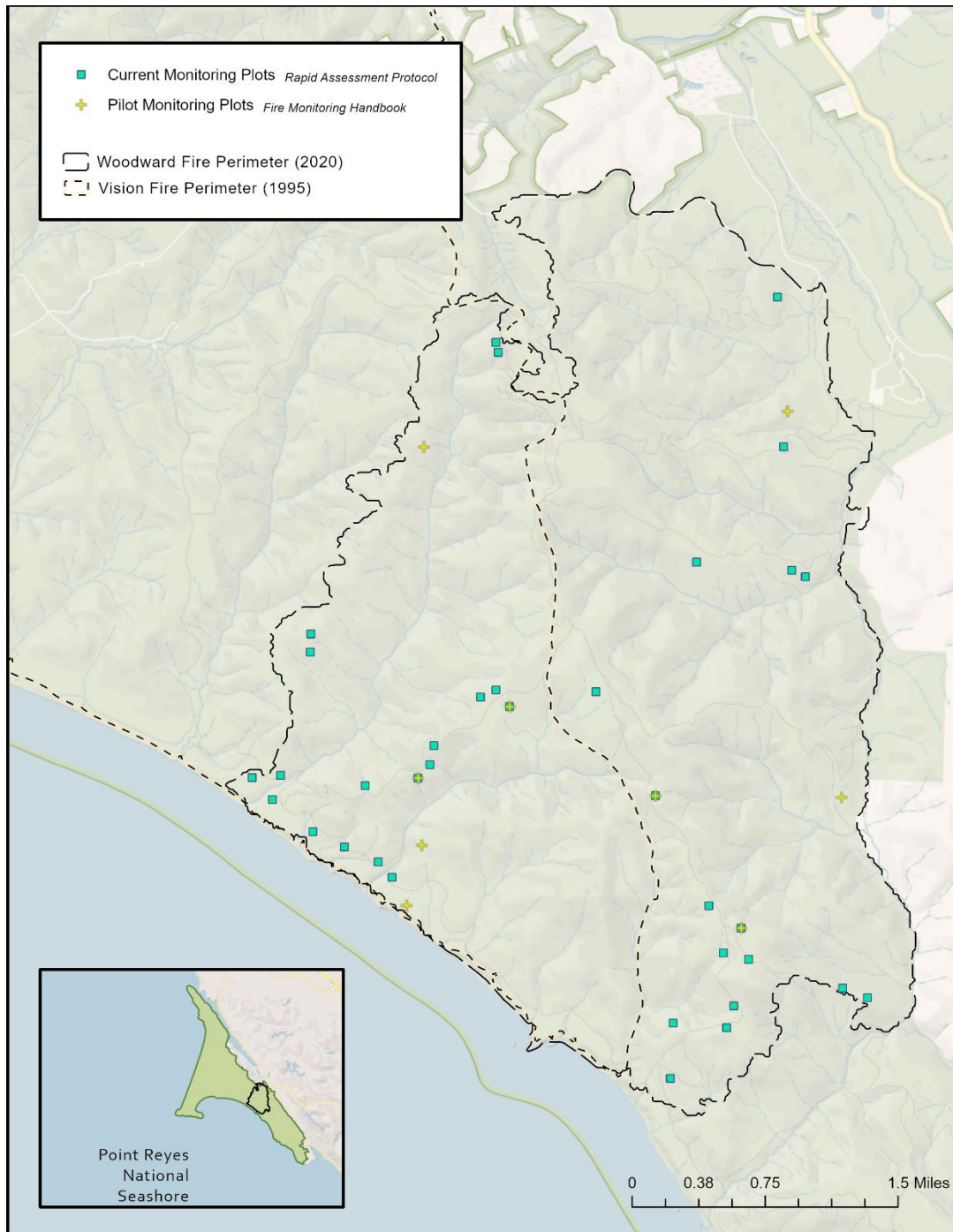
Table 2. Summary table of cover, richness, diversity, and regeneration data for plots in different habitats in the Woodward Fire burn area-2022. Seedling regeneration assessed within 15m² belt transect.

	Douglas Fir	Hardwood	Bishop Pine	Coastal Scrub	Ceanothus
	(mean)	(mean)	(mean)	(mean)	(mean)
BARC Rating	2.8	2.3	3.5	3.0	2.9
Reversed FMH Rating	3.4	2.6	4.5	3.6	4.0
2010-2019 Density Change (%)	4.8	-0.8	8.9		
2010-2019 Forest Gap (%)	1.7	4.0	2.9		
2019 Standing Dead (%)	<0.5	<0.5	<0.5	<0.5	<0.5
<i>Current Vegetation Species Composition and Structure</i>					
Vascular Percent Cover	121.8	105.3	136.6	126.3	113.0
Bareground Cover	8.6	12.5	8.5	4.1	6.5
LWD Cover	1.7	3.0	17.5	0.5	2.9
Native Plant Cover	98.2	90.3	68.6	89.3	89.3
Non-Native Plant Cover	23.6	15.0	68.0	32.3	20.6
Tree Cover	20.7	35.4	0.0	0.8	0.1
Shrub Cover	30.4	7.5	21.9	58.1	30.0
Herb Cover	71.2	63.1	117.1	68.6	85.6
Vine Species Cover (%)	21.9	12.6	28.9	25.7	27.7
Thistles (%)	1.4	1.4	2.3	1.8	3.4
Dead Plant Cover	16.3	2.6	38.0	12.8	36.7
Damaged Plant Cover	36.8	29.6	1.0	40.0	16.1
Native Species Richness	15.9	17.9	16.5	18.2	16.1
Non-Native Species Richness	8.3	4.6	8.5	8.8	11.0
Native Species Diversity (Simpsons-D)	0.2	0.2	0.1	0.1	0.2
Non-Native Species Diversity (Simpsons-D)	0.1	0.0	0.2	0.0	0.0
<i>Post-Fire Community Regeneration</i>					
# of Spp with Regenerating Individuals	1.8	2.1	0.0	1.7	1.3
# of Spp with Seedlings	2.1	2.9	4.5	1.8	1.8
# of Seedlings	9.5	24.3	12.5	0.8	85.6
# of BAPI Seedlings	0.6	0.0	0.0	0.7	1.4
# of CETH Seedlings	0.0	9.8	0.5	0.0	84.3
# of PIMU Seedlings	0.3	0.0	3.5	0.0	0.0
# of PSME Seedlings	8.1	2.5	8.5	0.0	0.0
# of SARA Seedlings	0.5	3.1	0.0	0.0	0.0
# of UMCA Seedlings	0.0	8.9	0.0	0.0	0.0
Seedling Density (seedlings/m2)	0.6	1.6	0.8	0.1	5.7
Average Height of Seedlings (cm)	44.5	26.1	14.4	40.7	44.4

Table 3. Summary table of fire impact-related variables such as plant death, damage, and regeneration data for different habitats in the Woodward Fire burn area within the damage assessment subplot -2022.

	Douglas Fir	Hardwood	Bishop Pine	Coastal Scrub	Ceanothus
	(mean)	(mean)	(mean)	(mean)	(mean)
BARC Rating	2.8	2.3	3.5	3.0	2.9
Reversed FMH Rating	3.4	2.6	4.5	3.6	4.0
2010-2019 Density Change (%)	4.8	-0.8	8.9		
2010-2019 Forest Gap (%)	1.7	4.0	2.9		
2019 Standing Dead (%)	<0.5	<0.5	<0.5	<0.5	<0.5
<i>Impact of Fire on Existing Vegetation (78.5 m²)</i>					
# of DAMAGED Plants	1.6	3.8	1.0	21.4	0.9
# of DEAD Plants	4.3	0.4	3.5	10.4	9.1
Density of DAMAGED Plants (/m2)	0.02	0.05	0.01	0.27	0.01
Density of DEAD Plants (/m2)	0.05	0.01	0.04	0.13	0.12
Average DBH of Trees (cm)	94.3	106.0	51.2		58.0
Average DBH of Subcanopy Trees (cm)	9.8				
Average Dripline of Shrubs (cm)		217.9	220.6	190.6	530.8
Average Max Scorch Height (m)	14.5	1.6	5.7	1.5	5.0
Average Max Char Height (m)	14.4	1.0	4.2	1.1	4.8
Average % of Canopy Scorched	77.7	53.6		91.7	99.6
# of Individuals Resprouting	0.5	4.8	0.0	12.3	0.1
Density of Resprouting Plants (/m2)	0.01	0.06	0.00	0.16	0.00
# of DAMAGED TREES	1.1	0.8	0.0	0.0	0.0
# of DEAD TREES	4.1	0.1	3.5	0.0	0.1
# of DAMAGED SHRUBS	0.0	4.4	1.0	21.4	0.9
# of DEAD SHRUBS	0.0	0.9	0.0	10.4	9.0
# of DAMAGED PSME	1.1	0.1	0.0	0.0	0.0
# of DEAD PSME	3.5	0.0	1.5	0.0	0.0
# of DAMAGED PIMU	0.0	0.0	0.0	0.0	0.0
# of DEAD PIMU	0.6	0.0	2.0	0.0	0.1
# of DAMAGED UMCA	0.0	0.5	0.0	0.1	0.0
# of DEAD UMCA	0.0	0.1	0.0	0.0	0.0
# of DAMAGED BAPI	0.0	2.1	0.0	19.4	0.0
# of DEAD BAPI	0.0	0.1	0.0	0.0	0.0
# of DAMAGED CETH	0.0	0.0	0.0	0.0	0.8
# of DEAD CETH	0.0	0.1	0.0	0.1	8.8

APPENDIX A: VEGETATION MONITORING PLOTS AT POINT REYES NATIONAL SEASHORE



APPENDIX B: VEGETATION PHOTOPOINT LOCATIONS AND COMPARATIVE PHOTOGRAPHS

HT PPT03 280° - Horse Trail (Hardwood Forest) BARC 2 -

01-17-21



07-10-2021



05-11-2022



OPT PP03 290° - Old Pine Trail (Douglas Fir) BARC 1/2 -

01-15-21



06-01-2021



07-22-2022



Sky PP14 310° - Sky Trail (Douglas Fir) BARC 4 -

03-30-21



07-23-2021



07-22-2022



Sky PP16 25° - Sky Trail (Douglas Fir) BARC 3/4 -

03-17-21



07-23-2021



07-22-2022



Sky PP19 235° - Sky Trail (Coastal Scrub) BARC 3 -

10-14-20



07-14-2021



05-24-2022



CT PP1 130° - Coast Trail (Coastal Scrub) BARC 3 -

12-07-20



06-29-2021



05-02-2022



CT PP4 15° - Coast Trail (Riparian Forest/Herbaceous Wetland) BARC 3 -

12-09-20



03-17-2021



06-22-2022



WVT PP7 70° - Woodward Valley Trail (Coastal Scrub) BARC 2 -

12-09-20



06-28-2021



05-02-2022



WVT PP3 180° - Woodward Valley Trail (Ceanothus Scrub) BARC 4 -

12-09-20



03-24-2021



05-02-2022



BVT PP4 255° - Bear Valley Trail (Hardwood Forest) BARC 1 -

11-20-20



05-19-2021



03-09-2022



Sky PP22 355° - Sky Trail (Coastal Scrub) BARC 4 -

12-09-20



07-23-2021



05-24-2022



CT PP2 280° - Coast Trail (Ceanothus Scrub) BARC 3 -

12-09-20



03-17-2021



06-22-2022



APPENDIX C: COMPARISON BETWEEN REMOTE IMAGERY-DERIVED BARC AND FIELD-DERIVED ADJUSTED FMH FIRE SEVERITY RATINGS WITHIN MONITORED PLOTS

Appendix Table A-1: Comparison between remote imagery derived BARC and Field-Derived Adjusted FMH fire severity ratings within Woodward Fire vegetation monitoring plots. Adj. FMH ratings involved visual observation of fire severity within four quadrants of the plots, which were then averaged for the overall plot. Numbers shown for Adj. FMH represent means with minimum and maximum values.				
BARC Rating ¹	1	2	3	4
Avg. Adj. FMH Ratings ²	2.3 (1.8-3.0)	3.2 (2.5-4.0)	3.6 (2.0-4.0)	4.1 (3.5-5.0)

¹ BARC Ratings: 1=Unburned; 2=Low; 3=Moderate; 4=High

² Adj. FMH Ratings: 1=Unburned; 2=Scorched; 3=Lightly Burned; 4=Moderately Burned; 5=Heavily Burned