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# Coast Redwood Community Structure and Composition following Selective-Harvest: A Case Study of Byrne-Milliron Forest

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COAST REDWOOD COMMUNITY STRUCTURE AND COMPOSITION  
FOLLOWING SELECTIVE-HARVEST: A CASE STUDY OF  
BYRNE-MILLIRON FOREST

A Thesis

Presented to

The Faculty of the Department of Environmental Studies

San José State University

In Partial Fulfillment

of the Requirements for the Degree

Master of Science

By

Amy K. Petersen

May 2014

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The Designated Thesis Committee Approves the Proposal Titled

COAST REDWOOD COMMUNITY STRUCTURE AND COMPOSITION  
FOLLOWING SELECTIVE-HARVEST: A CASE STUDY OF  
BYRNE-MILLIRON FOREST

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## ABSTRACT

### COAST REDWOOD COMMUNITY STRUCTURE AND COMPOSITION FOLLOWING SELECTIVE-HARVEST: A CASE STUDY OF BYRNE-MILLIRON FOREST

By Amy K. Petersen

The effects of selective-harvest on forest composition and structure in the southern range of the coast redwood (*Sequoia sempervirens*) forest have not been well documented. This case study was focused on the Byrne-Milliron Forest located in Santa Cruz County, where selective-harvest is currently the primary method of timber extraction. The purpose of this research was to determine how forest structure and composition changed in relation to time since the last harvest re-entry, the percentage cut per hectare and number of harvest re-entries. The study design consisted of 100 plots located within the Byrne-Milliron Forest, spanning over five harvest sites. Data were analyzed using one-way analysis of variance (ANOVA) and correlations were determined with Pearson product-moment correlation coefficient. Results indicated that the percentage cut per hectare was the strongest independent variable, with significant differences found among sites in relation to canopy cover, stand density, large woody debris (LWD) density, coast redwood sprouts, coast redwood associated herbaceous species and exotic species richness. These findings suggest that a lower percentage cut is more effective in maintaining healthy structure and composition in actively managed stands.

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## INTRODUCTION

Forests are valuable ecosystems, providing habitat to over half of the world's terrestrial species and supporting a vast number of ecological processes including nutrient cycling, hydrologic processes and carbon sequestration. However, logging over the last few centuries has decimated forests worldwide, resulting in loss of habitat, land fragmentation, regional climate change and erosion.

Over 700,000 hectares of forest, predominately coast redwood (*Sequoia sempervirens*), have been logged in the State of California since the start of Anglo-American settlement (Yaryan 2002). Historically, old-growth coast redwoods extended up the Pacific Coast from Monterey to the southern portion of Oregon. Due to a century and a half of destructive logging practices, old-growth redwood forests have been reduced to less than 5% of their original distribution (Russell et al. 2014). The remaining stands have been converted to second-growth forest and other land uses.

There has been substantial controversy regarding the management and restoration of the second-growth forests. Although some research has argued in favor of active management (Chittick and Keyes 2007; O'Hara et al. 2010), coast redwood forests exhibit extraordinary regenerative traits, including vegetative sprouting and clonal spread, that enable them to successfully recover and achieve old-growth characteristics without the aid of intensive management (Lutz and Halpern 2006; Russell and Michels 2011; Russell et al. 2014).

Continuous management of coast redwood forests has the potential to cause long-term structural and vegetative changes within the community (Russell and Michels 2010).

Research has shown that forests bordering harvest regions experience changes in microclimate and are more susceptible to crown dieback and windthrow (Matlack 1993; Russell et al. 2000). Previous studies have also highlighted the significant physiognomic and ecological impacts of logging, including loss of sensitive herbaceous species and vital structural components for habitat and nesting (Duffy and Meier 1992; Kahmen and Jules 2005; Baker et al. 2006).

Methods of timber extraction have been varied, ranging from removal of an entire stand (clear-cutting) to only removing selective trees within the harvest unit (selective-harvest). Although the negative impacts of clear-cutting are generally considered to exceed the impacts of other logging methods, practices such as selective-harvest may still pose issues for forest recovery. The goal of this research was to determine how structure and composition of coast redwood forests are recovering amidst active management implemented with selective-harvest.

## **LITERATURE REVIEW**

### **Coast Redwood Ecology**

The distribution of the *Sequoia* genus is limited to a few specific geographic locations. Species include the coast redwood, the giant sequoia (*Sequoiadendron giganteum*) and the dawn redwood (*Metasequoia glyptostroboides*) of China. The coast redwood is endemic to a narrow 724 km strip along the coast the California-Oregon border (U.S. Fish and Wildlife Service 1997). Its cousin, the giant sequoia, is located further inland along the Sierra Nevada Mountain range (Figure 1). Distribution has become limited due to climatic changes occurring since the Jurassic Period. Coast

redwoods in particular are situated only along the Pacific coast due to their need for summer fog during the dryer months of the season. The California current produces an advection, caused when a cold upwelling from the ocean floor creates an inversion layer. This results in the formation of fog, which redwoods rely on for foliar uptake (Burgess and Dawson 2004).

Coast redwood forests are found in a number of environments within their current range but most commonly thrive on alluvial terraces (Veirs 1982). They have extremely high biomass and are considered to be the most productive of all tree species. A mature forest contains an average of 1,400 metric tons of biomass per acre (Snyder 1992). Able to grow in excess of 100 m in height, the coast redwood is the tallest tree in the world. The tallest redwood currently on record resides in Redwood National Park and stands 116 m tall (Hodgson 2006).

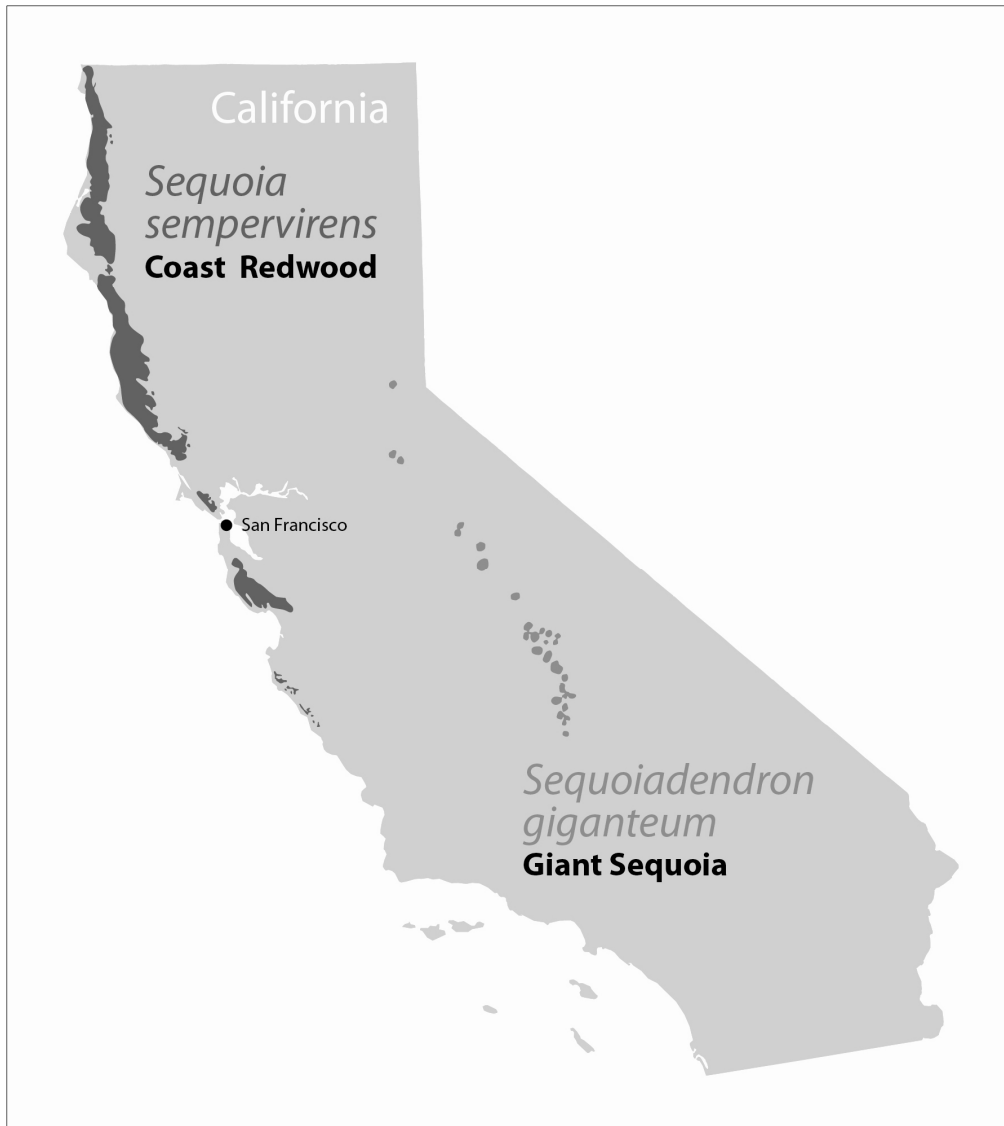


Figure 1. *Sequoia* genus distribution in California. Image by Derrick Wynes.

There are a number of associated species that reside in coast redwood communities. Common hardwood associates include big leaf maple (*Acer macrophyllum*), California bay (*Umbellularia californica*), tanoak (*Notholithocarpus denisflorus*), chinquapin (*Chrysolepis chrysophylla*) and Pacific madrone (Burns and Honkala 1990). Coniferous associates include Douglas-fir (*Pseudotsuga menziesii*),

western hemlock (*Tsuga heterophylla*), Sitka spruce (*Picea sitchensis*), ponderosa pine (*Pinus ponderosa*) and knobcone pine (*Pinus attenuate*). Common shrubs include coffeeberry (*Frangula californica*), huckleberry (*Vaccinium ovatum*) and creeping blackberry (*Rubus ersinus*) (Cuneo-Lazaneo and Lyons 1988). Ferns inhabiting this community include the western sword fern (*Polystichum californicum*), giant chain fern (*Woodwardia fimbriata*), coastal wood fern (*Dryopteris arguta*), goldenback fern (*Pentagramma triangularis*) and western bracken fern (*Pteridium aquilinum*).

Coast redwood communities support a wide variety of fauna, including several endangered and threatened species. Endangered birds that use this forest type for specialized nesting habitat include the marbled murrelet (*Brachyramphus marmoratus*) and the northern spotted owl (*Strix occidentalis caurina*) (Hunter et al. 1995; Ralph and Miller 1995). The marbled murrelet is predominately a seabird and depends on older coniferous stands for nesting purposes (Singer et al. 1991; Baker et al. 2006; Silvergieter and Lank 2011). With a population once estimated at 60,000, this species has dwindled roughly 90% over the last century (Ralph and Miller 1995). The northern spotted owl also roosts in coast redwood stands, preferring hollowed out snags characteristic of old-growth forests (Hunter et al. 1995; LaHaye and Gutierrez 1999).

Large woody debris (LWD) are also an important ecological component in a mature forest, providing habitat for plants and animals within the forest community (Franklin et al. 1981). Riparian species that benefit from LWD include Chinook salmon (*Oncorhynchus tshawtscha*), steelhead trout (*Oncorhynchus mykiss*) and Coho salmon (*Oncorhynchus kisutch*) (Russell 2009). In addition, LWD serves as habitat for many

species of fungi and lichen (Sawyer et al. 2000, 59). The coast redwood forest is host to over a hundred different species, which exist in the soil as well as in the forest canopy (Sawyer et al. 2000, 59; Ellyson and Sillett 2003). Many plants throughout the forest rely on symbiotic relationships with fungi present in the soil; these mycorrhizal associations aid in the uptake of water and nutrients. Research has shown that coast redwood alluvial soils support higher fungal species diversity than any other ecosystem in the world (Yaryan 2002).

### **Old-Growth Forest Characteristics**

The general definition of an old-growth forest varies among experts. According to some, the definition is based on the relative age of the trees, while others have argued that age is not as important of a factor as the physical traits associated with the structure and dynamics of the forest (Frelich and Reich 2003). Old-growth trees can exceed 115 m in height and form reiterated trunks that support a vast community of epiphytic plants (Sillett and Van Pelt 2007). Canopy complexity develops over time as second-growth forests mature, creating snags and lateral branches that serve as habitat for avian species (Oliver and Larson 1996). The structure of the canopy serves as an integral source of light distribution as well as a buffer to wind (Jennings et al. 1999).

In comparison with second-growth regions, stand density in old forests can be relatively sparse, ranging between 50 and 100 trees per hectare (Lorimer et al. 2009). Mature redwood stands have extremely high basal area which allow them to achieve dominance within the community (Gellman and Zielinski 1996). Old-growth forests are also characterized by an abundance of LWD, which serve as habitat for a number of



species and act as a buffer in streams and waterways (Franklin et al. 1981; Lienkaemper and Swanson 1987).

Herbaceous species found within mature coast redwood communities include hooker's fairybells (*Prosartes hookeri*), modesty (*Whipplea modesta*), Pacific starflower (*Trientalis latifolia*), redwood violet (*Viola sempervirens*), sorrel (*Oxalis oregana*) and Pacific trillium (*Trillium ovatum*) (Russell 2009; Russell et al. 2014). These forests are also characterized by low exotic species richness. In the absence of a major disturbance, studies have shown that exotic species decline significantly and may disappear completely in stands exceeding 60 years since harvest (Hageseth 2008).

### **Forest Disturbances**

Low to moderate disturbances within an ecosystem are a necessary component of community dynamics (Pickett and White 1985) and allow for changes in dominance while promoting species diversity (Huston 1979). Natural disturbances that occur in coast redwood forests include windthrow, flooding and fire.

Windthrow is the most common cause of mortality for coast redwoods. This is in part due to their shallow lateral root structure (Roy 1966). Redwood stands near fragmented edges are most at risk from windthrow (Russell et al. 2000). This phenomenon is also heightened when the ground is moist (McBride and Leffingwell n.d.). According to Jonsson (1993), windthrow resulting in tree mortality affects the forest structure in several ways. Initially, the fallen trees create gaps in the canopy, which provide more light saturation and allow for higher nutrient levels on the forest

floor. Fallen trees also contribute to the increase of LWD and their uprooting allows for a variance in forest floor topography.

Floods also serve as a natural disturbance in coast redwood communities although they are more prevalent in their northern range near Eel River (Stone and Vasey 1968). Following a flood, deposition of silt can sometimes bury the base of a redwood tree up to a meter high in debris. Coast redwoods are able to adapt through the development of new root systems from the submerged tree base. These roots have the ability to grow vertically to avoid saturated soil.

Coast redwood forests are also susceptible to fire and have a long fire history in the State of California. Environmental variables including slope, aspect, soil moisture, temperature and wind velocity all contribute to fire spread and intensity (Brown and Smith 2000). After being managed with fire for thousands of years by Native Americans, suppression methods began in the early 20<sup>th</sup> century (Anderson 2005). Although there has been a significant effort to suppress forest fires, smaller intensity fires occurring on an intermittent basis actually promote more diversity within the forest (Huston 1979). Suppression of these natural processes can lead to the accumulation of duff and debris, which result in higher intensity fires (Finney and Martin 1993).

### *Anthropogenic Disturbance*

Timber harvest is the primary cause of anthropogenic disturbance in coast redwood forests. Starting around the time of Anglo-American settlement, old-growth forests along the west coast began to disappear at a rapid rate. Clear-cutting was a common method of harvest, although selective-harvest was also implemented

(Barbour et al. 2001). The logging industry used a variety of methods for removal and transport of redwood lumber during this time (Sawyer et al. 2000, 27-29). Removal of logs by horse and oxen was an initial tactic that proved to be slow and laborious. The standard practice was to cut down the trees with hand axes, subsequently removing the bark and burning the undesirable portions of the tree that were left behind. The logs were then transported to the nearest stream to be floated down to the mill for processing. The availability of hand saws increased following the Civil War, boosting the economic feasibility of logging.

The implementation of steam railroad logging marked the beginning of industrial logging practices. The first steam engine, known as the steam donkey, moved large quantities of wood and was more efficient than earlier methods. After the trees were cut down, they were moved to a nearby skid trail and transported down to the railroad. Following WWII, the implementation of larger vehicles caused a transition from railroad to tractor-trucks; this resulted in a dramatic increase in the number of trees being logged (Sawyer et al. 2000, 29).

After WWII, the market for redwood lumber expanded to offer other wood products including fiberboard, pulp and particle board. This expansion led to the removal of trees that had previously been considered undesirable for lumber including felled logs, tree snags and stumps left from previous harvests. These new practices greatly reduced species richness and the structural formation of the forest ecosystem.

### *Effects of Timber Harvest*

Timber extraction has widespread impacts on forest ecosystems, affecting microhabitat variables well beyond the harvest site including species richness, solar radiation and crown dieback (Russell et al. 2000). Harvesting also changes the structure and composition of the adjacent forest area with edge effects spanning beyond 200 m (Murcia 1995; Russell et al. 2000). These changes are due in part to an increase in solar radiation, which causes evaporative stress among the adjacent redwood stands (Chen et al. 1992; Murcia 1995). As a result of long-term timber extraction, only patches of the original forest cover remain. These patches are often surrounded by timber-managed land, resulting in an edge effect around isolated stands (Russell and Jones 2001). Forests adjacent to harvested regions experience lasting edge effects in relation to dominance, basal area and canopy cover; these effects may be evident even a century after the initial disturbance (Hageseth 2008). One approach for calculating the distance of edge effects was introduced by Harris (1984), who focused on one variable, wind velocity, to establish how far the impacts of harvesting spanned to adjacent forests. Wind velocity was found to increase two to three tree heights into old-growth stands. Based on these results, Harris recommended that the consideration of edge effects be taken into account for the distance of three tree heights around the harvest site; this approach was referred to as the “three tree height rule.” Harris also suggested that to increase the effective size of an old-growth stand, the adjacent forest area should be composed of mature second-growth.

Research has also shown that changes affecting understory flora occur in the coast redwood community following logging. Studies have found that Pacific trillium

*(Trillium ovatum)*, an herbaceous species associated with old-growth forests, experiences up to 97% mortality in response to logging and may take centuries to make a full recovery (Jules 1998; Kahmen and Jules 2005; Russell and Michels 2010). Logging also has the potential to increase the presence of non-native species within forests. Exotic plants such as English ivy (*Hedera helix*) and pampas grass (*Cortaderia selloana*), which are sometimes present at the forest outskirts, are able to move further in due to the increase in solar radiation (Russell 2009).

## **Forest Recovery**

### *Adaptations to Disturbance*

The coast redwood has a number of specialized adaptations that provide both resistant and rehabilitative properties. Redwood bark contains phenolic compounds that prohibit susceptibility to fungal rot (Balogh and Anderson 1965). This species also lacks burnable resins, allowing it to withstand high intensity forest fires. Following a disturbance, vegetative sprouting is responsible for the majority of new redwood recruitments (Douhovnikoff et al. 2004). Sprouting results from the release of dormant buds from burl tissue located throughout the tree (Neal 1967; Dagley 2008). Young redwood seedlings are also capable of producing burls, which form a tissue accumulation just below the soil surface (Del Tredici 1999). In addition to releasing sprouts, burl tissue acts as a nutrient source, sending out carbohydrates and other nutrients in response to stress (Sawyer et al. 2000, 85).

### *Stages of Forest Recovery*

Oliver (1981) described forest recovery as a four step process; the stages included stand initiation, stem exclusion, understory re-initiation and old-growth. The first stage, stand initiation, occurs immediately following a disturbance. Plant re-establishment is achieved through vegetative sprouting and seed migration. The second stage, stem exclusion, occurs as tree species compete for available resources, resulting in an increase in canopy cover and a decrease in available solar radiation. Following stem exclusion, understory re-initiation is a transitional phase where self-thinning occurs within the forest; this process allows for individual stems to increase in diameter. The final stage, old-growth, is representative of mature trees which have increased in both height and diameter since the understory re-initiation phase. Understory species have also become fully acclimated and stable during this stage.

### **Successional Patterns**

Succession is a driving force in the structural and compositional dynamics of a forest. The process of succession occurs at two different levels within a given ecosystem. First, it influences the specific make-up of species that are present. Second, it directly influences other properties within the system including nutrient cycling, species diversity and productivity (Perry 1994).

### *History of Succession Theory*

Frederic E. Clements was one of the early researchers of successional theory. He described communities as “superorganisms” and believed that all communities trended

toward a “climax” which he considered to be the most mature state (Clements 1916).

The climax state, Clements believed, was determined by climate. He argued that a given community would arise when specific environmental conditions enabled it to do so.

Henry A. Gleason (1926), another prominent figure in early successional theory, rejected this idea. He felt that two main factors contributed to distinct variations between communities. First, he looked at plant species on an individual basis and argued that successional changes for each species were directly dependent on the physiognomy of that species; this idea included factors such as migration patterns and their natural distribution. Gleason also argued that a given species’ distribution was not solely determined by environmental conditions but was sometimes the result of stochasticity.

Modern ecologists generally dismiss Clements’ view of communities being superorganisms. Gleason’s concept of plants as independent entities responding to stochastic events has also been called into question. Ecologists do, however, attribute stochasticity as being a key component in successional processes. Modern theory points to disturbance variables including type and size of disturbance, seasonal time of occurrence and seed availability, all of which are randomized inputs (Perry 1994).

### *Climax Stage Theory*

Based on the Clemensian model, the climax stage theory proposes that coast redwoods are representative of the final successional stage, making them the dominant species within their range (Roy 1966). It has been argued that features including fire resistance and tolerance to poor soil conditions have allowed them to reach this status within their community (Busing and Fujimori 2002).

### *Gap-Phase Succession Theory*

The prominent regenerative pattern in coast redwood forests is gap-phase succession. This occurs when natural disturbances such as windthrow, fire, or flooding cause an opening in the forest canopy. Gaps are subsequently filled by subcanopy species, which allow for a shift in vegetative diversity (Busing and Fujimori 2002). This natural disturbance regime has served as a comparative model for many silvicultural practices including selective-harvest, variable density thinning and small group selection.

### **Coast Redwood Forest Restoration**

Studies analyzing restoration measures have found that edge effects surrounding harvest areas negatively impact forest recovery (Matlack 1993; Murcia 1995; Jules 1998; Russell and Jones 2001; Kahmen and Jules 2005). To counter these effects it is recommended by some researchers that land managers focus on implementing the smallest perimeter-to-area ratio possible to maintain the integrity of the forest (Russell and Jones 2001). In addition, research has suggested that a buffer zone be implemented between the harvest site and the adjacent forest to minimize structural and compositional changes following logging (Russell et al. 2000).

### *Thinning for Management*

A controversial restoration practice that has gained popularity in recent years is the implementation of thinning to speed up the emergence of spatial features associated with old-growth forests. Although thinning has been promoted as a tool for restoration, this practice has been found to be counterproductive (Russell et al. 2014). Research has



shown that density increases substantially in thinned stands in comparison with naturally regenerating forests, which progress naturally through self-thinning. As O'Hara et al. (2010, 131) suggested in a recent study, "Mean size of all trees greater than 1.37 m height increased only in the control and was slightly smaller in the thinning treatments. This was a product of the high levels of ingrowth in the thinned treatments that reduced average tree size, whereas self-thinning in the control plots tended to increase average tree size." Although it has not received as much attention in academic literature, selective-harvest is similar to thinning in many respects. Therefore, further research should be conducted on the potential impacts of selective-harvest in an effort to determine if this practice results in a similar outcome.

### **PROBLEM STATEMENT**

Old-growth coast redwood forests have been significantly altered by logging activities, resulting in highly fragmented second-growth stands throughout their range. Although these second-growth forests have made a rebound from the highly destructive logging practices of the 19<sup>th</sup> and 20<sup>th</sup> centuries, the recovery of this forest system as a whole is not well understood.

Historically, clear-cutting was the predominant method of timber extraction in coast redwood forests. In recent years, selective-harvest has become more common and is currently the only harvest method employed in Santa Cruz County where clear-cutting is no longer permitted. Although this practice is generally considered less damaging, it still may have lasting impacts on forest composition and structure.

## **Research Objective**

The objective of this study was to determine how the composition and structure of selectively-harvested coast redwood stands were influenced by the following independent variables: years since last harvest, number of harvest re-entries and the percentage cut per hectare.

## **Research Questions**

1. Which independent variable had the strongest influence on canopy cover, stand density and density of large woody debris (LWD)?
2. Which independent variable had the strongest influence on basal area and dominance of coast redwood?
3. Which independent variable had the strongest influence on floristic composition, including coast redwood associated species and exotic species?

## **Hypotheses**

The following predictions were made in regard to the research questions above:

- H<sub>1</sub> The percentage cut per hectare would be the strongest independent variable for canopy cover, stand density and density of LWD.
- H<sub>2</sub> Basal area and dominance of *Sequoia sempervirens* would decrease in relation to the number of harvests and percentage harvested, but would increase in relation to years since last harvest.
- H<sub>3</sub> The percentage cut per hectare would be the strongest independent variable for floristic composition, including coast redwood associated species and exotic species.

## METHODS

### Study System

Santa Cruz County has a Mediterranean climate typical of California's coast, defined by hot, dry summers and high precipitation during the winter. Annual rainfall is between 100-150 cm and morning fog is common during the summer months.

Common coast redwood associate hardwoods in this region include California hazelnut (*Corylus cornuta*), California box elder (*Acer negundo*), California bay, Pacific madrone, big leaf maple (*Acer macrophyllum*), coast live oak (*Quercus agrifolia*) and interior live oak (*Quercus wislizeni*) (Cooney-Lazaneo and Lyons 1981; Lyons and Cuneo-Lazaneo 1988). Common coniferous associates include California nutmeg (*Torreya californica*), Douglas-fir and knobcone pine (*Pinus attenuata*).

There are a variety of avian, mammalian and amphibian species that occupy Santa Cruz County. Birds found in this habitat include the Cooper's hawk (*Accipiter cooperii*), golden eagle (*Aquila chrysaetos*), long-eared owl (*Asio otus*), marbled murrelet, peregrine falcon (*Falco peregrinus*), osprey (*Pandion haliaetus*), sharp-shinned hawk (*Accipiter striatus*) and yellow warbler (*Dendroica petechial*) (California Department of Forestry and Fire Protection 2001). Mammals within the region include the dusky-footed woodrat (*Neotoma fuscipes*), Mexican black-tailed deer (*Odocoileus hemionus*), mountain lion (*Puma concolor*), raccoon (*Procyon lotor*) and deer mouse (*Peromyscus maniculatus*). Amphibian species present in this range include the California red-legged frog (*Rana aurora draytonii*), foothill yellow-legged frog (*Rana boylei*) and southwestern

pond turtle (*Actinemys marmorata pallida*) (California Department of Forestry and Fire Protection 2001; Bulger et al. 2003).

Selective-harvest has been the required method of timber extraction in Santa Cruz County since clear-cutting was banned in 1970; this was in adherence to specific regulations applicable to the Southern Subdistrict under the California Forest Practice Rules (California Department of Forestry and Fire Protection 2013). The coast redwood forests in this region are predominately second-growth with sparse patches of old-growth occurring intermittently throughout the county.

The study sites for this research included five different harvests within the Byrne-Milliron Forest, located in Santa Cruz County (Figure 2). The forest is located five miles north of Corralitos and is composed of coast redwood, mixed chaparral, montane hardwood/coastal scrub and eucalyptus. Clear-cut in the late 1880s, the property predominately consists of even-aged coast redwood and Douglas-fir (California Department of Forestry and Fire Protection 2001).

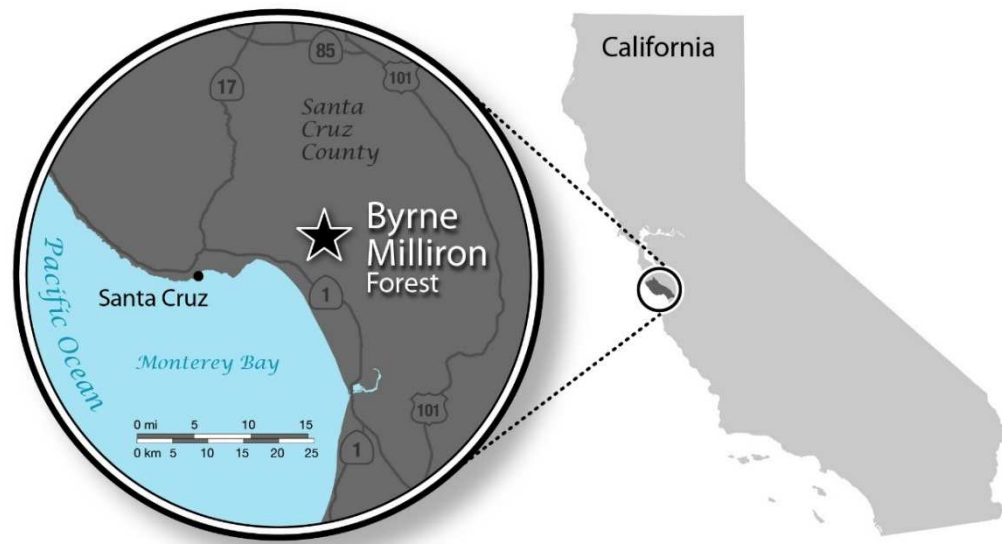


Figure 2. Study system. Image by Derrick Wynes.

The preserve spans 163 hectares with elevations ranging between 125-500 m and predominately west facing slopes (Overtree and Kitayama 2013). Situated atop the Purisima Formation, the soil types on the preserve include the Ben Lomond-Felton complex, Lompico-Felton complex, Nisene-Aptos complex and a small portion of Pfeiffer gravelly sandy loam located near the entrance (California Department of Forestry and Fire Protection 2001). Several small tributaries on the property lead to the nearby Browns Creek, although only one runs continuously throughout the year.

Named for its original owner Carlton Byrne, the preserve was acquired by the Land Trust of Santa Cruz County in 1984. The Land Trust initially only purchased the Byrne property, which consisted of 130 hectares. In 2007, the Milliron property, an additional 32 hectares, was acquired through a conservation easement and then later

purchased in 2008 (Overtree and Kitayama 2013). For the purpose of this study the Milliron property has been excluded.

There have been a total of seven selective harvests on the preserve since the Land Trust took ownership. The property is now broken up into several management units (Figure 3) including the Central Unit (56 hectares), the Early Successional Unit (22 hectares), the Late Successional Unit (15 hectares) and the Southern Unit (24 hectares). Many of these units have undergone varying levels of cutting intensity implemented with the use of either tractor, skyline cable, or both methods combined (Table 1).

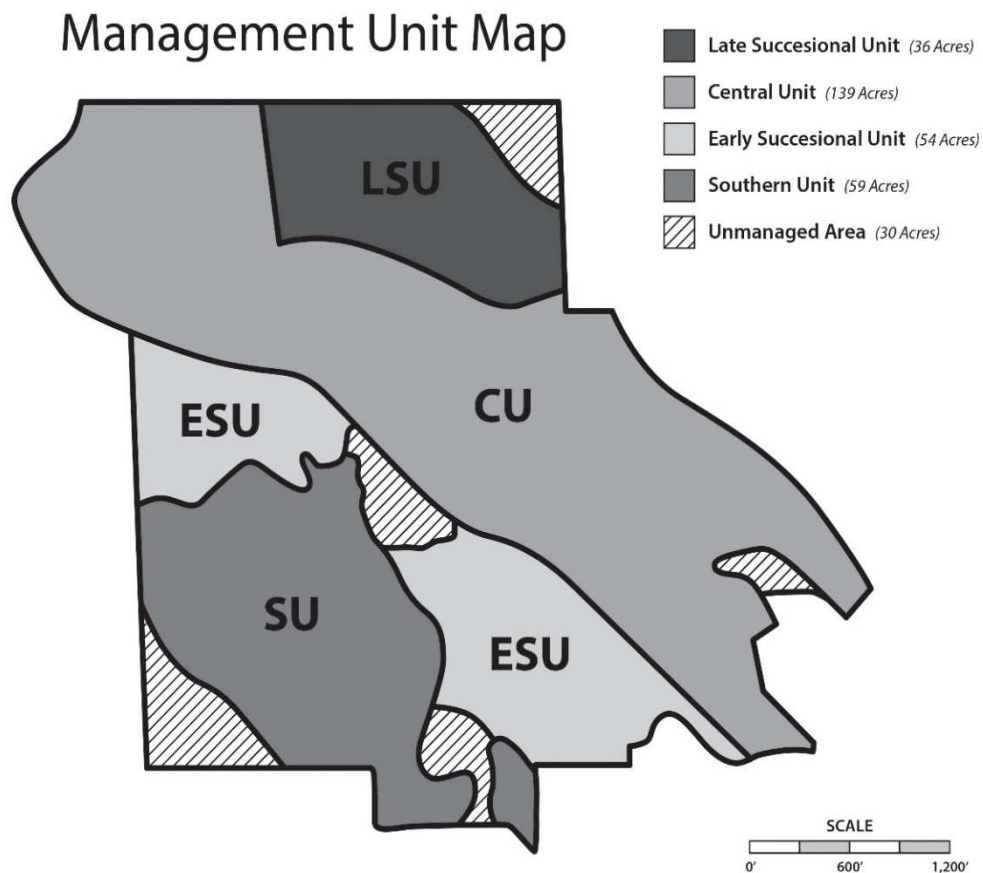


Figure 3. Management units at Byrne-Milliron Forest. Image by Derrick Wynes.

Table 1. Harvest details among study sites.

	1987/2004	1990/2007	1996	2001	2007
Unit name <sup>1</sup>	CU	CU	SU	CU	LSU
Harvest area (Ha)	22; 19	24; 17	26	12	15
Harvest methods <sup>2</sup>	T	S	T,S	T	T,S
Years since harvest	9	6	17	12	6
Number of re-entries	2	2	1	1	1
Percentage cut <sup>3</sup>	22.78	32.49	25.45	29.47	17.48

Five of the selective-harvests have occurred in the Central Unit; two of these sites were harvested twice. The first site was harvested in 1987 and re-entered in 2004 and the second site was harvested in 1990 and re-entered in 2007. Both initial harvests were completed under Timber Management Plans (THP) and the latter were completed under one Non-Industrial Timber Management Plan (NTMP) prepared in 1991 (Overtree and Kitayama 2013). All harvests after 1996 were approved under this plan. The fifth harvest occurred in 2001 and was the first entry in that region since the clear-cut in the 1880s (California Department of Forestry and Fire Protection 2001). Other selective harvests completed include one in the Southern Unit, which was entered in 1996, as well as one harvest in the Late Successional Unit, entered in 2007 for the first time since the clear-cut. Unlike the other management units at Byrne-Milliron Forest, this harvest area has different management goals, which include minimal logging and an overall reduction in disturbance in an effort to promote late successional features and maintain habitat features such as snags and LWD. All harvests completed under THPs had subsequent plantings following the logging operations. Primary goals for the forest initially set forth

<sup>1</sup> CU = Central Unit; SU = Southern Unit; LSU = Late Successional Unit

<sup>2</sup> S = skyline cable; T = tractor

<sup>3</sup> Most recent harvest

by Carlton Byrne and since upheld by the Land Trust include sustainable forestry, recreation and natural resource protection.

### **Study Design**

Data collection for the study began in May 2012 and ended in July 2013. Site selection was determined by analyzing ground accessibility, timber harvest maps and applicable THPs and NTMPs. The replicated sample design consisted of five second-growth sites that had undergone selective-harvest since the property was acquired by the Land Trust of Santa Cruz County in 1984. Twenty 0.032 hectare (20 m diameter) sample plots were randomly selected within each of the five sites (Figure 4). All plot locations were situated at least 10 m from sensitive habitats and 200 m from main access roads (Russell and Michels 2010). Plot locations were selected at random with the use of a random number generator and coordinates were recorded with a Garmin GPS device. Each plot was further divided into sample quadrants to determine relative herbaceous cover.



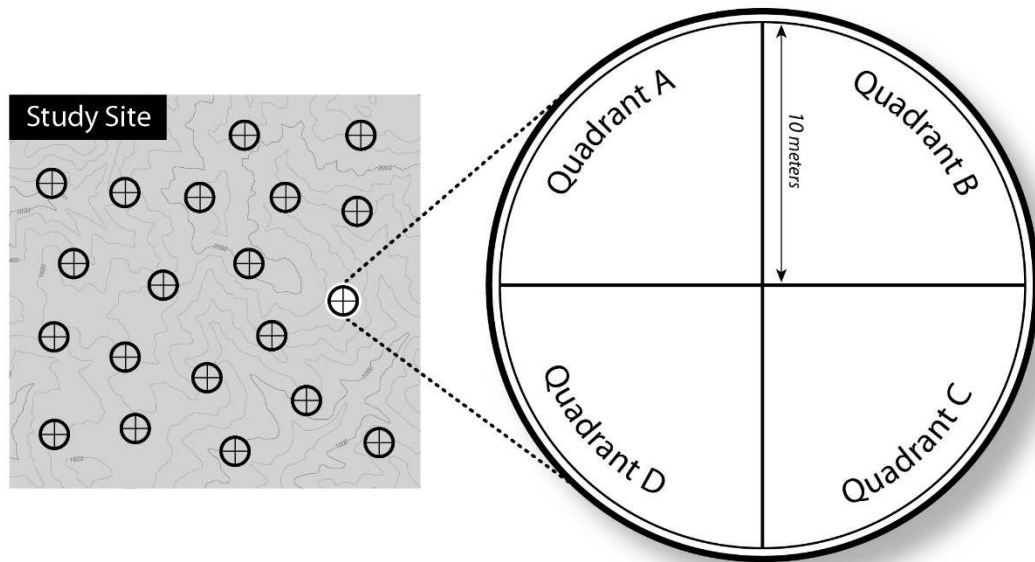


Figure 4. Site Schematic and 20-Meter Diameter Sample Plot. Image by Derrick Wynes.

### Data Collection

Physiographic variables recorded at each plot included slope, aspect and elevation. All tree species >10 cm were counted and measured using a DBH (diameter at breast height) tape. Seedlings and sprouts were also tallied, identified and categorized by whether or not they exceeded one meter in height. Canopy cover was determined using a convex spherical densiometer, with readings taken in each of the four cardinal directions. LWD was also counted and circumference and length were recorded for each occurrence. Herbaceous species within each plot were identified and visual estimates were made to determine relative species composition. In the event of an unidentifiable plant, a sample of the specimen was collected for later identification using the Jepson Manual (Hickman 1993).

## **Analytical Methods**

A Pearson's product-moment correlation matrix was used to analyze relationships between the independent variables (years since last harvest, number of harvest re-entries and the percentage cut per hectare) and dependent variables (aspect, elevation, slope, canopy cover, stand density, size classes of coast redwood and tanoak, size and number of LWD, native and exotic species richness, coast redwood associated species, percentage of broad leaved helleborine, forget-me-not and English holly, as well as basal area and dominance of coast redwood). To look at stem size variation in further depth, coast redwood and tanoak were divided into seven different size classes: <10 cm, 10-24 cm, 25-49 cm, 50-99 cm, 100-149 cm, 150-199 cm and >200 cm DBH (Giusti 2007). LWD was also broken up into size classes including short (2-8 m), medium (9-15 m) and long (>15 m) (Sinclair 2013). In addition, native, exotic and coast redwood associated herbaceous species were examined to determine species prevalence and richness at each site. One-way analysis of variance (ANOVA) were conducted to determine whether there were significant differences between each of the earlier mentioned dependent variables and the number of harvests, percentage harvested and date of last harvest re-entry. Post hoc analyses used the Bonferroni correction. All statistical analyses were conducted using IBM SPSS Statistics 21 and Microsoft Office Excel 2013.

## **RESULTS**

There were a number of variations in forest structure and composition among study sites, with significant differences determined for canopy cover, stand density, density of LWD, coast redwood sprouts, coast redwood associated herbaceous species

and exotic species richness. Physiographic variables including aspect, canopy cover, elevation and slope were relatively similar among sites (Table 2). The median aspect was 42° in the 1987/2004 site, 71° in the 1990/2007 site, 90° in the 1996 site, 68° in the 2001 site and 84° in the 2007 site, respectively.

Table 2. Physiographic characteristics among study sites.

Physiographic Variables	1987/2004		1990/2007		1996		2001		2007	
	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>
Canopy cover	0.97	0.004	0.98	0.002	0.93	0.02	0.98	0.004	0.98	0.002
Elevation	1213	25.42	928	19.68	1071	20.68	1475	21.46	1094	30.24
Slope (%)	41.9	2.97	36.05	1.72	39	2.84	41.6	2.62	39.8	3.18

### Canopy Cover

The 2007 site had the highest canopy cover, followed by the 1990/2007 site, the 2001 site, the 1987/2004 site and the 1996 site (Figure 5). When years since harvest was analyzed, ANOVA indicated that the 1996 site was significantly different from all other harvest sites including the 2007 site ( $p = <0.001$ ), the 1987/2004 site ( $p = 0.006$ ) and the 2001 site ( $p = 0.002$ ); a Pearson's product-moment correlation coefficient indicated this was a weak negative relationship ( $r = -0.04$ ). Analysis of canopy cover and the percentage cut per hectare also indicated that the 1996 harvest site was significantly different from all other sites including the 2007 site ( $p = 0.001$ ), the 1987/2004 site ( $p = 0.01$ ), the 2001 site ( $p = 0.004$ ) and the 1990/2007 site ( $p = 0.004$ ). Again, this was a weak negative relationship ( $r = -0.04$ ). Analysis of canopy cover and number of harvest re-entries indicated there was not a significant difference among sites ( $p = 0.2$ ).

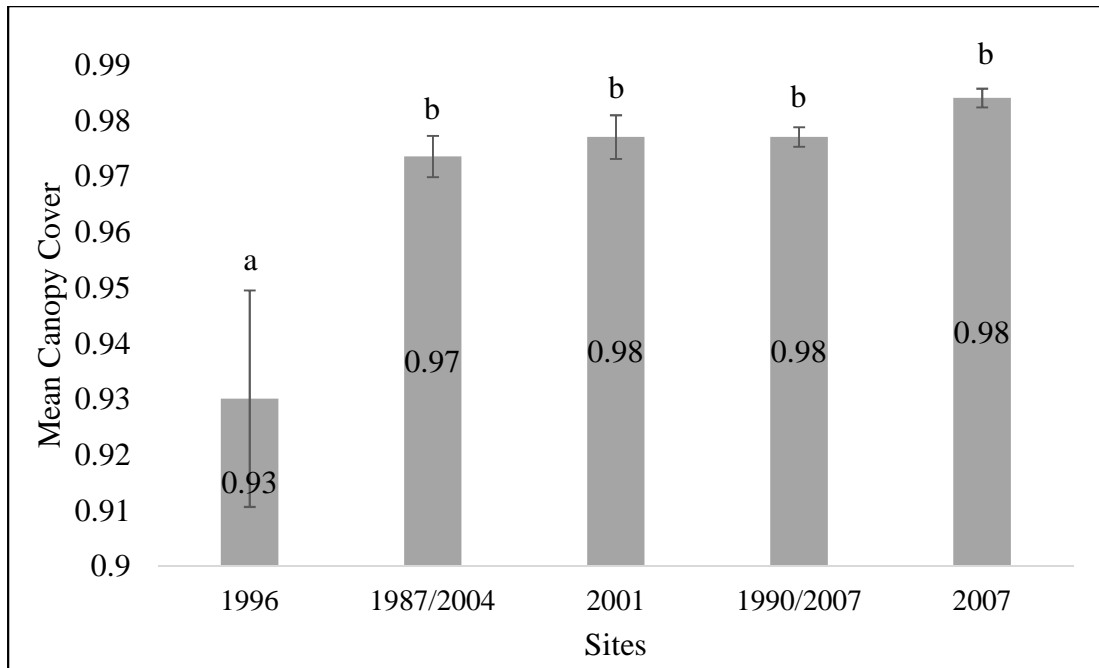


Figure 5. Mean canopy cover among study sites with 95% confidence intervals (sites with the same letter were not significantly different from one another).

### Stand Density

Mean species density varied among study sites (Table 3). Species identified at the Byrne-Milliron Forest included coast redwood, tanoak (*Lithocarpus densiflorus*), Pacific madrone, Douglas-fir and *Quercus ssp.*, which were determined to either be coast live oak (*Quercus agrifolia*) or interior live oak (*Quercus wislizenii*) (Figure 6). The 1990/2007 harvest site had the highest stand density of all sites, followed by the 1987/2004 site, the 1996 site, the 2007 site and the 2001 site (Figure 7).

Table 3. Mean stand density among study sites.

Stand Density	1987/2004		1990/2007		1996		2001		2007	
	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>
All species	11.9	0.96	12.85	1.41	11.15	0.98	5.95	1.06	7.05	0.68
Coast redwood	10.1	0.81	11.25	1.24	9.1	1.09	8.85	1.04	6.55	0.69
Tanoak	49.15	11.49	69.15	14.67	49.65	8.61	25.95	5.78	16.85	2.69
Pacific madrone	0.6	0.5	0.1	0.07	0.3	0.15	3.1	1.44	1.25	0.95
Oak	2.35	1.25	4.6	1.22	5.4	3.12	0.55	0.26	0.65	0.25
Douglas-fir	0.4	0.28	0.4	0.28	0	0	0.6	0.29	0	0
Big-leaf maple	0.05	0.05	0.05	0.05	0.4	0.35	0	0	0.35	0.3

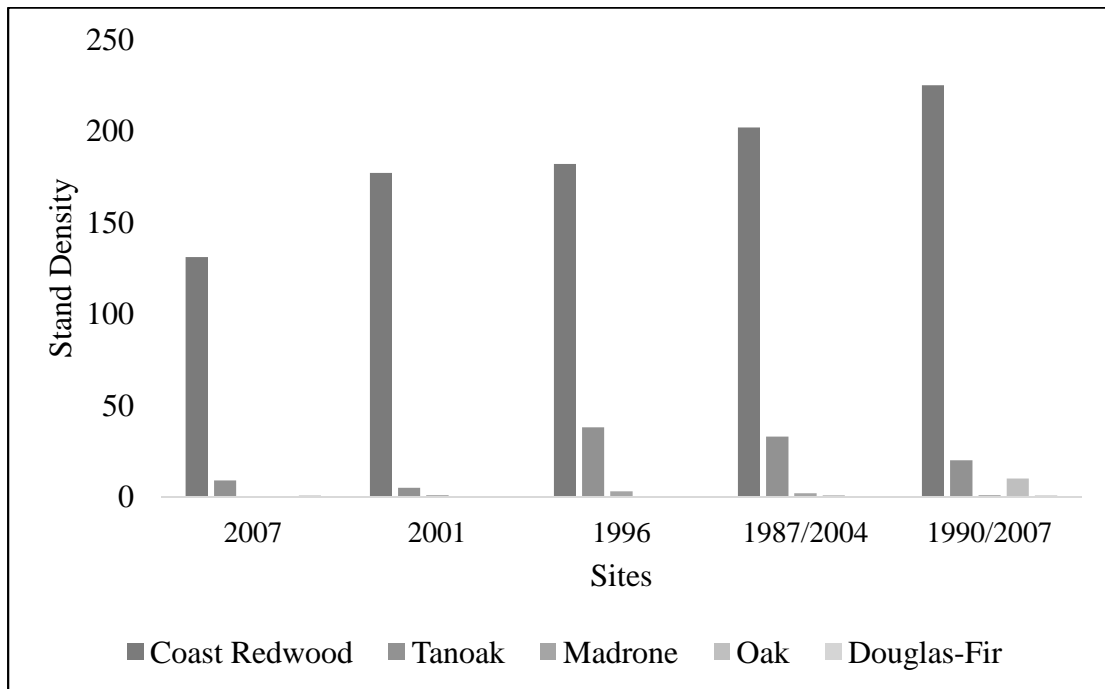


Figure 6. Frequency of tree species among study sites.

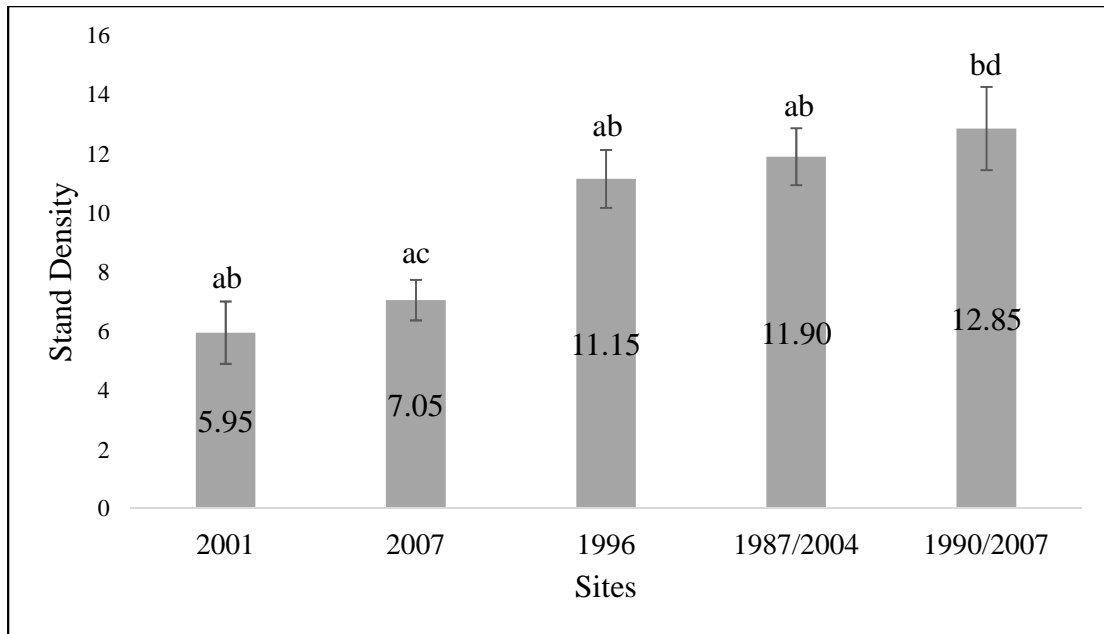


Figure 7. Mean stand density among study sites with 95% confidence intervals (sites with the same letter were not significantly different from one another).

When ANOVA was used to analyze stand density and years since last harvest, results showed there was not a significant difference among study sites ( $p = 0.052$ ). However, when analyzed with number of harvest re-entries, there was a significant difference between one and two re-entries ( $p = 0.001$ ); this was a positive relationship according to Pearson’s product-moment correlation coefficient ( $r = 0.15$ ). When stand density was analyzed with the percentage cut per hectare, the 2007 site had a significantly lower density than the 1990/2007 site ( $p = 0.001$ ). Stand density had a positive relationship with this independent variable ( $r = 0.27$ ).

In addition to having the highest mean stand density, the 1990/2007 site also had the highest mean density of coast redwood, followed by the 1987/2004 site, the 1996 site, the 2001 site and the 2007 site. When coast redwood density was analyzed with years since last harvest, a one-way ANOVA indicated that the difference among sites was not

significant ( $p = 0.8$ ). When analyzed with number of harvest re-entries, there was a significant difference between sites re-entered once versus sites re-entered twice ( $p = 0.007$ ). A Pearson's product-moment correlation coefficient indicated this relationship was positive ( $r = 0.1$ ). There were also significant findings for coast redwood density and the percentage cut per hectare, with results indicating that the 2007 site had a significantly lower density of coast redwood in comparison with the 1990/2007 site ( $p = 0.012$ ); this relationship was positive ( $r = 0.26$ ).

Density of tanoak varied across sites. The 1990/2007 site had the highest mean, followed by the 1996 site, the 1987/2004 site, the 2001 site and the 2007 site. Mean tanoak density was determined to have a significant positive relationship with years since harvest ( $p = 0.049$ ;  $r = 0.15$ ), but was not significantly different among sites when analyzed with number of harvest re-entries ( $p = 0.32$ ) or the percentage cut per hectare ( $p = 0.076$ ).

Other observations included Pacific madrone, oak, Douglas-fir and big leaf maple. Density of Pacific madrone was highest in the 2001 site, followed by the 2007 site, the 1987/2004 site, the 1996 site and the 1990/2007 site. Mean density of oak was highest in the 1996 site, followed by the 1990/2007 site, the 1987/2004 site, the 2007 site and the 2001 site. Douglas-fir was scarce among sites, with the 2001 site having the highest mean density, followed by the 1990/2007 site and the 1987/2004 site. There were no observations of this species in either the 1996 or 2007 sites. Big leaf maple was also not very prevalent, with a mean  $<0.50$  for all sites. The 1996 site had the highest density

of big-leaf maple, followed by the 2007 site, the 1987/2004 site and the 1990/2007 site; there were no observations in the 2001 site.

### **Coast Redwood Size Classes**

Coast redwood size class distribution varied among sites (Figure 8; Table 4). The 1987/2004 site had the highest number of stems <10 cm, followed by the 1996 site, the 2001 site, the 1990/2007 site and the 2007 site. When analyzed with years since harvest, a one-way ANOVA indicated a significant difference between the 1987/2004 site in comparison with the 2007 site ( $p = 0.002$ ) and the 2001 site ( $p = 0.046$ ). A Pearson's product-moment correlation coefficient indicated a positive relationship between years since last harvest and coast redwoods <10 cm ( $r = 0.18$ ).

Although there was not a significant difference among sites for number of harvest re-entries and stems <10 cm ( $p = 0.19$ ), there were significant findings for the percentage cut per hectare and coast redwood in this size class. The 1987/2004 site had a significantly higher number of stems in comparison with the 2007 site ( $p = 0.008$ ) and the 1990/2007 site ( $p = 0.03$ ); this relationship was negative ( $r = -0.05$ ).

The 1990/2007 site had the highest number of coast redwood 10-24 cm, followed by the 1987/2004 site, the 2001 site, the 1996 site and the 2007 site. A one-way ANOVA indicated there was not a significant difference among sites for stems in this size class and years since last harvest ( $p = 0.15$ ). However, when analyzed with number of harvest re-entries, stems 10-24 cm were significantly lower in sites with one re-entry compared with two re-entries ( $p = <0.001$ ); this relationship was positive ( $r = 0.22$ ). The 1996 site,



the 2001 site and the 2007 site, which were only re-entered once, had a lower number of stems 10-24 cm in comparison with the sites re-entered twice.

When ANOVA was used to analyze the percentage cut per hectare, the mean number of stems 10-24 cm in the 2007 site was significantly lower than the 1987/2004 site ( $p = <0.001$ ), the 1990/2007 site ( $p = <0.001$ ), the 1996 site ( $p = 0.046$ ) and the 2001 site ( $p = 0.038$ ). A Pearson's product-moment correlation coefficient indicated this relationship was positive ( $r = 0.36$ ).

The 1990/2007 site had the highest mean number of coast redwood 25-49 cm, followed by the 1987/2004 site, the 1996 site, the 2001 site and the 2007 site. When ANOVA was used to analyze this size class with years since last harvest, there was not a significant difference among sites ( $p = 0.069$ ). There was also not a significant difference among sites for number of harvest re-entries ( $p = 0.35$ ) or the percentage cut per hectare ( $p = 0.74$ ).

The 1990/2007 site had the highest mean number of coast redwood 50-99 cm, followed by the 1996 site, the 2001 site, the 2007 site and the 1987/2004 site. There was not a significant difference for coast redwood stems in this size class when analyzed with years since last harvest ( $p = 0.71$ ), number of harvest re-entries ( $p = 0.42$ ) or the percentage cut per hectare ( $p = 0.37$ ).

The 1987/2004 site had the highest mean number of stems 100-149 cm, followed by the 1996 site, the 2007 site, the 2001 site and the 1990/2007 site. There were not significant findings for coast redwood stems in this size class when analyzed with years

since last harvest ( $p = 0.76$ ), number of harvest re-entries ( $p = 0.55$ ) or the percentage cut per hectare ( $p = 0.71$ ).

The 2007 site had the highest mean number of coast redwood 150-199 cm, followed by the 1987/2004 site and the 2001 site. There were no observations of this size class found in the 1996 or 1990/2007 sites. When analyzed with years since last harvest, results indicated there was not a significant difference among sites ( $p = 0.16$ ). There was also not a significant difference among sites for number of harvest re-entries ( $p = 0.09$ ). However, there were significant findings for this size class when analyzed with the percentage cut per hectare. The 2007 site had a significantly larger number of stems in comparison with the 1990/2007 site ( $p = 0.002$ ), the 1996 site ( $p = 0.002$ ), the 1987/2004 site ( $p = 0.018$ ) and the 2001 site ( $p = 0.018$ ); this was a negative relationship ( $r = -0.35$ ).

There were only a small number of occurrences where stems exceeded 200 cm. This included one occurrence in the 2001 site and one occurrence in the 1990/2007 site. When ANOVA was used to analyze stems in this size class with years since last harvest, these occurrences were not found to be significant ( $p = 0.63$ ). Results for stems >200 cm and number of harvest re-entries were also not significant ( $p = 0.78$ ). In addition, there was not a significant difference found when analyzed with the percentage cut per hectare ( $p = 0.56$ ).

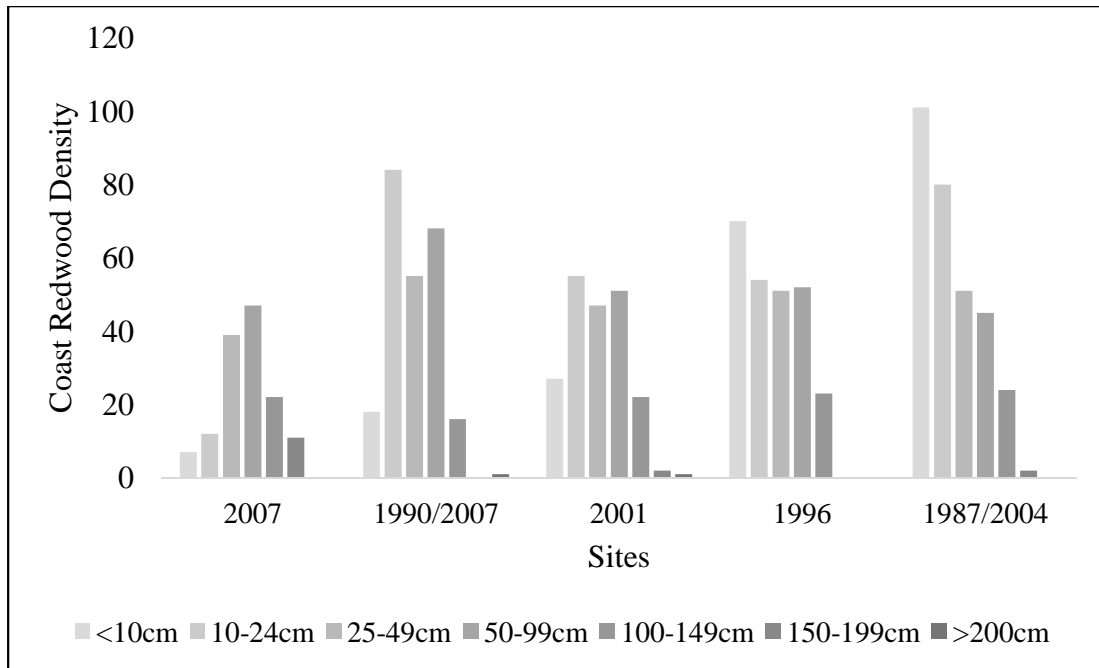


Figure 8. Coast redwood size class distribution among study sites.

Table 4. Means and standard errors for coast redwood size classes.

Coast Redwood Size Classes	1987/2004		1990/2007		1996		2001		2007	
	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>
<10 cm	5.05	1.45	0.90	0.39	3.50	1.47	1.35	0.44	0.35	0.22
10-24 cm	4.00	0.62	4.20	0.53	2.70	0.48	2.75	0.60	0.60	0.24
25-49 cm	2.55	0.40	2.75	0.49	2.55	0.54	2.35	0.36	1.95	0.33
50-99 cm	2.25	0.32	3.40	0.64	2.60	0.39	2.55	0.40	2.35	0.36
100-149 cm	1.20	0.26	0.80	0.19	1.15	0.21	1.10	0.23	1.10	0.18
150-199 cm	0.10	0.10	0	0	0	0	0.10	0.07	0.55	0.18
>200 cm	0	0	0.05	0.05	0	0	0.05	0.05	0	0

### Tanoak Size Classes

Tanoak size classes varied among sites, with stems <10 cm occurring with the highest prevalence (Table 5). The 1990/2007 site had the highest number of stems in this size class, followed by the 1996 site, the 1987/2004 site, the 2001 site and the 2007 site.

When ANOVA was used to analyze years since last harvest, there was not a significant

difference among sites ( $p = 0.21$ ). However, there were significant findings for tanoak stems <10 cm and number of harvest re-entries ( $p = 0.004$ ). A Pearson's product-moment correlation coefficient indicated this was a weak positive relationship ( $r = 0.05$ ). When tanoak <10 cm was analyzed with the percentage cut, there was a significant difference between the 1990/2007 site in comparison with the 2007 site ( $p = 0.004$ ) and the 2001 site ( $p = 0.009$ ); this relationship was positive ( $r = 0.24$ ).

Tanoak 10-24 cm had the highest density in the 1996 harvest site, followed by the 1990/2007 site, the 1987/2004 site and the 2007 site. There were no observations of this size class present in the 2001 site. When analyzed with years since last harvest, a one-way ANOVA indicated there was a significant difference between the 2001 site and the 1996 site ( $p = 0.006$ ); this relationship was positive ( $r = 0.21$ ). There was not a significant difference for tanoak 10-24 cm and number of harvest re-entries ( $p = 0.56$ ). However, there was a significant difference among sites when this size class was analyzed with the percentage cut per hectare. This was a positive relationship ( $r = 0.12$ ). A post hoc analysis following ANOVA indicated a significant difference between the 1996 site in comparison with the 2001 site ( $p = 0.008$ ) and the 2007 site ( $p = 0.021$ ).

The 1987/2004 site had the highest mean of tanoak 25-49 cm, followed by the 1996 site, the 2007 site and the 1990/2007 site. There were no observations of this size class in the 2001 site. A one-way ANOVA determined there was not a significant difference among sites for tanoak 25-49 cm and years since last harvest ( $p = 0.24$ ), number of harvest re-entries ( $p = 0.41$ ) or the percentage cut per hectare ( $p = 0.38$ ).

The 1987/2004 site had the highest mean number of tanoak 50-99 cm, followed by the 1996 site, the 2001 site and the 1990/2007 site. There were no tanoak in this size class observed in the 2007 harvest site. When ANOVA was used to analyze years since last harvest, the 2007 site had significantly fewer tanoak in comparison with the 1987/2004 site ( $p = 0.001$ ), the 2001 site ( $p = 0.023$ ) and the 1996 site ( $p = 0.023$ ); this was a weak negative relationship ( $r = -0.003$ ). Tanoak in the 50-99 cm size class were also significantly different among sites when the number of re-entries was analyzed using ANOVA ( $p = 0.026$ ). According to Pearson's product-moment correlation coefficient, this was a positive relationship ( $r = 0.38$ ). When analyzed with the percentage cut per hectare, the number of stems in this size class was significantly higher in the 1987/2004 site in comparison with the 2007 site ( $p = 0.006$ ), the 1996 site ( $p = 0.04$ ), the 2001 site ( $p = 0.04$ ) and the 1990/2007 site ( $p = 0.016$ ); this relationship was negative ( $r = -0.06$ ).

There was only one occurrence of a tanoak in the 100-149 cm size class, which was observed in the 2001 harvest site. There were no observations of tanoak in the last two size classes among any of the study sites.

Table 5. Means and standard errors for tanoak size classes.

Tanoak Size Classes	1987/2004		1990/2007		1996		2001		2007	
	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>
<10 cm	16.4	4.47	25.9	6.08	18.25	3.79	6.75	2.20	5.45	1.217
10 - 24 cm	0.40	0.17	0.80	0.33	1.20	0.35	0.05	0.05	0.15	0.11
25 - 49 cm	0.85	0.56	0.30	0.18	0.65	0.31	0	0	0.35	0.25
50 - 99 cm	0.60	0.24	0.05	0.05	0.10	0.07	0.10	0.07	0	0
100 - 149 cm	0	0	0	0	0	0	0.05	0.05	0	0
150 - 199 cm	0	0	0	0	0	0	0	0	0	0
> 200 cm	0	0	0	0	0	0	0	0	0	0

## Large Woody Debris

The density of LWD varied among sites. The 2007 site had the most LWD per plot, followed by the 1987/2004 site, the 1990/2007 site, the 2001 site and the 1996 site (Figure 9; Table 6).

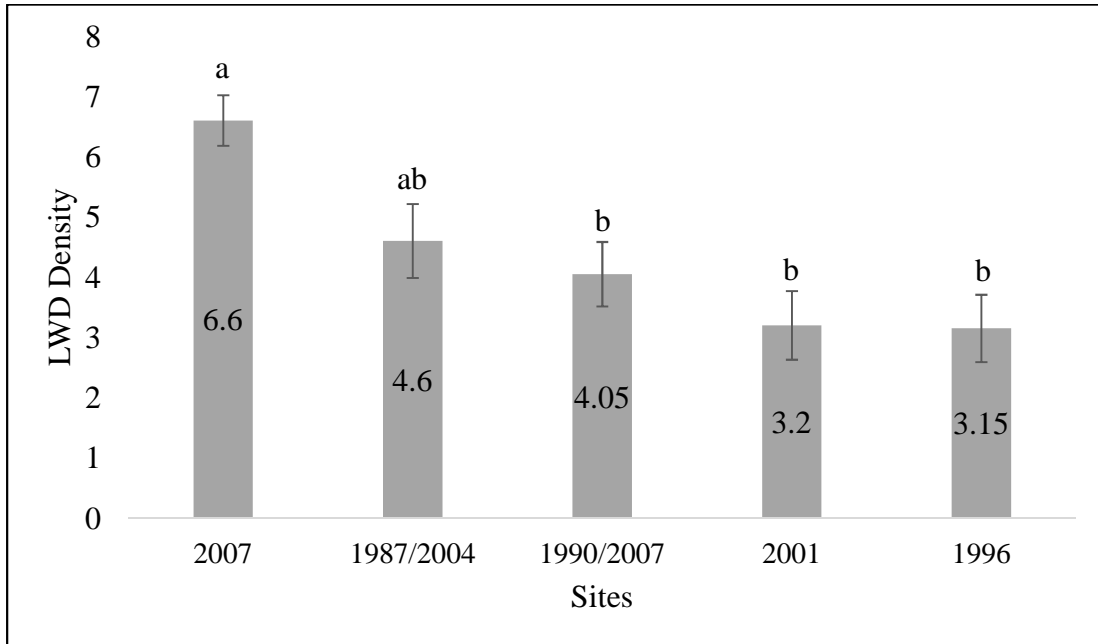


Figure 9. Mean large woody debris (LWD) among study sites with 95% confidence intervals (sites with the same letter were not significantly different from one another).

Table 6. Means and standard errors for large woody debris (LWD) density.

LWD Density	1987/2004		1990/2007		1996		2001		2007	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
Short LWD	4.05	0.61	3.70	0.52	2.50	0.44	2.60	0.50	6.35	0.82
Med. LWD	0.35	0.15	0.15	0.08	0.35	0.13	0.40	0.11	0.85	0.27
Long LWD	0.10	0.07	0.20	0.12	0.25	0.14	0.15	0.11	0.25	0.10
Cumulative LWD	4.60	0.61	4.05	0.54	3.15	0.56	3.20	0.57	6.60	0.42

When analyzed with years since last harvest, a one-way ANOVA indicated that LWD density was significantly higher in the 2007 site compared with the 1996 site

( $p = 0.015$ ) and the 2001 site ( $p = 0.016$ ). A Pearson's product-moment correlation coefficient indicated the relationship between mean LWD density and years since last harvest was negative ( $r = -0.32$ ). Results were not significant when analyzed with number of harvest re-entries ( $p = 0.9$ ). However, analysis of LWD density and the percentage cut per hectare showed a significant difference between the 2007 site in comparison with the 1996 site ( $p = <0.001$ ), the 2001 site ( $p = <0.001$ ) and the 1990/2007 site ( $p = 0.013$ ). A Pearson's product-moment correlation coefficient indicated a negative relationship between density of LWD and the percentage cut per hectare ( $r = -0.36$ ).

When size classes for LWD were evaluated, the 2007 site had the highest number of short LWD, followed by the 1987/2004 site, the 1990/2007 site, the 2001 site and the 1996 site (Figure 10). When ANOVA was used to analyze years since last harvest, the 2007 site had a significantly higher number of short LWD compared with the 1996 site ( $p = 0.008$ ) and the 2001 site ( $p = 0.012$ ); this relationship was negative ( $r = -0.34$ ). When analyzed with number of harvest re-entries, ANOVA indicated there was not a significant difference between one versus two re-entries ( $p = 0.92$ ). When short LWD was analyzed with the percentage cut per hectare, the 2007 site had a significantly higher amount in comparison with the 1996 site ( $p = <0.001$ ), the 2001 site ( $p = <0.001$ ) and the 1990/2007 site ( $p = 0.022$ ). According to Pearson's product-moment correlation coefficient, the relationship between these two variables was negative ( $r = -0.35$ ).

The 2007 site also had the most LWD in the medium size class, followed by the 2001 site, the 1987/2004 site, the 1996 site and the 1990/2007 site. A post hoc analysis

following ANOVA indicated there was not a significant difference among sites for medium LWD and years since harvest ( $p = 0.85$ ) or number of harvest re-entries ( $p = 0.066$ ). There was, however, a significant difference found for medium LWD and the percentage cut per hectare, with the 2007 site having a significantly higher amount of LWD in this size class in comparison with the 1990/2007 site ( $p = 0.032$ ). A Pearson's product-moment correlation coefficient indicated this was a negative relationship ( $r = -0.27$ ).

Analysis of long LWD indicated the 1996 site had the same mean as the 2007 site, followed by the 1990/2007 site, the 2001 site and the 1987/2004 site (Figure 10). Further analysis using ANOVA indicated there was not a significant difference among sites for long LWD and years since last harvest ( $p = 0.73$ ), number of harvest re-entries ( $p = 0.5$ ) or the percentage cut per hectare ( $p = 0.84$ ).

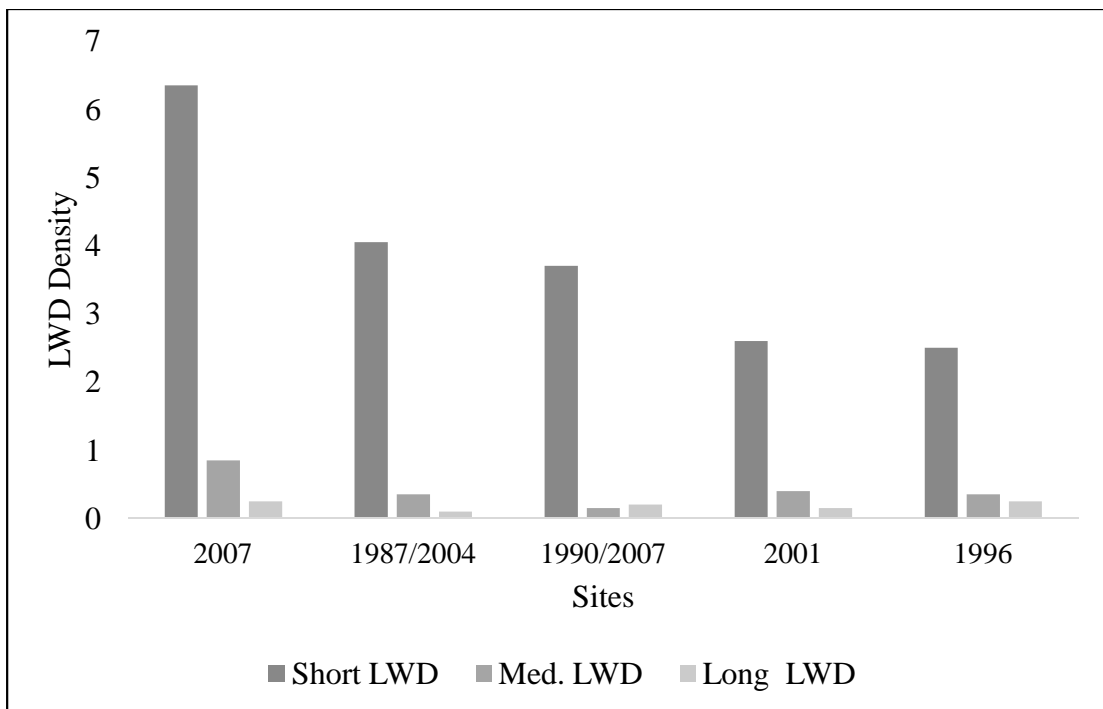


Figure 10. Large woody debris (LWD) size class distribution among study sites.



### **Coast Redwood Basal Area**

Mean coast redwood basal area was highest in the 1990/2007 site, followed by the 1996 site, the 2001 site, the 1987/2004 site and the 2007 site (Table 7). A one-way ANOVA indicated there were not significant differences for basal area of coast redwood and years since last harvest ( $p = 0.9$ ), number of re-entries ( $p = 0.44$ ) or the percentage cut per hectare ( $p = 0.76$ ).

### **Coast Redwood Dominance**

Dominance of coast redwood was highest in the 2001 site, followed by the 1990/2007 site, the 2007 site, the 1996 site and the 1987/2004 site (Table 7). Further analysis of dominance did not indicate significant results for years since last harvest ( $p = 0.18$ ), number of re-entries ( $p = 0.61$ ) or the percentage cut per hectare ( $p = 0.3$ ).

### **Coast Redwood Sprouts**

The 1996 site had the highest mean of coast redwood sprouts, followed by the 1990/2007 site, the 1987/2004 site, the 2001 site and the 2007 site (Table 7). Further analyses using ANOVA established there were not significant differences for mean sprouts and years since harvest ( $p = 0.68$ ), number of harvest re-entries ( $p = 0.31$ ) or the percentage cut per hectare ( $p = 0.35$ ). When separated by height, the 1987/2004 site had the highest number of sprouts >1 m, followed by the 1990/2007 site, the 2001 site, the 2007 site and the 1996 site. When analyzed with years since last harvest, there was a significant difference between the 1996 site and the 1987/2004 site ( $p = 0.026$ ); this relationship was negative ( $r = -0.2$ ). A one-way ANOVA indicated there was also a

significant difference among sites when sprouts >1 m were analyzed with number of harvest re-entries ( $p = 0.009$ ). A Pearson's product-moment correlation coefficient indicated that sprouts of this size increased in relation to number of re-entries ( $r = 0.18$ ). In addition, there were significant findings for sprouts >1 m and the percentage cut per hectare. Again, there was a significant difference between the 1996 site and the 1987/2004 site ( $p = 0.04$ ). It was determined that sprouts >1 m increased in relation to the percentage cut ( $r = 0.07$ ). When sprouts <1 m were analyzed, the 1996 site had the highest mean, followed by the 1990/2007 site, the 1987/2004 site, the 2001 site and the 2007 site. Post hoc analyses following ANOVA indicated there were not significant differences for sprouts <1 m and years since last harvest ( $p = 0.13$ ), number of harvest re-entries ( $p = 0.92$ ) or the percentage cut per hectare ( $p = 0.12$ ).

Table 7. Means and standard errors for coast redwood basal area, dominance and sprout density.

Coast Redwood	1987/2004		1990/2007		1996		2001		2007	
	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>
Basal area	20.80	3.65	27.81	6.22	22.15	4.77	20.85	4.05	20.42	3.34
Dominance	90.65	5.21	96.85	1.60	90.75	4.73	99.3	0.41	96.05	2.80
Sprouts<1m	54.4	13.61	74	24.36	113.55	44.51	41.2	5.41	31.15	6.80
Sprouts>1m	71.65	12.26	67.3	10.92	31.3	6.29	62.85	11.10	43.25	5.92
All sprouts	126.05	17.44	141.3	32.30	144.85	46.69	104.05	12.61	74.4	10.89

### Native Understory Species Richness

Native understory species richness varied among study sites, with a total of 52 plants identified during the course of data collection (Appendix A). The 2001 site had the highest native species richness, followed by the 1990/2007 site, the 1987/2004 site,

the 2007 site and the 1996 site. When ANOVA was used to analyze years since last harvest, there was not a significant difference for species richness among sites ( $p = 0.09$ ). There were also not significant findings for number of harvest re-entries ( $p = 0.32$ ) or the percentage cut per hectare ( $p = 0.05$ ).

### Coast Redwood Associated Species

Common coast redwood herbaceous associates including hooker's fairybells, modesty, Pacific starflower, Pacific trillium, redwood violet and sorrel were further analyzed to determine potential differences among sites. The 2007 site had the highest mean of coast redwood associated species, followed by the 1987/2004 site, the 1996 site, the 1990/2007 site and the 2001 site (Figure 11).

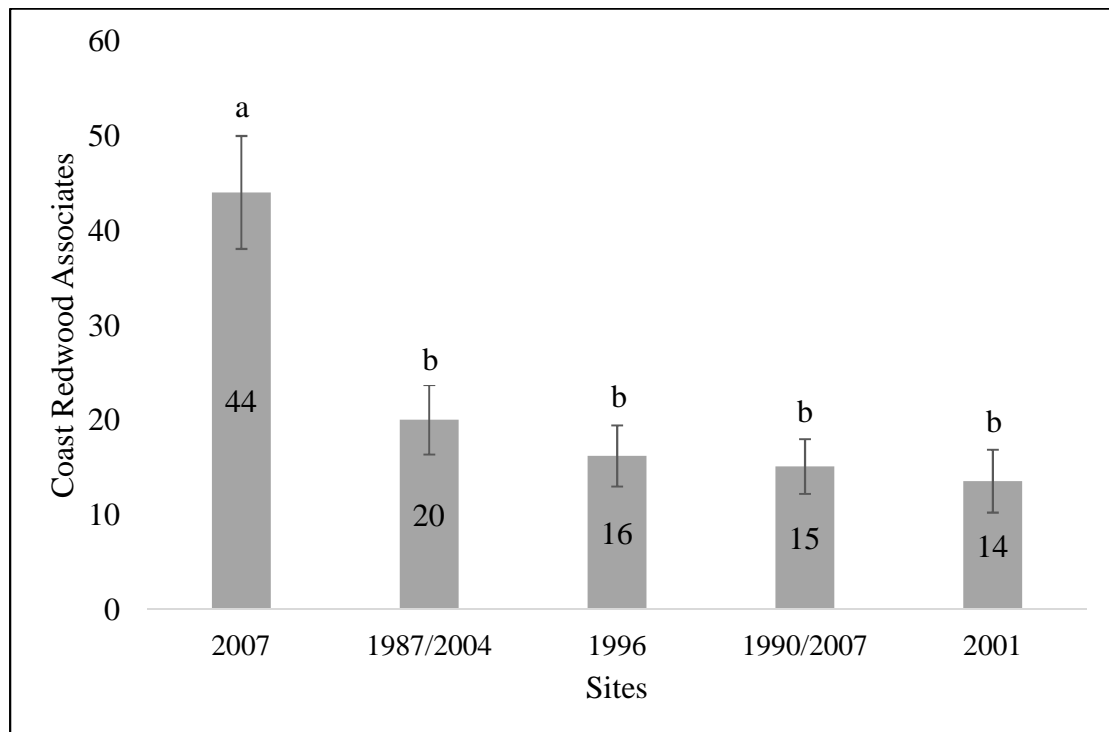


Figure 11. Mean percentage of coast redwood associated species among study sites with 95% confidence intervals (sites with the same letter were not significantly different from one another).

Hooker's fairybells was only present in three of the five study sites. The 2001 site had the highest mean percentage, followed by the 1990/2007 site and the 1987/2004 site; there were no observations in either the 1996 or 2001 sites (Table 8). The 2007 site had the highest mean percentage of modesty, followed by the 1990/2007 site, the 1987/2004 site and the 2001 site; there were no sightings in the 1996 site. The 1990/2007 site had the highest mean percentage of Pacific trillium, followed by the 1996 site, the 2007 site, the 2001 site and the 1987/2004 site. In addition, the 1990/2007 site had the highest mean percentage of Pacific starflower, followed by the 1987/2004 site, the 2001 site and the 1996 site. There were no observations of Pacific starflower in the 2007 site. Redwood violet was only observed in three of the five study sites, with the 1987/2004 site having the highest mean, followed by the 2001 site and the 2007 site. There were no observations of redwood violet in either the 1990/2007 or 1996 sites. When the percentage of redwood sorrel was analyzed, the 2007 site had the highest mean, followed by the 1987/2004 site, the 1996 site, the 1990/2007 site and the 2001 site.

These common coast redwood associates were collectively analyzed with years since last harvest, number of harvest re-entries and the percentage cut per hectare to determine potential differences among study sites. When ANOVA was used to analyze years since last harvest, results indicated that the percentage of coast redwood associates was significantly higher in the 2007 site compared with the 2001 site ( $p = 0.026$ ). A Pearson's product-moment correlation coefficient indicated this was a negative relationship ( $r = -0.25$ ). The frequency of coast redwood associates was also significantly different between the 2007 and 1996 sites ( $p = 0.016$ ), with the 2007 site

having a much higher occurrence. There was not a significant difference for the percentage of coast redwood associates when analyzed with number of harvest re-entries ( $p = 0.68$ ). However, there was a significant difference in the frequency of these species between one and two re-entries ( $p = 0.021$ ). According to Pearson's product-moment correlation coefficient, this relationship was negative ( $r = <0.01$ ). When the percentage of coast redwood associates was analyzed with the percentage cut per hectare, the 2007 site had a significantly higher amount of these species in comparison with all other sites ( $p = <0.001$ ). The percentage of coast redwood associates was found to decline in relation to the percentage cut per hectare ( $r = -0.47$ ). When ANOVA was used to analyze the frequency of these species with the percentage cut per hectare, the 1996 site was significantly different in comparison with the 1990/2007 site ( $p = 0.006$ ). According to Pearson's product-moment correlation coefficient, this relationship was positive ( $r = 0.12$ ).

Table 8. Means and standard errors for native species richness and coast redwood associated species.

	1987/2004		1990/2007		1996		2001		2007	
	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>
Coast redwood associated species	20.03	3.66	15.09	2.89	16.20	3.22	13.54	3.31	43.99	5.95
Native richness	12.45	0.78	13.80	0.59	11.25	0.55	14.00	1.05	12.00	0.69
Hooker's fairybells	0.01	0.01	0.09	0.05	0	0	0.23	0.13	0.00	0
Modesty	0.31	0.20	0.54	0.22	0	0	0.13	0.06	1.48	0.51
Pacific starflower	0.64	0.26	1.08	0.72	0.11	0.06	0.31	0.13	0	0
Pacific trillium	0.28	0.07	0.64	0.08	0.59	0.18	0.40	0.12	0.49	0.16
Redwood violet	0.26	0.24	0	0	0	0	0.16	0.14	0.10	0.10
Sorrel	18.53	3.39	12.75	2.88	15.50	3.28	12.31	3.29	41.93	6.11

## Exotic Understory Species Richness

The 1990/2007 site had the highest mean frequency of exotic species, followed by the 1987/2004 site, the 1996 site, the 2001 site and the 2007 site (Figure 12; Table 9). When examined further, exotic species richness was not significantly different among sites when analyzed with years since last harvest ( $p = 0.82$ ) or number of harvest re-entries ( $p = 0.1$ ). There was, however, a significant difference between the 2007 site and the 1990/2007 site when analyzed with the percentage cut per hectare ( $p = 0.007$ ). A Pearson's product-moment correlation coefficient indicated this relationship was positive ( $r = 0.27$ ).

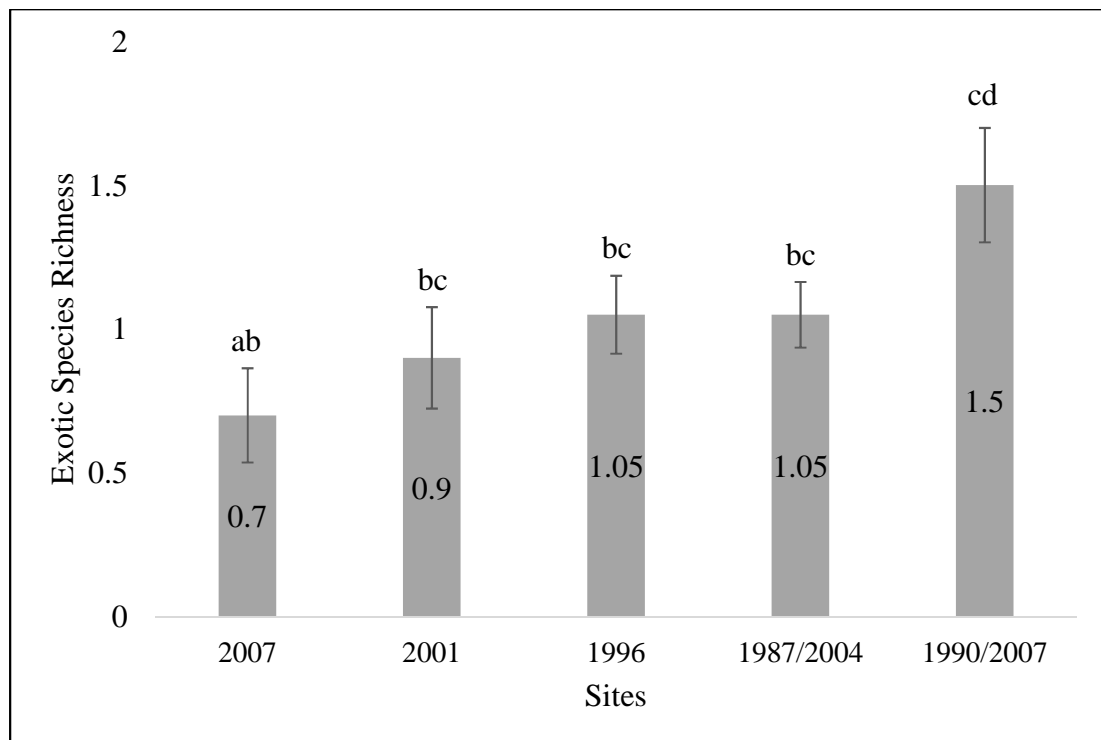


Figure 12. Mean exotic species richness among study sites with 95% confidence intervals (sites with the same letter were not significantly different from one another).

In addition to analyzing exotic species richness, common exotic species including broad-leaved helleborine (*Epipactis helleborine*), English holly (*Ilex aquifolium*) and forget-me-not (*Myosotis latifolia*) were further evaluated to determine potential differences among sites.

The 1987/2004 site had the highest mean percentage of helleborine, followed by the 1996 site, the 2001 site and the 2007 site (Table 9). A one-way ANOVA indicated there was not a significant difference among sites for years since last harvest and the percentage ( $p = 0.18$ ) or frequency ( $p = 0.09$ ) of helleborine. There was also not a significant difference found for the percentage of helleborine when analyzed with number of harvest re-entries ( $p = 0.16$ ). However, there was a significant difference among sites for the frequency of this species and number of harvest re-entries ( $p = 0.007$ ); according to Pearson's product-moment correlation coefficient, this relationship was positive ( $r = 0.27$ ). When ANOVA was used to analyze the percentage cut per hectare, there were not significant results for either the percentage ( $p = 0.22$ ) or frequency ( $p = >0.05$ ) of helleborine.

English holly was only observed in two of the five sites; the 1987/2004 site and the 1990/2007 site (Table 9). When number of harvest re-entries was analyzed using ANOVA, the percentage ( $p = 0.025$ ) and frequency ( $p = 0.012$ ) of the shrub were significantly higher in sites re-entered once versus sites re-entered twice. There were not significant findings for years since last harvest when analyzed with the percentage ( $p = 0.54$ ) or frequency ( $p = 0.4$ ) of English holly. Similar results were determined for

the percentage cut per hectare; the percentage ( $p = 0.2$ ) and frequency ( $p = 0.064$ ) of holly were not significant.

The 1987/2004 site had the highest mean percentage of forget-me-not, followed by the 1996 site, the 1990/2007 site, the 2001 site and the 2007 site (Table 9). When ANOVA was used to analyze years since last harvest, the frequency of forget-me-not was significantly higher in the 1996 site compared with the 2001 site ( $p = 0.011$ ) and the 2007 site ( $p = 0.03$ ), although the percentage of the herb was not significantly different among sites ( $p = 0.3$ ). The relationship between frequency and years since harvest was positive ( $r = 0.24$ ) according to Pearson's product-moment correlation coefficient. When analyzed with number of harvest re-entries, there were not significant differences for either the percentage ( $p = 0.1$ ) or frequency ( $p = 0.49$ ) of forget-me-not. When ANOVA was used to analyze the percentage cut, a significant difference in frequency was determined for the 1996 site in comparison with the 2007 site ( $p = 0.0005$ ) and the 2001 site ( $p = 0.016$ ); this was a positive relationship ( $r = 0.14$ ). There was not a significant difference among any of the sites for the percentage of forget-me-not and the percentage cut per hectare ( $p = 0.23$ ).

Table 9. Means and standard errors for exotic species.

	1987/2004		1990/2007		1996		2001		2007	
	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>	<i>Mean</i>	<i>S.E.</i>
Exotic richness	1.05	0.11	1.50	0.20	1.05	0.14	0.90	0.18	0.70	0.16
Helleborine	1.00	0.64	0.74	0.30	0.81	0.29	0.19	0.12	0.05	0.04
English holly	0.10	0.10	0.21	0.14	0	0	0	0	0	0
Forget-me-not	1.00	0.64	0.74	0.30	0.81	0.29	0.19	0.12	0.05	0.04



## DISCUSSION

The results of this study indicated that the percentage cut per hectare was the strongest independent variable for a substantial number of the structural and compositional features analyzed, including canopy cover, stand density, LWD density, coast redwood associated species and exotic species richness. A number of these variables including canopy cover, LWD density and coast redwood associated species declined in response to higher harvest yields.

A Pearson's product-moment correlation coefficient determined that canopy cover had two equally strong independent variables, years since last harvest and the percentage cut per hectare. Although the results obtained from ANOVA indicated that these findings were significant, both were weak correlations and other factors may have influenced these relationships. Previous research has indicated that canopy cover increases over time in the absence of outside disturbance (Russell et al. 2014), although it is possible that the time intervals since the last harvest entry were too short for long-term relationships to become evident. These stands are likely still in the early stages of recovery and may still be undergoing stand initiation (Oliver 1981). Results also indicated that canopy cover decreased in relation to the percentage cut per hectare. Incidentally, the 1996 site, which had significantly lower canopy cover in comparison with the other sites, had the lowest frequency of native and coast redwood associated species and the highest frequency of the exotic species forget-me-not.

As predicted in the hypotheses, mean stand density was most strongly correlated to the percentage cut per hectare, which was also the strongest independent variable for

density of coast redwood. This was not an unexpected finding as coast redwoods sprout prolifically following a harvest event (Cole 1983). These results were further supported by analyses of size classes for both coast redwood and tanoak. Stems in smaller size classes were found to increase significantly while larger diameter coast redwoods were found to decline in response to heavier harvests. The 2007 site, which had the lowest percentage cut of all sites, had the lowest mean number of stems <25 cm and the most even distribution of size classes. The sites entered twice, including the 1987/2004 site and the 1990/2007 site, had the highest stand density among all sites examined. These sites also had the highest number of stems <25 cm. Previous findings have indicated that tree density among coast redwoods may take decades to develop structural and compositional features more reminiscent of a mature forest (Veirs and Lennox 1982; Thornburgh et al. 2000; Hageseth 2008). This further supports the idea that a lower cut promotes features more reminiscent of a mature forest.

Results for density of LWD indicated a decline in relation to the percentage cut, which was also predicted in the hypotheses. This could be due in part to a regulation specific to the Southern Subdistrict which requires that remaining slash not exceed 30” (California Department of Forestry and Fire Protection 2013). It is possible that some pre-existing LWD was also removed during the clean-up process in an effort to adhere by these guidelines. The 2007 site, which had wildlife habitat goals specifically pertaining to the retainage of LWD, was determined to have the highest mean among all sites. The 2007 site also had the highest amount of short and medium LWD, further supporting the

fact that the management goals set in place by the Land Trust of Santa Cruz County were being met.

Although findings for basal area of coast redwood were not deemed significant, previous studies have indicated that over time in the absence of disturbance, basal area increases in coast redwood forests (Gellman and Zielinski 1996; Sillett and Van Pelt 2007; Russell and Michels 2010; Sinclair 2013). It is possible that not enough time has passed for a relationship between basal area and years since harvest to become apparent, especially since the longest period following harvest was <20 years, a considerably small amount of time in the lifespan of this species. Basal area is generally viewed as an indicator of forest maturation and development (Burns and Honkala 1990) and since Byrne-Milliron is an actively managed forest, it would make sense that repeated re-entries might stymie its progress toward the development of larger diameter trees.

Coast redwood dominance was also deemed to be insignificant when analyzed with the independent variables. The fact that basal area and coast redwood dominance are often synonymous with one another may explain why results were not significant for this variable. Other conditions including light availability, soil quality and stand age may have played a factor as well.

The percentage cut per hectare was also the strongest independent variable for coast redwood associated species and exotic species richness. Coast redwood associated species were found to decline significantly in relation to this variable; this was the strongest correlation of all dependent variables analyzed. The 2007 site had a significantly higher percentage of these associated species in comparison with the other

study sites. The 2001 site, which had one of the highest harvest yields, incidentally had the lowest percentage of coast redwood associated species.

Exotic species richness increased with both number of harvest re-entries and the percentage cut per hectare. Sites re-entered twice, including the 1987/2004 site and the 1990/2007 site, exhibited the highest occurrence of exotic species. The 1987/2004 site also had the highest frequency of both broad-leaved helleborine and forget-me-not. The site with the highest cut (1990/2007) also had the highest exotic species richness. In addition, these were the only sites with English holly present. Frequency of both English holly and broad-leaved helleborine were found to increase with the number of harvest re-entries. These results were not surprising since research has shown that exotic species are more likely to increase following a disturbance (Ebrecht and Schmidt 2003). The 2007 site, which experienced the lowest level of disturbance, had the lowest exotic richness of all sites, as well as the lowest frequency of both broad-leaved helleborine and forget-me-not.

In many respects, the 2007 site serves as a model for sustainable selective-harvest. The management goals set in place by the Land Trust of Santa Cruz County have ensured that this site maintained its structural and compositional integrity following harvest. In addition to a lighter harvest, specific regard for retainage of snags, large diameter trees, LWD and development of a complex canopy structure have allowed this area of the Byrne-Milliron Forest to maintain a healthy ecosystem amidst active management.

The results obtained from ANOVA and Pearson's product-moment correlation coefficient indicated that several of the dependent variables including mean stand density,

coast redwood density, tanoak density and coast redwood sprouts >1m increased significantly in relation to number of harvest re-entries and the percentage cut per hectare. This supports previous findings that standard thinning practices increase stand density and result in an increase of stems in smaller size classes (Russell and Jones 2001; O'Hara et al. 2010). Although studies have promoted heavy thinning to speed up the attainment of desirable structural features (Oliver et al. 1994; Berrill 2009), this method does not prove to be applicable to coast redwood forests due to their prolific sprouting and ability to self-thin without outside management (Sachs et al. 1993; Lutz and Halpern 2006; Floyd et al. 2009; Russell et al. 2014).

## RECOMMENDATIONS

A number of findings within this study support the concept that coast redwood communities take a considerable amount of time to recover following heavier harvest regimes. It is evident that in order to promote the ecosystem integrity of working forests, a lighter cut is more beneficial and may result in a quicker recovery time following harvest.

The Land Trust of Santa Cruz County has done an exemplarily job in maintaining a healthy forest structure and composition at Byrne-Milliron Forest, especially within the 2007 harvest site. It is recommended that they use this site as a model for future management applications. It is apparent that the initial goals set forth have proved successful, resulting in an abundance of coast redwood associated species, LWD and other features reminiscent of a mature stand.

Variables not covered in this study but recommended for further evaluation include method of harvest as well as spatial arrangement of coast redwood in response to active management. The 2007 site used predominantly skyline cable, whereas other sites mainly implemented the use of tractors and skidders. It is possible that these methods caused a higher level of disturbance, resulting in a slower recovery period following harvest. In addition, researching the spatial integrity of coast redwood stems in response to selective-harvest may allow foresters and land managers to better understand how this extraction method affects structure and stem distribution.

Further research is also needed in the southern range of coast redwood communities. Stands in the northern range have a myriad of different conditions, ranging

from an increase in precipitation to a variety of understory species not present in Santa Cruz County. The majority of previous studies have been confined to the northern range, which can be limiting when trying to establish current relationships and mechanisms of recovery in the southern range. Additional studies in this region will help to eliminate this gap in the research.

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## APPENDIX A: NATIVE SPECIES LIST

<i>Actaea rubra</i>	<i>Polygala californica</i>
<i>Adenocaulon bicolor</i>	<i>Polystichum californicum</i>
<i>Aquilegia Formosa</i>	<i>Rosa gymnocarpa</i>
<i>Aralia californica</i>	<i>Rutablebus leucodermis</i>
<i>Artemisia douglasiana</i>	<i>Rubus parviflorus</i>
<i>Asyneuma prenanthoides</i>	<i>Rubus ursinus</i>
<i>Baccharis pilularis</i>	<i>Sanicula crassicaulis</i>
<i>Calystegia purpurata</i>	<i>Scoliopus bigelovii</i>
<i>Cardamine californica</i>	<i>Smilacina racemosa</i>
<i>Ceanothus spp.</i>	<i>Smilacina stellate</i>
<i>Clinopodium douglasii</i>	<i>Solanum douglasii</i>
<i>Clintonia andrewsiana</i>	<i>Solanum umbelliferum</i>
<i>Corallorhiza maculate</i>	<i>Stachys bullata</i>
<i>Corylus cornuta ssp. californica</i>	<i>Symphoricarpos mollis</i>
<i>Drymocallis glandulosa</i>	<i>Tiarella trifoliata</i>
<i>Dryopteris arguta</i>	<i>Toxicodendron diversilobum</i>
<i>Equisetum telmateia</i>	<i>Trientalis latifolia</i>
<i>Eriodictyon californicum</i>	<i>Trillium ovatum</i>
<i>Fragaria vesca</i>	<i>Urtica dioica</i>
<i>Frangula californica</i>	<i>Vancouveria planipetala</i>
<i>Galium californicum</i>	<i>Vicia gigantean</i>
<i>Heuchera micrantha</i>	<i>Viola ocellata</i>
<i>Lonicera hispidula</i>	<i>Viola sempervirens</i>
<i>Marah fabaceus</i>	<i>Whipplea modesta</i>
<i>Mimulus aurantiacus</i>	<i>Woodwardia fimbriata</i>
<i>Nemophila parviflora</i>	
<i>Osmorhiza brachypoda</i>	

## APPENDIX B: EXOTIC SPECIES LIST

*Ageratina adenophora*

*Cortaderia jubata*

*Cotoneaster pannosus*

*Epipactis helleborine*

*Genista monspessulana*

*Hedera helix*

*Ilex aquifolium*

*Myosotis latifolia*

*Torilis arvensis*