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September 13–16, 2016

Sequoia Conference Center, Eureka, CA

Forest Service

Pacific Southwest Research Station

General Technical Report PSW-GTR-258

October 2017

Disclaimer

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Coast Redwood Science Symposium—2016: Past Successes and Future Directions

September 13–16, 2016

Sequoia Conference Center, Eureka, CA

Richard B. Standiford, Yana Valachovic

Technical Coordinators

U.S. Department of Agriculture, Forest Service
Pacific Southwest Research Station
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General Technical Report
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Abstract

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There is no more iconic tree or more closely watched forest ecosystem than coast redwood. With its limited range and high value, the coast redwood forest is a microcosm of many of the emerging science and management issues facing today's forested landscapes. As new information is collected and new management approaches and treatments tried, it is critical that policies and strategies guiding use and management within the redwood region be reviewed and updated based on objective scientific information. With changes in California's demographic makeup, land ownership, and the regional economy, great interest has developed in areas such as forest sustainability and restoration, watershed assessment, fish and wildlife habitat conditions, and new silvicultural strategies. This symposium is part of a continuing effort to promote the development and communication of scientific findings to inform management and policy decisions. The symposium includes plenary speakers, concurrent sessions, a poster session, reception, and field trip opportunities to view and explore the North Coast.

Keywords: Coast redwood, *Sequoia sempervirens*, California forests, redwood conservation, forest wildlife management, forest policy.

Key Sponsors:

- Save the Redwoods League (link)
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Poster Session Sponsor:

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Previous Redwood Symposia:

- June 18-20, 1996; Arcata, CA; Coast Redwood Forest Ecology and Management – University of California Online Proceedings

- March 15-17, 2004; Rohnert Park, CA; Proceedings of the Redwood Region Forest Science Symposium: What Does the Future Hold? - USDA Forest Service PSW-GTR 194

- June 21-23, 2011; Santa Cruz, CA; Proceedings of coast redwood forests in a changing California: A symposium for scientists and managers - USDA Forest Service PSW-GTR 238

Contents

Dedication of the Proceedings – Dr. Lowell Diller

Photo Credit: Janice Diller

The conference proceedings organizers wish to honor Dr. Lowell Diller, a tireless advocate for the role of science and education in redwood conservation, who passed away on March 4, 2017. Lowell was a contributor to all four redwood symposia, and served as an organizer, reviewer, moderator, and field trip organizer for these symposia.

Lowell was born in Welland, Ontario, Canada on March 14, 1947. He graduated from Oregon State University, and then served his country in Vietnam. He received his PhD in Zoology from the University of Idaho in 1981. After serving in the Biology Department at Frostburg State University in Maryland, Lowell was hired by Simpson Timber Company (now Green Diamond) in Korbel, California as senior wildlife biologist in 1990. He and his crews dedicated their careers to studying terrestrial and aquatic species of coastal redwood forests, however, his best-known work was in northern spotted owl conservation. These efforts provided an understand of the relationship between the owl, its habitat and prey base and produced many high quality peer-reviewed publications on wildlife, aquatic ecology, and biodiversity in managed coastal forests, and to Lowell's long-term involvement in many interagency recovery teams and task forces. While he worked in contentious issues, Lowell always maintained scientific objectivity and his assessments and recommendations were always data driven. Lowell was also an Adjunct Professor in the Humboldt State University Wildlife Department, where he taught a class on management of reptiles and amphibians. He was a masterful lecturer, a wonderful mentor, and he helped inspire people of all ages to learn more about the natural world.

Lowell received numerous awards and recognition for his contributions to the wildlife field. A partial list includes: an Award of Appreciation from The Western Section of the Wildlife Society for Outstanding Support as President in 2003 and an Award of Appreciation as the 2010 Recipient of the Barrett A. Garrison Outstanding Mentor Award; the Ralph W. Schreiber Conservation Award from the American Ornithologist Union in 2010; the Wings Across the Americas' Research and Partnership Award from the United States Forest Service in 2009; the Professional of the Year from the Wildlife Society California North Coast Chapter in 2001 and 2012; a Lifetime Achievement and Mentorship Award from The Society for Northwest Vertebrate Biology in 2017; a California State Legislature Assembly Resolution recognizing his service in 2014; and a United States Congressional Record Celebrating his work in 2014.

He is survived by his wife, Janice Standard, and his children Jennifer "Fern" Diane and Alexandrea "ZB" Beth, as well his siblings Wendell Diller, Glenna Pulver, Rose Ada Combs, Karolyn Mengershausen, and David Diller, and his in-laws Charlotte Standard, James Standard, Elie Standard, Raymond Standard, and Lana Standard.

We will miss his leadership, wit, and dedication to redwood science and education, and we are pleased that his legacy will live on.

General Session

Changes in the Redwood Region from 1996-2016[1]

Yana S. Valachovic[2] and Richard B. Standiford[3]

Abstract

This introductory paper highlights some of the changes in redwood region land ownership, markets and infrastructure to help contextualize the dynamic nature of the forest industry in California and to help set the stage for a management and policy dialog among symposia participants. Twenty years have passed from the first redwood symposium in 1996 and with this so too have the conservation and management issues changed among regional stakeholders. Through 70 talks delivered in plenary and concurrent sessions, a poster session, and four field trip choices, session participants had the opportunity to learn more about how forests are managed today, gain an enhanced understanding of scientific advancements, and see first-hand some of the changes in both private and public forest land management. This paper reflects the conference organizer's personal observations and available regional data.

Keywords: coast redwood, economics, infrastructure, *Sequoia sempervirens*.

Background

The first redwood symposium was held in Arcata in 1996 (LeBlanc 1996) and has since travelled throughout the redwood region with a 2004 symposium in Rohnert Park (Standiford et al. 2007) and a 2011 symposium in Santa Cruz (Standiford et al. 2012). Bringing the redwood symposium back to Humboldt County provided an opportunity for reflection upon the twenty years of change that have taken place in the working forest land of California's redwood region. In 2016, with changes in California's demographic makeup, land ownership, and the regional economy, great interest has developed in areas such as forest sustainability and restoration, watershed assessment, fish and wildlife habitat conditions, and new silvicultural strategies. These themes were discussed in this symposium and are documented in these proceedings.

Ownership transitions

Today coast redwood forests (*Sequoia sempervirens*) are limited to a narrow band along California's coastal mountain range with a small population in southwestern Oregon (Figure 1a). A large percentage of these forests are privately held in both larger industrial holdings and in smaller non-industrial holdings (Figure 1b). During the last twenty years, there have been many changes in ownership across the region encompassing both redwood and Douglas-fir (*Pseudotsuga menziesii*) forests. Likely the most significant changes in ownership were observed in the industrial land base where publicly traded companies (e.g. Louisiana Pacific, Georgia Pacific, and Pacific Lumber Company) sold to privately owned family enterprises (e.g. Green Diamond Resources Company, Humboldt and Mendocino Redwood Companies). There were slight reductions in the industrial land base. Of note, was the approximately 33,000 acres transferred to new public ownerships with the creation of the Headwaters Reserve managed by the Bureau of Land Management (7,500 acre), the 24,700-acre addition to the California Department of Park's holdings in Del Norte County, and the recently established McKay Community Forest (1,000 acres) owned and managed by the County of

[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.

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Humboldt. The Yurok Tribe also acquired 22,000 acres. During the last twenty years, non-profit organizations emerged as new working forest owners, primarily in Mendocino and Sonoma Counties (i.e. Redwood Forest Foundation and Conservation Fund). Timber investment management organizations (TIMOs) also bought and sold forests during this period with Hawthorne Timber Company buying from Georgia Pacific and eventually selling to a new TIMO market participant in California, Lyme Redwood Timberland. Some of the smaller family lands transferred to other forest landowners, but many were also subdivided for rural development and/or cannabis cultivation, especially in Humboldt County's Douglas-fir zone.

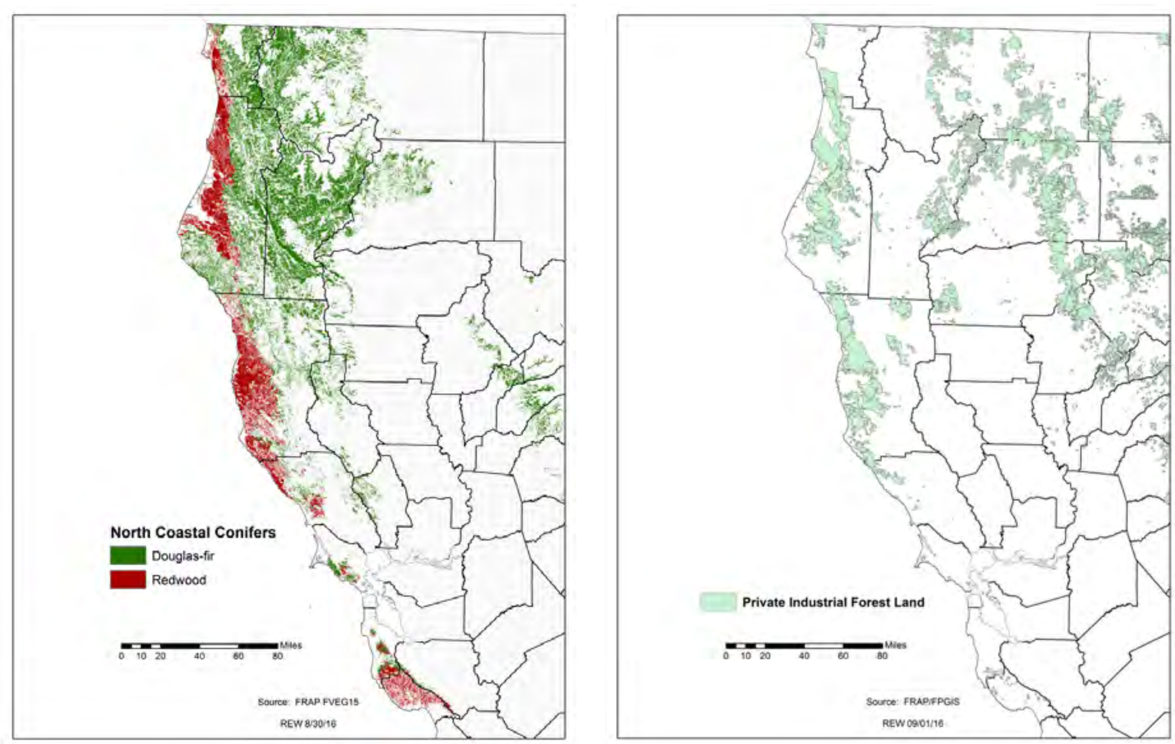

Figure 1—A. Distribution of Coast redwood and Douglas-fir across coastal counties in California. B. Private industrial ownership in northern California. Map sources: CAL FIRE, Fire and Resource Assessment Program.

Infrastructure loss

During the last twenty years, there were significant changes to the manufacturing sector. Sawmill reduction was common throughout California (Figure 2) and many sawmills that process redwood and Douglas-fir closed through the redwood region (e.g. Orick, Klamath, Korbel, Fortuna, Branscomb, Willits, etc). Additionally, there have been closures of the biomass power facilities that help dispose of mill residuals and a closure of the only pulp mill (near Eureka) in the redwood region. The remaining sawmill and biomass infrastructure is limited (Figure 3) and the log hauling distances have increased. The loss of manufacturing and biomass presents a considerable challenge to the region as it affects the viability of a skilled workforce, reduces competition for logs and can suppress prices, increases trucking costs, and creates a challenge for disposal of sawmill residuals and reduction of marketable byproducts. At present, sawmill facilities appear to generally be tied to the industrial land base and long-term log purchasing arrangements may have the unintended effect of reducing timber demand from the non-industrial family forest owners.

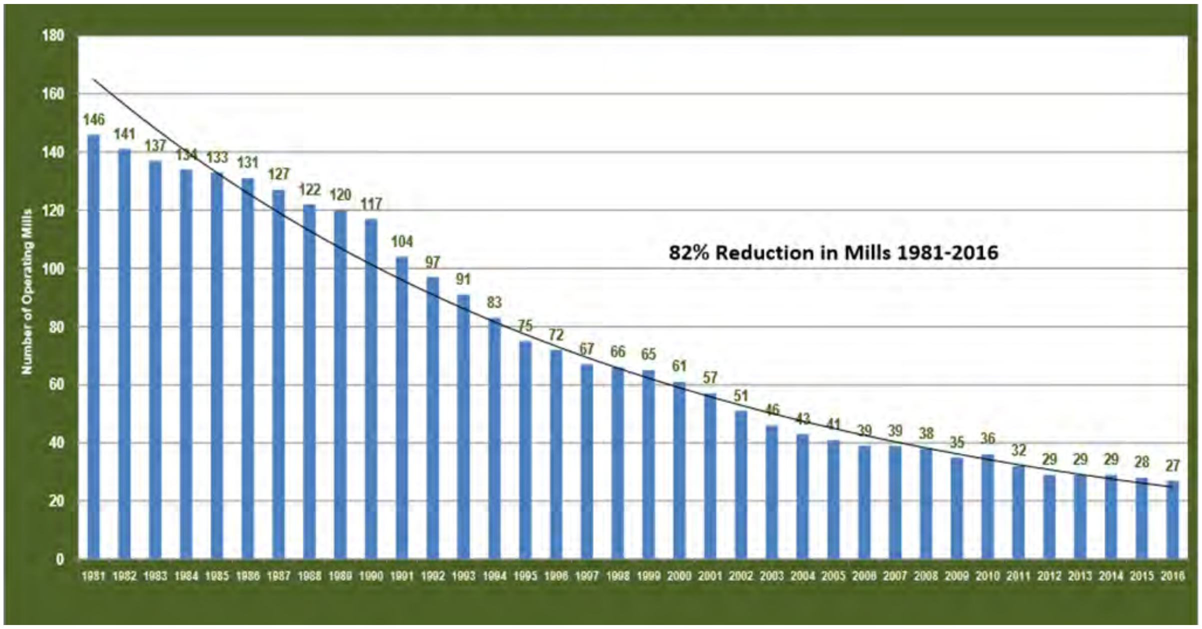

Figure 2—Reduction in the number of sawmills in California from 1981-2016 based on mill census data (BBER 2016).

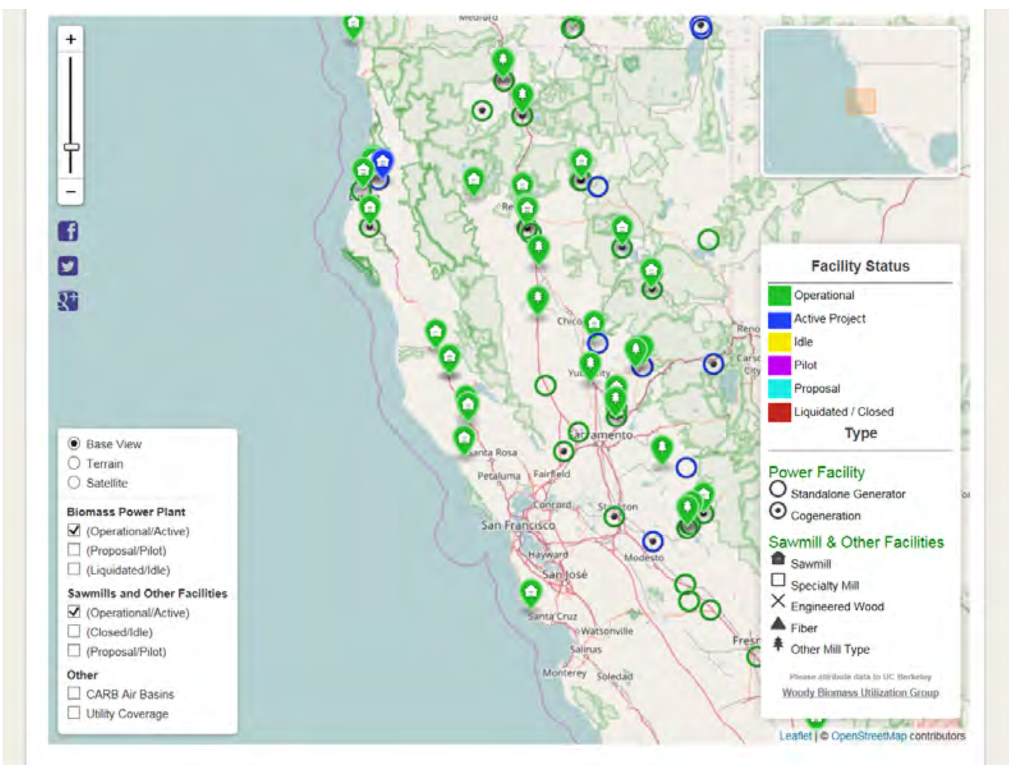

Figure 3—Operational sawmill and biomass power plants in northern California. (UC ANR 2017)

Timber harvest and market changes

While timber harvest has been decreasing statewide, the north coast continues to produce much of the total timber harvest (Figure 4), with Humboldt County remaining as the biggest timber-producing county in California (Figure 5). The effect of the transition away from old-growth and the 1990 listing of the northern spotted owl can be seen in decreased harvest levels from the coastal forest totals. The 2009 national recession was also significant, with Figures 4 and 5 demonstrating that markets have not rebounded and north coast harvest levels are lower than pre-recession levels. The new family and non-profit ownerships are bringing different management styles and goals to the region. They generally have more flexibly to operate on longer time horizons, especially while they rebuild timber inventories, improve road infrastructure and design, and transition to cable logging systems from predominately ground-based yarding systems.

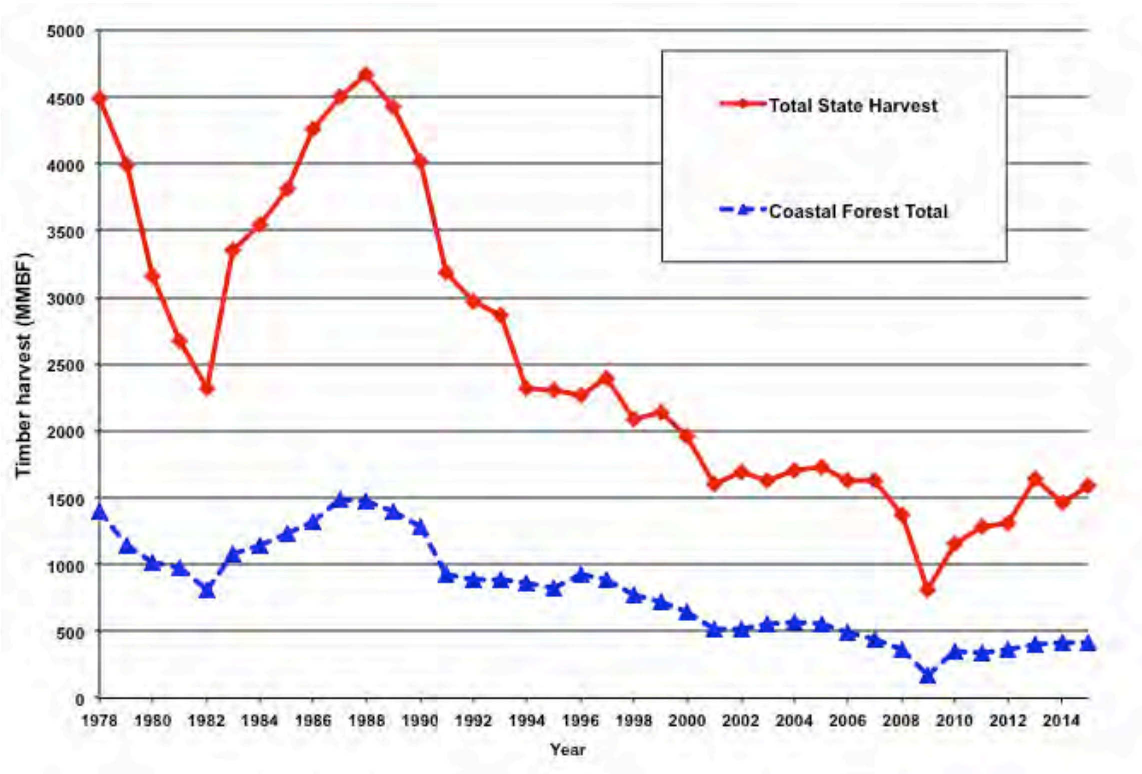

Figure 4—North coast harvest compared to the state total harvest (SBOE various years).

Most evident during this twenty-year period has been completion of the transition to a young-growth forest economy (Standiford 2012). Redwood prices have remained significantly higher than Douglas-fir. For example, 2016 young growth redwood stumpage prices in Humboldt and Del Norte Counties was $600 per thousand board feet for medium sized logs (150-300 board feet per log), while young growth Douglas-fir stumpage for this same region and size class averaged $270 per thousand board feet (SBOE 2016). Douglas-fir prices are tied in with trends in the housing markets, while redwood is more closely tied in with remodel and specialty uses. There has been some growth in new markets over the twenty-year period from forest certification and carbon offsets. Although certified logs have not resulted in a significant price premiums, this system has been important in securing long-term purchase agreements with big box store retailers, has required greater investment in management planning and enhanced community engagement. Forest carbon offset markets are emerging with California's cap and trade policies brought about by passage of the California Global Warming Solutions Act of 2006 (California Assembly Bill 32). Carbon sales have provided an alternative method to generate revenue from the Douglas-fir zone, especially where harvest plan preparation costs and long distances to milling infrastructure have made timber harvest economically challenging. While there are considerable forest inventory investment and maintenance costs, carbon has the potential to generate revenue on a property-wide revenue rather than a unit by unit basis.

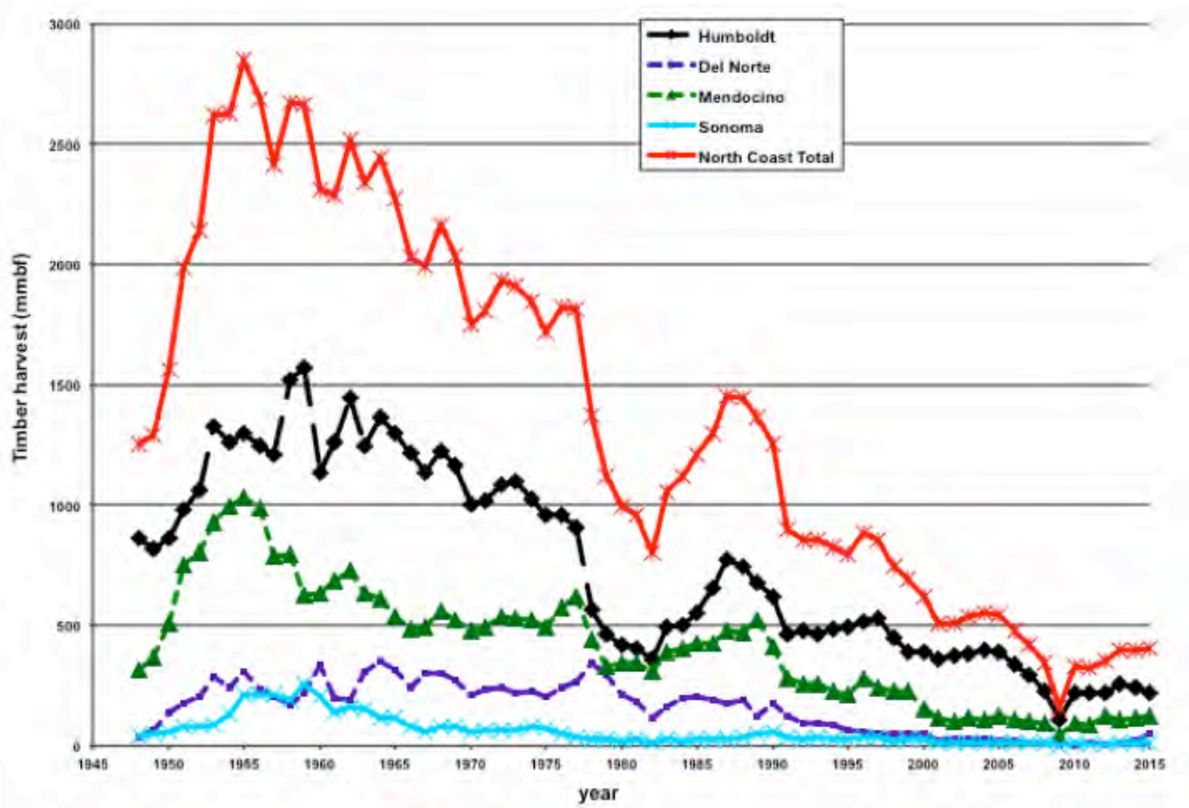

Figure 5—North coast harvest from 1945-2016 by county (SBOE various years).

Science, policy and management partnerships

Within these recent market, ownership, and infrastructure changes, there have also been significant developments under the Endangered Species Act, the Clean Water Act, and the California Forest Practice Rules that govern private forestland management. To satisfy these regulations and to respond to sensitive species listings, larger forest landowners have negotiated property wide habitat conservation agreements both for single species (i.e. northern spotted owls) and multiple species in attempt to create regulatory stability and maintain management options. To develop these agreements and to monitor their effects, many companies now employ interdisciplinary scientists and resource professionals who work in the fields of watershed science, wildlife biology, and botany. Additionally, women are more commonly working as scientists, foresters, and managers, further contributing to a diversified workforce.

The net effect of this changing workforce, and the focus on science, has resulted in significant advancements in our understanding of the ecosystems within the redwood region. As is evident by this, and the past three redwood symposia, the last twenty years have seen tremendous advances in understanding the interrelationships between forest management and conservation and the region's wildlife, plant, fish, and watershed resources.

Conclusions

While no one can predict the future, the next twenty years are likely to be as dynamic as the past twenty years. Forest landowner demographics will continue to propel this change as most non-industrial forests are owned by older generations (Ferranto et.al. 2011) and family succession is guaranteed. New market forces, changing policies, competing land uses, growing human populations, and changes in climate are all going to challenge forest landowners to continue to be well-informed and to balance sustainability and profitability through economic and environmentally based decision making. Additional changes in forest products infrastructure and markets are anticipated.

During the last twenty years, the "timber wars" have largely come to rest, but the robust dialog over forest management is likely to continue in the redwood region. Hopefully this dialog can be better informed by symposia such as this, and by the scientists and managers, who have dedicated their time to better understanding this region, and the forest and people that live here.

Acknowledgments

The ideas presented here are a product of rich discussions over the years with many colleagues and forest landowners and managers. We thank the CAL FIRE's Fire and Resource Assessment Program team for their production of several maps used in this presentation.

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Understanding *Sequoia sempervirens*[1]

Emily E. Burns[2]

"Scattered as the leaves of the forest are the printed references to the Redwoods of California."
– Joseph Grant, Save the Redwoods League, 1935

Humans have no doubt observed the coast redwood, *Sequoia sempervirens* (D. Don) Endl., in wonder for thousands of years. It is no small task to seek understanding of a species whose sheer size cannot readily be assessed from the ground and whose generation time can exceed that of humans by 100 times. Fortunately, the scientific method has provided a steadfast approach to study, describe, and explore many aspects of *S. sempervirens* ecology in recent centuries.

This review focuses on select aspects of coast redwood ecology that illustrate how redwood research has progressed and proliferated over time. Redwood research developed slowly after the first botanical description of the species in 1824, but in recent decades the pace of ecological investigation has accelerated. Major methodological innovations including molecular genetics, canopy-access rope techniques, and dendrochronology have resulted in profound discoveries that shape our contemporary understanding of Earth's tallest tree and the forest it defines.

First roots of redwood research trace back to European discovery of coast redwoods in California. On October 10, 1769, Franciscan missionary Fray Juan Crespi documented the first historical observation of redwood in his diaries of the Portolá Expedition near Monterey Bay (Bolton 1927). He writes of the party traveling in an area likely along Soquel Creek, "over plains and low hills, well forested with very high trees of a red color, not known to us. They have a very different leaf from cedars, and although the wood resembles cedar somewhat in color, it is very different, and has not the same odor; moreover, the wood of the trees that we have found is very brittle. In this region there is a great abundance of these trees and because none of the expedition recognizes them, they are named redwood from their color."

Another 22 years passed before the first known botanical collections of redwood occurred which led to formal botanical description of the species in the 19th century. In 1791, the Czech botanist Thaddeus Haenke planted the first redwood seeds in Europe near Granada, Spain that were collected during the Malaspina Expedition (Jepson 1910). In 1795, Archibald Menzies brought additional specimens from the Santa Cruz region back to England from the Vancouver Expedition and these specimens were used by the English botanist Aylmer Bourke Lambert in 1824 to name the coast redwood *Taxodium sempervirens*, in recognition of its morphological similarities to bald cypress (Jepson 1910). Stephen Endlicher subsequently changed the genus in 1847 to *Sequoia*, a name with unknown and debated origin (Lowe 2012).

The evolutionary relationship of *S. sempervirens* to other conifers remained dependent on morphological trait assessments and therefore highly unresolved until the late 20th century. For many decades, botanists debated the phylogenetic arrangement of *S. sempervirens* within the conifer lineage until R. Pilger assigned *S. sempervirens* to the Taxodiaceae in 1926 on the basis of ovulate cone similarities (Brunsfeld et al. 1994, Eckenwalder 1976). It remained there for nearly 50 years until a strong case to merge the Taxodiaceae and Cupressaceae families was made on the basis of vegetative and reproductive traits from all life cycle stages (Eckenwalder 1976). Brunsfeld et al. (1994) confirmed through a cladistic analysis of genetic markers that including *S. sempervirens* and the other

[1] A version of this paper was presented at the Coast Redwood Science Symposium September 13-15, 2016, Eureka, California.

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Taxodiaceae species (except *Sciadopitys*) in the Cupressaceae indeed created a monophyletic conifer lineage.

The application of genetic markers not only resolved the long-standing mystery of *S. sempervirens* evolutionary relationships with other species, but also enabled a new field of research to begin on inheritance and intraspecific diversity. Through neutral genetic marker studies (including allozymes, restriction length polymorphisms, and microsatellites) it has been shown that both chloroplast and mitochondrial DNA in *S. sempervirens* are paternally inherited (Neale et al. 1989) and there is evidence for an autoallopolyploid origin for this hexaploid species (Douhovnikoff et al. 2004, Rogers 1997). Genetic marker studies have further revealed a high frequency of clonality within stands, ramet distances of up to 40 m (Douhovnikoff et al. 2004), and common genetic diversity within fairy rings (Rogers 2000). In addition, genetic investigations have revealed a disjunction between northern and southern *S. sempervirens* populations (Brinegar 2011, Douhovnikoff and Dodd 2011). These findings demonstrate a need to integrate contemporary forest management and conservation practices with genetic screening so that forestry techniques can be used in the future with enhanced understanding of how stand manipulation directly impacts genetic diversity within stands and across the ecosystem range.

As the field of molecular genetics was developing in the late 20th century, a new field of research was beginning in the coast redwood forest. Ascension into *S. sempervirens* crowns with rope access techniques catalyzed the study of epiphytes, plants growing on tree crowns. Stephen Sillett's pioneering first investigations of species inhabiting the canopy transformed the world's perspective of biodiversity in the coast redwood forest, revealing a lush ecosystem more than 50 m above the ground. Structurally complex *S. sempervirens* crowns with reiterated trunks, massive limbs, and dead wood were found to support as much as 742 kg of epiphytic biomass in a single crown (Sillett 1999, Sillett and Van Pelt 2007). Numerous vascular plants, bryophytes, lichens, and microorganisms grow in the canopy (Carrell and Frank 2015, Sillett and Van Pelt 2000, Williams and Sillett 2007), but the most common species found in temperate rainforest sites was the leather leaf fern, *Polypodium scouleri*, a perennial epiphyte that grows extensive mats of succulent rhizomes and fibrous roots over branches (Sillett and Bailey 2003). Fern mats were shown to trap organic matter, rain, and fog, facilitating the development of arboreal soils that can form a layer up to 1 m deep and are characterized by low pH and low soil moisture content during the dry season (Enlow et al. 2006, Enlow et al. 2010, Sillett and Bailey 2003). These organic soils support a diverse arthropod community (Jones 2005) and cryptic vertebrates including the wandering salamander, *Aneides vagrans* (Spickler et al. 2006).

Within a few years of rope methods providing redwood canopy access, physiological research on *S. sempervirens* also expanded into the treetops. A common theme among these studies was the investigation of how climate influences redwood growth. Dendrochronology was applied to study whole-tree wood production by sampling cores from *S. sempervirens* crowns. Contrary to the long-standing assertion that growth rates decline as redwoods age (Fritz 1929), Sillett et al. (2015) showed unequivocally that old trees produce more wood than younger, smaller trees because of their higher leaf and cambium surface area for growth. By sampling wood cores and measuring bole volume at multiple heights, Sillett et al. (2015) documented the distribution of whole-tree wood production and important contribution of the tree crown to total wood volume which explained past observations of small annual growth increments at the tree base. In a complementary study, Carroll et al. (2014) decoded climatic signals embedded in *S. sempervirens* tree rings throughout the ecosystem range. Carroll and colleagues created the longest dendrochronological record for the species spanning 1,685 years, overcoming numerous obstacles including commonly missing rings that plagued coast redwood tree ring studies in the past. With the benefit of this new dendrochronological record and detailed whole-tree volume measurements across sites, Sillett et al. (2015) discovered that wood production in old-growth *S. sempervirens* forests throughout the ecosystem range has increased in recent decades, especially since the 1970s in the northern extent of the range.

Environmental changes in the 20th century caused this redwood growth surge, but the relative influence of abiotic factors is unknown (Sillett et al. 2015). Proposed climate changes affecting *S. sempervirens* include warming temperatures, increasing atmospheric carbon dioxide, and reduced air particulates from burning restrictions (Fernández et al. 2015, Sillett et al. 2015). In addition, declining fog in the 20th century (Johnstone and Dawson 2010) may contribute to the recent redwood growth surge because fewer foggy days increases solar radiation and promotes photosynthesis (Sillett et al. 2015).

The assertion that fog decline could stimulate *S. sempervirens* growth contrasted with more than 60 years of research that demonstrated the benefits of fog. Fog interception in the redwood forest and use as a water resource by *S. sempervirens* was first measured at ground level (Azevedo and Morgan 1974, Dawson 1998, Oberlander 1956), but once the canopy became accessible, Stephen Burgess and Todd Dawson made the remarkable discovery of sapflow reversal in *S. sempervirens* crowns during fog inundation (Burgess and Dawson 2004). This documentation of foliar uptake and direct atmospheric hydration in the canopy inspired a new cohort of researchers to study the benefits of fog for *S. sempervirens*. Specifically, it was shown that fog both raised the water status of *S. sempervirens* when soil moisture is low (Earles et al. 2016, Limm et al. 2009, Simonin et al. 2009) and delivered significant nitrogen to the forest (Ewing et al. 2009, Templer et al. 2015).

The short-term benefits of fog have not been refuted, but the recently documented redwood growth surge illustrates the challenge the scientific community faces to integrate our understanding across disciplines of how *S. sempervirens* responds to its complex environment. After nearly 250 years of botanical and ecological investigation into this iconic species, there is still compelling need to advance conservation and restoration science for the coast redwood forest. We know that the remarkable growth, longevity, and environmental resilience of redwood has allowed ancient *S. sempervirens* forests to accumulate record-breaking aboveground biomass of more than 5100 mg ha^{-1} over the millennia (Van Pelt et al. 2016), but the next centuries will be markedly different. The ecosystem’s future depends on how the forest responds to anthropogenic forces of climate change, habitat fragmentation, and biodiversity loss in the years ahead.

To prepare ourselves for managing a vibrant coast redwood forest through the 21st century, we must strive to invest more in critical research fields that will improve our ability to sustain *S. sempervirens* and the ecosystem it defines. First, we need enhanced understanding of genetic adaptations in *S. sempervirens* to anticipate the species’ ability to withstand further environmental change. With a majority of the coast redwood ecosystem undergoing continual forest management, it is critical to assess the genetic conservation status of the species and seek to sustain or restore genetic diversity throughout the ecosystem range to bolster *S. sempervirens* population resilience as conditions change. Second, we need better understanding of biodiversity in the forest to develop conservation strategies that recognize and support a broad range of interdependent native taxa. While there has been recent progress identifying macroepiphytes in the canopy, microbial diversity from the treetops to belowground and food web ecology is still poorly studied in the coast redwood forest. Third, we need to better understand the ecological cycles of fire, carbon, and nutrients that sustain ancient forest conditions so that we can restore vital processes on the landscape. Coast redwood forests today have dramatically reduced carbon storage from past logging activities and experience heavily modified fire and flood regimes, alterations that may further degrade the structure and function of this iconic ecosystem over time unless mediated.

To advance these frontiers of redwood research, we need continued public and private investment in science and a diversified scientific community to accelerate the field forward. Redwood research has proliferated most quickly in the past when new perspectives and methodological innovations entered the scientific community and we must facilitate the same opportunities in the century ahead.

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The Enigmatic Fire Regime of Coast Redwood Forests and Why it Matters[1]

J. Morgan Varner[2] and Erik S. Jules[3]

Abstract

Of perhaps all forests in North America, the fire regime of coast redwoods (*Sequoia sempervirens* (D. Don) Endl.) is most enigmatic. Widely considered a temperate rainforest, a large number of fire history studies depict a forest dominated by frequent surface fire regimes. Coast redwood also has a long list of traits that allow it to persist and dominate under such a chronic fire regime: thick bark, flammable litter, ability to resprout, and rapid pruning. Determining how redwood fire regimes functioned is a major question for restoration and conservation efforts. The origins of frequent fires in redwood fire history studies is often assigned to Native American land uses, with little attention to lightning or the region's fire-prone adjacent ecosystems. Results from the few fires studied in the region suggest that we have much to learn from science and management perspectives about how fire behaves, its effects, and the elements of its enigmatic fire regime.

Keywords: fire-adapted traits, fire history, lightning, Native American fire use, *Sequoia sempervirens*.

Background: Redwoods – Rainforests or Fireforests?

Coast redwood (*Sequoia sempervirens* D. Don) Endl.) ecosystems are often characterized as temperate rainforests (Noss 2000), perhaps in spite of the abundant fire history evidence to suggest another story. Contemporary redwood forests offer many clues to their fire-prone past with large basal hollows ("goose pens") abundant fire scars on bark surfaces and within. These pieces of evidence contrast sharply with the protracted fire return intervals in contemporary redwood ecosystems (Lorimer et al. 2009, Oneal et al. 2006). This disconnect and the other traits of redwoods represent a classic enigma—how did frequent fires burn in these wet forests?

Redwood's Fire Regime

For a species with such a small native range, there have been a surprising number of fire history studies in redwood forests. Kane (these proceedings) reviewed the fire history studies in coast redwood ecosystems, noting their high frequency across the range. Stephens and Fry (2005) plotted seven fire history studies over the range of coast redwood, showing mean fire return intervals (mFRI) that ranged from 2 to 87 years, with most studies showing mFRIs ranging from 6 to 25 years (see also Lorimer et al. 2009). Although less studied than mFRIs, the season of historical fires in redwoods has been recorded in several studies and these have found that fires have been recorded in latewood and dormant periods (Brown and Baxter 2003, Brown and Swetnam 1994, Stephens and Fry 2005). Overall, the fire history studies conducted in redwood forests consistently show frequent fires that contrast sharply with the notion of a rainforest ecosystem.

What Was the Source of Redwood Fires?

A persistent question regarding fire in redwoods is the origin of such a frequent fire regime. We categorize two hypotheses regarding the source of fire in redwoods as endogenous or exogenous sources. Endogenous sources include those origins within redwood ecosystems: humans and lightning. Native American ignitions are widely assumed to be the primary source of the frequent fires found in the fire scar literature (Lorimer et al. 2009). Tribal populations were high in the region

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and the many uses of fire have been highlighted elsewhere. While lightning strikes are less frequent in redwoods than in the fire-prone Sierra Nevada or Klamath Mountains, lightning strikes average 3.0 per km^2 per year in redwoods. These lightning ignitions and more recent human ignitions coupled with the substantial capacity of redwoods to have “holdover” fires in the heavy amounts of downed wood or in crown duff (so-termed arboreal histosols that create “fire caves” in redwood crowns; Enloe et al. 2006) may have been sufficient to support such high frequency fire regimes.

Exogenous, or fires that originated outside of redwood forests, offer another possible source for the high frequency fire history of redwoods. As with endogenous fire, these were either ignited by lightning or humans. As with the redwood region, much of the adjacent inland areas were inhabited by a diversity of Native American tribes (Lorimer et al. 2009, Stuart and Stephens 2006). It is notable that redwood’s adjacent ecosystems include many of the most fire-prone ecosystems in the Pacific west. Mixed evergreen forests and Jeffrey pine (*Pinus jeffreyi* Balf.) woodlands border many northern redwood forests (Stuart and Stephens 2006). Oak woodlands, dominated by open stands of Oregon white oak (*Quercus garryana* Douglas ex Hook.) or California black oak (*Quercus kelloggii* Newb.) abut many central and northern redwood forests. In more southerly sites, upland fire-prone oak woodlands and chaparral neighbor the more dissected redwood stands. Fires that originated in those adjoining ecosystems are capable of burning into redwoods under dominant east winds that characterize most of the region during the late summer-early fall peak fire season. This landscape approach has been little studied, but offers a somewhat novel explanation for the high fire frequencies of the past.

Recent wildfires in redwood ecosystems provide further evidence of these ecosystems’ capacity to spread fire. The 2003 Canoe Fire in Humboldt Redwoods State Park (fig. 1) burned 5,554 ha (13,774 ac) in alluvial flats and upland mixed forests. In 2008, several redwoods wildfires were ignited by the June lightning event, most notable of these were the 1,518 ha (3,750 ac) Orr Fire in Montgomery Woods State Park and the 65,920 ha (162,818 ac) Basin Complex in Big Sur. The large > 50,000 ha (130,000 ac) Soberranes Fire was burning across much of the Basin Fire’s footprint during the 2016 conference. These fires all defied the general contention that fires in redwoods were rare and small.

Figure 1—Photograph of large coast redwood within the 2003 Canoe Fire in Humboldt Redwoods State Park, California. Coast redwood has thick bark, prunes rapidly, and has flammable litter that fuels surface fires in these ecosystems. (L. Quinn-Davidson photo)

Redwood's Fire-adapted Traits

Species traits reflect selection pressures and can often inform species relationships to fire. Coast redwood possesses a number of traits widely held to be reflective of a past of frequent fire. Bark thickness is a primary trait that enables trees to survive heating of underlying cambium and xylem structure. Coast redwood's bark thickness provides an obvious advantage in surface fire regimes. Many fire-adapted species have highly flammable litter that can kill neighbors (Varner et al. 2015). Among the western conifers, coast redwood has the third most flammable litter, behind only ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) and Jeffrey pine, two notable fire-adapted trees (Fonda et al. 1987). Other adaptations that redwood has include rapid pruning, rapid height growth, great maximum height, the capacity to resprout basally and epicormically, and seed germination requirements for exposed mineral soil, and (Agee 1993, Lorimer et al. 2009). Each of these traits may have evolved independently of fire, but collectively they confer an advantage for redwood and are suggestive of its past fire regimes.

Why it Matters

Determining how fires spread and what the ecological consequences of frequent fires were and might be in the future are key questions for the conservation and restoration of redwood ecosystems. If redwoods were dominated by frequent fires in their past, understanding the consequences of those fires may provide us with a better model for how to restore redwood dominance. Frequent fires can consume large wood, particularly decayed wood that is so apparent in many old-growth redwood stands (Graham 2009). Fire also differentially selects understory plant species based on their tolerance to heating and their life history. How might the vegetation of redwood ecosystems differ with fires at the frequency recorded in the fire history record? How might have vertebrates and invertebrates adapted to these fire regimes? These questions become more relevant in light of the number and extent of the recent wildfires in redwoods across the region. Understanding the past effects will increase our ability to predict future changes in redwood ecosystems and perhaps embrace fire as a necessary tool to sustain redwood ecosystems.

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Conservation Strategies – Where We Were and Where We're Going[1]

Tom Tuchman[2]

Abstract

Twenty years ago the Redwood region was dominated by the so called "timber wars". On public lands the Northwest Forest Plan forced a new ecosystem management approach. On private lands, the Headwaters Forest initiative and Redwood Summer events were forcing state and federal regulators and landowners to rethink forest management strategies on private lands. The foundation for these debates focused on traditional views of public versus private ownership and to what degree environmental protection and intensive forestry could be applied.

Twenty years later, the Redwood region has become a leader in developing progressive forestry practices and, importantly, new governance structures that provide the financial flexibility to integrate environmental, social and economic attributes. This presentation will summarize a number of historic events that helped build a new forestry foundation along with new conservation strategies that are bringing people together as opposed to tearing them apart. Opportunities and challenges will also be discussed.

Keywords: *Forest policy, redwood conservation strategies*

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Redwoods Sawed and Saved: What Happened to the Redwoods of Humboldt County?[1]

Jerry Rohde[2]

Abstract

The early day logging of coast redwood in Humboldt County was destructive, dramatic, and dangerous. Giant old-growth redwoods were felled by pairs of "choppers" using double-bitted axes and long crosscut saws. Felling a single tree could take a week. After bucking, logs were initially moved by water to mills on the shores of Humboldt Bay. Logging railroads gradually took over part or all of log transport. In extreme situations, movement of logs required use of inclined railways and trestles built directly over streams. As late as the 1970s, loggers were subjected to a mortality rate of 1.25% per year, a startlingly high figure that meant a timber cutter who worked 40 years in the woods had a 50% chance of being killed.

Decades of unrestrained cutting eventually resulted in effective preservation efforts that saw the creation of several state redwood parks. Early efforts to preserve stands of redwoods resulted in small successes, such as the establishment of Eureka's Sequoia Park. During the 1910s, pressure to protect the trees increased as the new Redwood Highway brought scenery loving tourists to the North Coast. In 1918 activists formed the Save-the-Redwoods League. Subsequent fundraising and land purchases resulted in the creation of Humboldt State Redwood Park, with the first acquisition, the Bolling Grove, dedicated in 1921. Meanwhile, cutting by the Pacific Lumber Company continued near the highway. Ultimately, significant tracts of old-growth forest, mostly west of the Eel River, were protected by inclusion in the park. Areas to the east, such as the drainages of Bridge Creek and Perrott Creek, fell to the logger's ax. The race between sawing and saving was on, only reaching its climax in the 1960s and 1970s with the establishment and expansion of Redwood National Park.

*Key*words: *Redwood history, redwood cultural values*

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Redwoods—Responsibilities for a Long-Lived Species/Resource[1]

Robert Ewing[2]

Abstract

What responsibilities do humans have to ensure that redwoods survive? And what values and strategies are required to accomplish such a purpose? A basic assumption is that the saving of a species, or more broadly of an ecosystem, is ultimately about human survival and that there is a responsibility to use all tools available to this end. To date, our actions to sustain redwoods include setting aside acreage in parks and reserves, sustainably harvesting redwoods for commercial use, and educating ourselves and the public on the benefits and services of the redwood forest. These are current approaches for managing the remaining narrow strip of coastal redwoods running from Monterey to Del Norte Counties in Northern California. Fossil evidence indicates that redwoods have been around for at least 130 million years and once had a distribution across much of the Northern Hemisphere. The historic redwood range was reduced through natural causes such as climate changes, glaciation, and volcanic eruptions. More contemporary disruptions have come from extensive market-driven logging in the early to mid-20th century, the recent conversion of private forestland from primarily a timber to primarily a financial asset, and temperature and precipitation variability due to climate change. Are current strategies adequate to enable the long-term viability of the species? And what if they aren't? As natural and human influences become increasingly intermingled, the redwood community is challenged to think creatively about solutions. The growing consensus on best practices for managing redwood properties—for commercial uses and protection—and for more science to improve these practices is encouraging. Next steps include more effective anticipation of outside threats to redwood viability and a pilot to set reestablishment of redwoods across their prior range in California and beyond. A final hurdle is to fully embrace the long time horizon and flexible outlook required to meet our responsibilities for sustaining coastal redwoods.

Keywords: Redwood conservation, redwood forest policy

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Working Forests, Forest Health and Management Challenges in the Redwood Region[1]

Ken Pimlott[2]

Abstract

As California continues into a fifth year of drought, tree mortality enhanced by the unprecedented bark beetle epidemic contributes to wildfires that continue to increase in frequency and severity. Recent fires have posed increasing fire suppression challenges, life safety concerns, post fire watershed impacts and lasting damage to forested landscapes. The ability of California forestlands to sequester and store carbon has become a matter of national and international significance. Greenhouse gas emissions to the atmosphere are altering the climate, and public and private land managers play a major role in climate adaptation and mitigation responses. Nearly a century of fire exclusion in California, coupled with other management decisions on both private and public land, has resulted in forests that are at an increasing risk of loss due to large-scale disturbances. These high risk conditions cause our forests to be susceptible to catastrophic wildfire and epidemic levels of tree mortality due to drought and insect attacks, both of which are forecasted to get worse with a warmer and drier climate. Rather than being the reliable carbon sink they should be, our forests are now in some years emitting carbon. Ultimately, to counter these trends, forest managers need to significantly increase the pace and scale of the region's forest restoration work, such as prescribed fire. In response to the myriad of state forest health issues, California is activity engaged with efforts such as the Tree Mortality Task Force, the Forest Climate Action Team, Assembly Bill 1492 Multi-Agency Forest Practice coordination, cooperative research and ongoing collaborative efforts to address land use changes such as timberland conversion and urbanization. No single activity is going to solve the wide range of threats to California's forests. It is going to take a balanced approach of all the management options available. Without a balanced and cooperative effort on behalf of all landowners, land managers, stakeholders and special interests, we run the risk of future generations not being able to experience or enjoy the benefits of the forests we enjoy today.

*Key*words: *Redwood conservation, redwood forest policy, forest climate change, forest carbon policy*

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SESSION 1 – Monitoring and Growth

Expanding the Network of Crossdated Tree-ring Chronologies for *Sequoia sempervirens*[1]

Allyson L. Carroll,[2] Stephen C. Sillett,[2] Ethan J. Coonen,[2] and Benjamin G. Iberle[2]

Abstract

Crossdated tree-ring chronologies for the Arcata Community Forest (ACF) and Muir Woods National Monument (Muir Woods) expand the spatial coverage of dated coast redwood (*Sequoia sempervirens* (D. Don) Endl.) series. Crossdating relies on the common pattern of ring-width variation among tree populations, and dated chronologies have many applications, including climate analysis, growth analysis, tree age calculations, reconstructing fire histories, and archeological dating. While coast redwood poses many challenges to crossdating (e.g., ring wedging, locally-absent rings, and discontinuous rings), recent work established a network of chronologies at eight locations along the species' latitudinal distribution upon which we build. Here we used a combination of methods including coring standing live trees at multiple heights, cutting cross-sections from already exposed ends of downed trees along trail crossings, coring downed trees, and removing wedges from stumps using chainsaws. The ACF chronology is currently composed of six standing second-growth trees (1882-2015) and three stumps (1273-1714). We crossdated the stumps using existing chronologies > 36 km away as references ($r = 0.40$, $p < 0.001$, with the composite northern redwood chronology). The Muir Woods chronology is preliminary with only three trees sampled but spanned 1318-2013 and captured the oldest extent of any crossdated coast redwood south of Mendocino County. The ACF and Muir Woods chronologies showed synchrony with other chronologies in their sub-regions of the range, reflecting a shared climate signal. ACF correlated strongest with Jedediah Smith Redwoods State Park ($r = 0.47$), Prairie Creek Redwoods State Park ($r = 0.47$), and Humboldt Redwoods State Park ($r = 0.46$), and Muir Woods correlated strongest with Montgomery Woods State Natural Reserve ($r = 0.50$) and Samuel P. Taylor State Park ($r = 0.48$) (common period 1882–2009; $p < 0.001$). These chronologies add to an increasing inventory of dated redwood tree-ring series for use in research and management.

Keywords: Arcata Community Forest, coast redwood, crossdating, Muir Woods National Monument, *Sequoia sempervirens*, stumps, tree rings

Introduction

A network of crossdated tree-ring series for coast redwood (*Sequoia sempervirens* (D. Don) Endl.) was recently developed, opening up this species for detailed study utilizing accurately dated time series. Carroll et al. (2014) presented eight coast redwood ring indices spanning the latitudinal gradient of the species distribution from the northern rainforests to the warmer and drier southern forests along the Big Sur coast. These chronologies were based on an intensive within-tree sampling regime with multiple tree cores retrieved from regular height intervals along the main stem of standing redwoods in old-growth forests, where core number per tree averaged 11 and tree number per location averaged 10 but ranged from 5 to 22 (Carroll, unpublished data). Although coast redwoods can preserve millennia of tree-ring records, previous attempts at crossdating were generally limited by complex growth rings where the climatically induced ring-width patterns are difficult to decipher. Coast redwood commonly has growth rings that are discontinuous around the circumference of the stem, patterns of ring wedging, and locally absent or "missing" rings (Fritz 1940). Nevertheless, prior to Carroll et al. (2014) some success was achieved in crossdating coast redwood,

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namely Schulman's recognition of some climatically driven ring-width variation (Schulman 1940) and Brown and Swetnam's (1994) crossdated chronology near Redwood National Park (1750-1985) used for a fire history. The eight recently created redwood chronologies range from 358 to 1685 years in length with the maximum at Redwood National Park (Carroll et al. 2014).

The existing coast redwood chronologies have been the basis of several important insights. Inter-annual climate analysis revealed a latitudinal gradient of dendroclimatic response, as redwoods in southern locations exposed to warmer and drier conditions showed stronger ring-width correlations to growing season soil moisture and precipitation and a negative relationship with maximum temperature, while redwoods at the two northernmost locations expressed a positive relationship with minimum temperature (Carroll et al. 2014). Dated redwood chronologies allowed for a long-term view of size-independent wood volume growth that showed increasing tree-level productivity in the recent decades for redwoods in northern locations (Sillett et al. 2015). Dated tree-rings at regular height intervals allowed age calculations for standing redwoods using rates of trunk radius change with the oldest known standing tree being 2510 years at Redwood National Park (Sillett et al. 2015). While many redwood fire histories have relied on ring counts as crossdating can be problematic (e.g., Jones and Russell 2015), fire histories based on crossdated tree-rings are more accurate (Brown and Swetnam 1994). In this respect, crossdated redwood chronologies promise further advancement in this field, as fire scars and other indicators can be dated to a specific calendar year using known chronologies as a reference. Additional areas of research that can be furthered with crossdated tree-ring series include archeological dating of structures and artifacts, dendro-seismological investigations of earthquakes, reconstruction of disturbance histories, and isotopic analysis of wood cellulose.

While the establishment of the recent network of crossdated redwood chronologies provides a strong foundation for temporally-explicit coast redwood research, more work is needed to extend and improve these chronologies. Even though old coast redwoods can be very difficult to crossdate, each new tree adds replication to a growing database. New locations expand the spatial extent of the network, providing reference chronologies for future crossdating and data for redwood growth and climate analyses. The sampling for recent advances involved multiple increment cores taken from the stems of standing redwoods in old-growth forests (Carroll et al. 2014, Sillett et al. 2010, Sillett et al. 2015, Van Pelt et al. 2016). While labor-intensive, this methodology provided the within-tree replication necessary for crossdating and the scale of measurement needed for tree-level analyses. Here we expand tree-ring analysis of coast redwood, including sampling stumps and fallen trees. Specifically, our goals are 1) to create crossdated tree-ring chronologies for Muir Woods National Monument (Muir Woods) and the Arcata Community Forest (ACF) using cores from standing and downed trees, cross-sections from downed trees, and wedge sections from stumps and 2) to test the application of dating unknown redwood ring-width series using existing chronologies from other locations in the range.

Methods

Tree-ring series from coast redwoods were sampled at two locations representing different forest types and sub-regions of the species distribution (fig. 1). Arcata Community Forest (ACF) (40.871° N, -124.071° W) is a second-growth forest with stumps remaining from logging in the 1880s. The ACF is managed by the City of Arcata, California, and lies northeast of Humboldt Bay, ~7 km from the Pacific Ocean separated by flat bottomlands. Tree selection at ACF was driven by the goals of associated research (Coonen and Sillett 2015) while also providing a litmus for redwood crossdating and chronology extension. Muir Wood National Monument (Muir Woods) (37.895° N, -122.575° W) is an old-growth redwood forest under the management of the National Park Service that lies ~14 km north of San Francisco and ~4 km from the Pacific Ocean separated by hills of the Marin Headlands. Tree 76 at Muir Woods was intensively studied as part of the 2014 BioBlitz at the Golden Gate National Recreation Area.

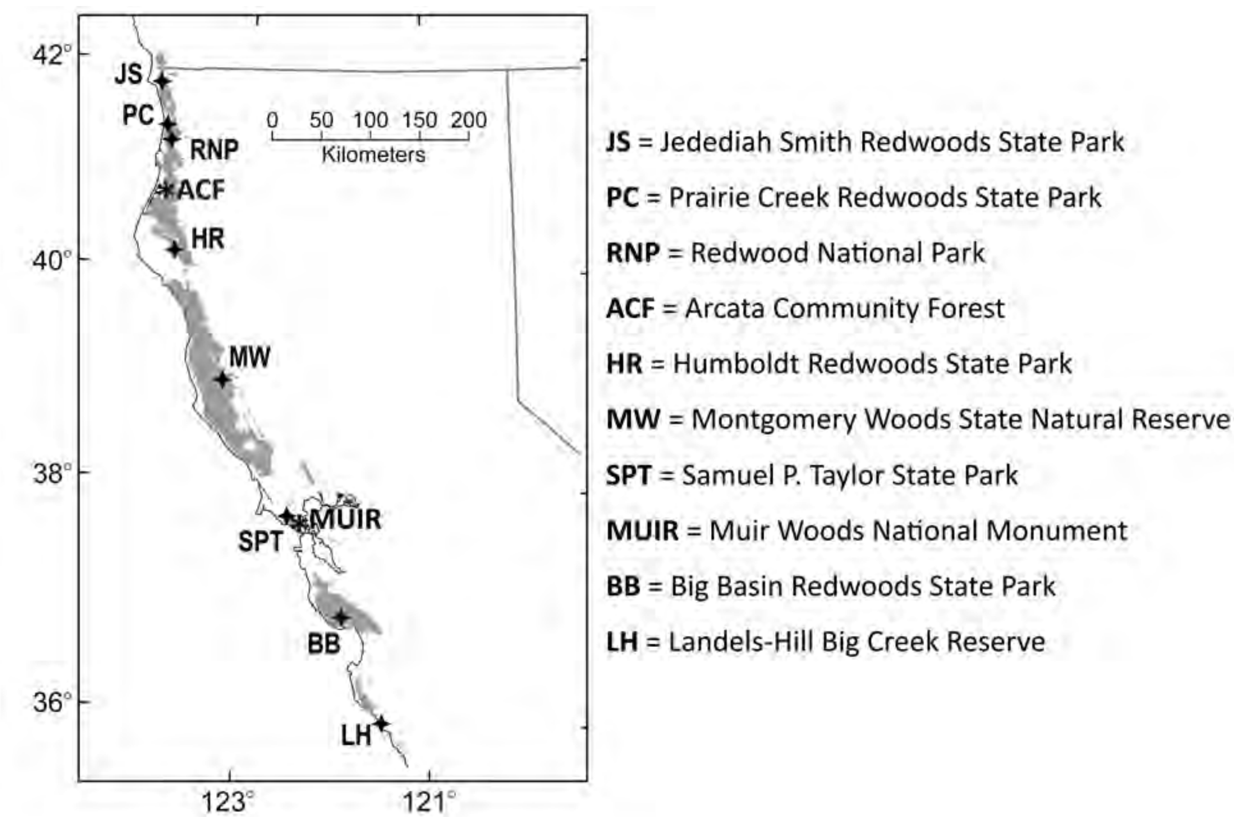

Figure 1—Map showing locations of two new (ACF and MUIR) and eight existing sampling locations. Shaded areas show native range of *Sequoia sempervirens*.

A combination of sampling strategies was implemented at both locations. Tree cores from standing live redwoods were sampled at regular height intervals (~ every 10 m) along the main trunk, accessed by climbers on free hanging ropes using 61 and 81 cm (24 and 32 inch) increment borers. Cross-sections of two recently fallen trees at Muir Woods were sampled from cut-faces where the trees crossed the trail. The Vortex tree fell on June 15, 2011, and the Solstice tree fell on December 21, 2012. The Solstice cross-section was removed at what was 11.3 m above ground level, while the height of the Vortex sample could not be determined. In addition to the cross-section, ten increment cores were taken from the downed trunk of the Solstice tree at opportunistic locations with minimal rot and easy access, ranging from 15.0 to 26.5 m above base. Three stumps at ACF were sampled by extracting wedges of wood via chainsaw. A skilled sawyer utilized angled plunge cuts to sample triangular wedges (58 to 86 cm long) at locations with the most intact wood remaining.

All samples were prepared in the lab by sanding with progressively finer sandpaper to 600 grit after increment cores were glued into wooden mounts. We scanned each sample at 1200 dpi and measured the ring widths to 0.001 mm using WinDendro software (Régent Instruments 2009). Crossdating followed the protocol established in Carroll et al. (2014) that involved a combination of visual techniques and correlation analysis on overlapping segments using Cofecha software (Holmes 1983). Each ring was cataloged according to a classification system for crossdating confidence (Carroll et al. 2014) so that complex series could be parsed into usable sections most relevant for this and ongoing research.

Crossdating followed a hierarchical flow from within trees to among trees within locations by comparing ring-width patterns. Standing live trees and fallen trees with known dates of death were anchored by the last year of growth. The stump samples had only heartwood remaining as bark and sapwood had decayed and, in addition, there were no dated temporal comparisons at ACF. So, we first created a floating chronology linking ring-width patterns among the stump samples and then compared this to the closest dated redwood chronologies and a composite northern redwood chronology to assign calendar years.

Final chronologies were generated for each tree and location to provide a standardized tree-ring index most useful for crossdating and comparison with other chronologies. First, we selected only series with high crossdating confidence ≥ 50 years in segment length. High confidence refers to annually resolved rings and can include missing rings, while moderate confidence captures sections where missing rings could have an alternative placement. An inter-annual chronology was generated for each tree using a 32-year cubic smoothing spline to remove long-term trends using ARSTAN software (Cook 1985). Then, tree-level chronologies were combined into location-specific chronologies with variance stabilized. Since there is one series per tree for stumps at ACF, the stump section was generated with one pass with the 32-year cubic smoothing spline. Although the section 1500-1392 on Stump 955 had moderate confidence in crossdating, it was included in the chronology to provide continuity as the only dated series in this section; all other stump sections were high confidence and segment length ≥ 50 years. Correlation analysis was used to compare the inter-annual variation among the network of coast redwood chronologies including Jedediah Smith Redwoods State Park (JS), Prairie Creek Redwoods State Park (PC), Redwood National Park (RNP), Humboldt Redwoods State Park (HR), Montgomery Woods State Natural Reserve (MW), Samuel P. Taylor State Park (SPT), Big Basin Redwoods State Park (BB), and Landels-Hill Big Creek Reserve (LH) (fig. 1).

Results

Portions of all redwoods sampled at ACF and Muir Woods National Monument (Muir Woods) were dated, exemplifying varying rates of crossdating success from 53.6 to 100.0 percent of available rings annually resolved (table 1). In total we analyzed 12 trees, 99 series, and 3572 rings. For the second-growth trees at ACF, we confidently dated an average of 94.4 percent of the rings for the co-dominant trees with no missing rings compared to 69.3 percent for the subordinate trees with an average of 10 missing rings. The 76 m-tall tree at Muir Woods provided a prime example of thorough dating of a standing live redwood in an old-growth forest from cores taken at multiple heights along the stem, establishing a solid base (1448-2013) for the reference chronology at Muir Woods.

Table 1—Descriptive statistics and crossdating summary for trees at Muir Woods National Monument and Arcata Community Forest

Location	Tree	Tree type	Tree height (m)	DBH (cm)	Tree age	# series	First year	Last year	# rings	% years dated (hi conf)	% years dated (mod conf)	% missing rings
ACF	11*	Live	48.8	68.8	133	13	1884	2013	130	59.6	9.4	2.0
ACF	13*	Live	52.1	77.4	134	11	1882	2013	132	79.0	10.8	0.1
ACF	19*	Live	63.9	130.0	128	10	1889	2013	125	100.0	0.0	0.2
ACF	21*	Live	62.0	175.6	138	10	1893	2013	121	100.0	0.0	1.0
ACF	873	Live	53.1	89.4	~137	12	1882	2015	134	91.2	0.0	0.4
ACF	1081	Live	57.1	139.3	~135	10	1884	2015	132	86.3	0.0	0.9
ACF	955	Stump	–	256.2 ^{f}	>485	1	1273	1757	485	68.7	22.5	0.6
ACF	984	Stump	–	174.7 ^{f}	>322	1	1429	1750	322	66.5	0.0	0.0
ACF	1147	Stump	–	280.6 ^{f}	>196	1	1610	1805	196	53.6	0.0	0.0
Muir	76	Live	76.0	249.5 ^{f}	777	17	1448	2013	566	98.3	1.7	0.7
Muir	Solstice	Fallen	–		–	12	1477	2012	536	55.8	2.5	11.0
Muir	Vortex	Fallen	–		–	1	1318	2010	693	100.0	0.0	0.0

Trees with * detailed in Coonen and Sillett (2015). First and last year based on ring counts.
At ACF, trees 11 and 13 are subordinate crown class, while trees 19, 21, 873, and 1081 are co-dominate.
DBH for stumps is estimated original heartwood size.
DBH with ^{f} denotes functional DBH, defined in Sillett et al. (2015) and Van Pelt et al. (2016).

The two downed trees at Muir Woods illustrated the different possibilities for discerning redwood tree-rings. While only one sample was acquired from the Vortex tree, it had every year present and dated from pith to last year of growth (1318-2010) with no missing rings, making it the oldest known dated piece of coast redwood in the southern portion of the range. In comparison, the Solstice tree had a cross-section and ten cores sampled, with a combined average of only 55.8 percent of the rings annually-resolved with high confidence. The cross-section provided more opportunities to find sections with visible rings, as our path extended from pith to bark with 92.7 percent crossdated and 11 accurately placed missing rings. Sections of all the Solstice cores were dated, ranging from 12.4 to 92.5 percent rings annually resolved on cores that captured 215 to 496 rings. Three cores had > 100 missing rings with a maximum of 148.

We effectively dated the three ACF stumps with between 53.6 to 68.7 percent of the years confidently placed. Notable ring width patterns allowed for the stump rings to be cross-referenced among themselves first, generating a floating chronology (table 2). We then compared the floating chronology to known chronologies and a composite northern redwood chronology (NSESE) to place exact calendar years, first using the clearest section of tree rings that dated to 1501-1713 (table 3). Although the reference chronologies were > 36 km away, relatively strong correlations provided confidence for the dates (e.g., $r = 0.40$, $p < 0.001$, composite northern redwood chronology for common period 1273 to 1714).

Table 2—Stump crossdating showing correlations of 50-year segments lagged 25 years for 1501–1713

Tree	Time span	1501–1549	1525–1574	1550–1599	1575–1624	1600–1649	1625–1674	1650–1699	1675–1713
1147	1610–1713					0.43	0.31	0.41	0.38
984	1501–1713	0.42	0.44	0.33	0.41	0.46	0.53	0.57	0.58
955	1501–1713	0.42	0.44	0.33	0.49	0.46	0.44	0.66	0.56
		0.42	0.44	0.33	0.45	0.45	0.42	0.55	0.51

Bottom row shows average segment correlation.
The preliminary stump chronology was floating and not yet dated to 1501–1714.

Table 3—Stump crossdating showing correlations with known coast redwood chronologies

	stump 955 1501–1713	stump 984 1501–1713	stump 1147 1501–1713	3 stumps 1273–1713
JS	0.40	0.18	0.35	0.31
PC	0.31	0.08	0.35	0.37
RNP	0.24	0.10	0.37	0.30
HR	0.27	0.21	0.01	0.24
NSESE	0.39	0.16	0.35	0.40

We added the two new tree-ring chronologies (ACF and Muir Woods) to the coast redwood network (fig. 2). The 1882-2015 section of the ACF chronology derived from six trees in a second-growth forest, while the 1273-1714 section derived from three stumps. The Muir Woods chronology only has three trees but extends to 1318 and has strong sample depth for Tree 76. These chronologies expand the spatial coverage of coast redwood tree-ring series (fig. 3). When compared to other tree-ring chronologies, ACF correlated strongest with Jedediah Smith Redwoods State Park, Prairie Creek Redwoods State Park, and Humboldt Redwoods State Park ($r = 0.47, 0.47, 0.46$, respectively, $p \leq 0.0001$), and Muir Woods correlated strongest with Montgomery Woods State Natural Reserve and Samuel P. Taylor State Park ($r = 0.50, 0.48$, respectively, $p \leq 0.0001$; table 4).

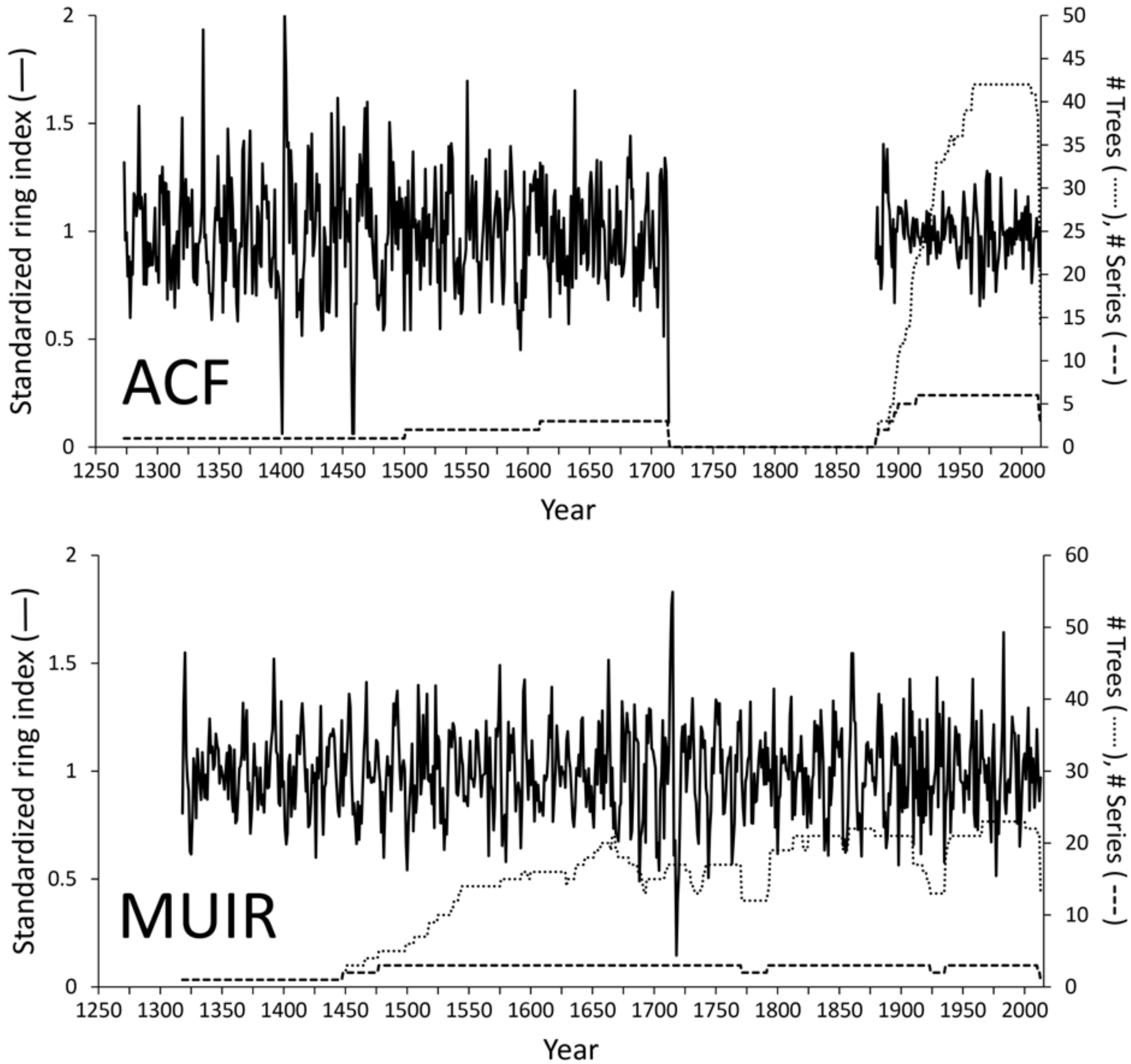

Figure 2—Standardized tree-ring chronologies and sample depths for S*equoia sempervirens* at Arcata Community Forest (ACF) and Muir Woods National Monument (MUIR).

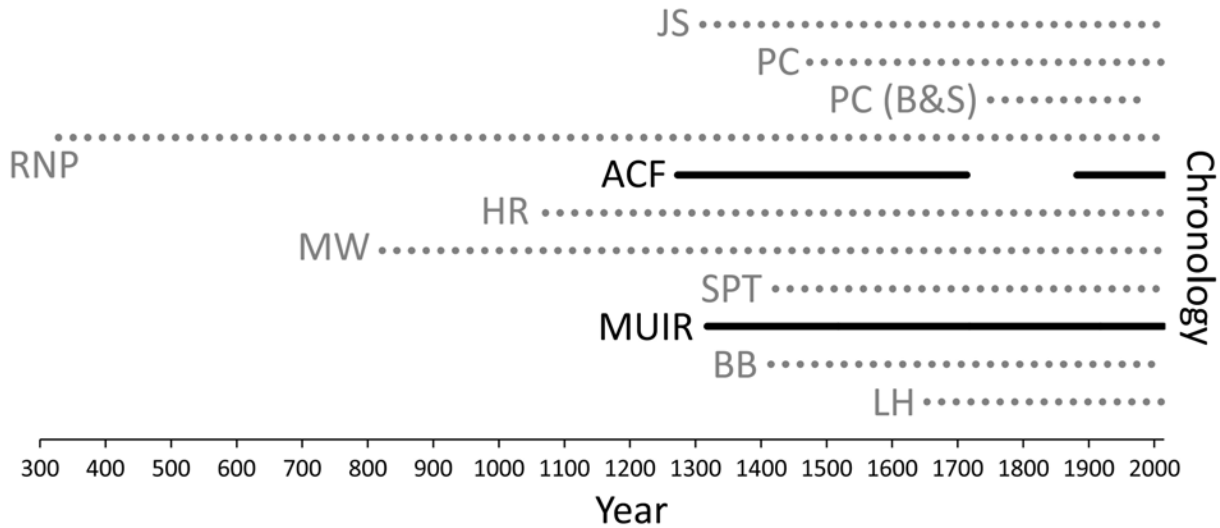

Figure 3—Lengths of crossdated chronologies for *Sequoia sempervirens* at 11 locations listed by latitude. Dashed lines indicate existing chronologies from Carroll et al. (2014) and Brown and Swetnam (1994).

Table 4—Correlations between tree-ring indices of coast redwood chronologies by latitude

	ACF	MUIR
JS	0.47	0.38
PC	0.47	0.31
RNP	0.28	0.21
HR	0.46	0.19
ACF	—	0.31
MW	0.33	0.50
SPT	0.32	0.48
MUIR	0.19	—
BB	0.30	0.43
LH	0.24	0.43

Product-moment correlations (r) for the common period 1882–2009, with statistically significant correlations at r = 0.23, p < 0.01.

Discussion

Progress continues in our effort to expand the network of crossdated coast redwood tree-ring chronologies. The new sampling had varying degrees of crossdating success, ranging from 53.6 to 100.0 percent rings dated, compared to the range of 3.7 to 100.0 percent for 76 trees in other locations (Carroll et al. 2014). Among second-growth redwoods at ACF, crossdating rates differed by crown class, where suppressed trees averaged fewer rings crossdated than unsuppressed trees (69.9 and 94.4 percent crossdated rings, respectively), similar to the results in old-growth forests (66.6 and 87.9 percent crossdated rings, respectively; Carroll et al. 2014). Suppressed trees are affected by competition from neighbors, which influences incremental growth rates (Coonen et al. 2015). Selection of trees, including crown class, can also influence climate-growth relationships (Carnwath et al. 2012) and to further clarify these influences on coast redwood, future sampling will include a broad spectrum of trees.

For old trees, crossdating remains challenging and time-consuming with variable outcomes. The Vortex tree exemplifies one end of the spectrum where one sample captured every ring for from

1318-2010 with a discernable climatically-correlated ring-width pattern. When crossdating trees at Muir Woods, ACF, and the original eight long-term study locations, perfect series such as this one provided the foundation upon which more complex crossdating was built. The Solstice tree represents a common example of redwood crossdating with variable resolution of annual rings; for this tree, sections of annually resolved rings stretched > 200 years separated by decades with no annual resolution and some cores had > 100 locally absent rings. The cross-section provided more opportunities to locate rings compared to an increment core and also accessed tree rings beyond the extent of our longest (81.3 cm; 32 inch) borer. This inner series contained the longest stretch of crossdated rings for this tree, extending 294 years from the pith. Redwood ring series obtained near pith are often more decipherable as ring widths tend to decline as trunks enlarge (Sillett et al. 2015), and tight rings (e.g., < 0.05 mm for Solstice) can be difficult to discern.

Each sampling strategy employed here has strengths and weaknesses. The most thorough method is coring standing redwoods at multiple heights along the main trunk above basal buttressing via rope climbing. This method provides the replication often needed for difficult crossdating at the tree-level, particularly for older trees, but is labor-intensive and requires specialized training. Downed trees and stumps are more easily accessed, but tree selection is limited. Downed trees can be difficult to core as decaying wood can easily jam the borer. While sampling the Solstice tree, our borer jammed with decayed material on several occasions, even though the tree had been on the ground less than two years. Opportunistic sampling of cross-sections of cut faces along road or trail crossings allows for a more complete view of annual rings at a given height than increment cores, and, while limited in selection, can be useful for chronology extension and material acquisition for isotopic analysis. Sampling stumps required a more technical chainsaw cut than the cross-sections.

Utilizing various methods of obtaining tree-ring series from live and dead trees is important as we aim to extend coast redwood chronologies both spatially and temporally, including dating series beyond the oldest material accessible by increment core from living trees. We are building coast redwood chronologies in a manner similar to the multi-millennia giant sequoia chronologies, combining samples from living trees, fallen trees, and stumps (Brown et al. 1992, Douglass 1919). As with giant sequoia (*Sequoiadendron giganteum* (Lindl.) Buchholz), decay-resistant heartwood allows wood to persist many years after tree death.

Crossdating the ACF stumps offered a case study in the effectiveness of dating redwoods with chronologies from distant forests. Dating remnant material using references > 36 km away is a promising example of crossdating in a forest with no living comparable references. However, the nature of coast redwood suggests that success will be variable (Brown and Swetnam 1994, Carroll et al. 2014, Fritz 1940). Preliminary results from a more intensive effort to crossdate downed logs with two increment cores per log revealed comparably low crossdating rates; for example, of 24 downed logs at Humboldt Redwoods State Park, five were dated with high confidence and three were dated with moderate confidence when compared to a reference chronology < 10 km away (Carroll, unpublished data). In general, the correlation of coast redwood chronologies is stratified by distance with differences between the sub-regions of the range, as exemplified by this study (table 4) and Carroll et al. (2014). A notable example was the use of the 1580 as a marker year for the Muir Woods chronology, the year 1580 has the smallest ring width in many giant sequoia chronologies but not northern coast redwood chronologies (Brown et al. 1992, Carroll et al. 2014).

The ACF and Muir Woods ring indices add to the largest archive of crossdated coast redwood tree rings, stored and cataloged at Humboldt State University with a current tally of 354 trees, 2564 series, and 419,024 rings. All samples are curated understanding the value of these resources for further research, especially given the investment in acquiring and dating redwood samples and as new techniques are developed (Creasman 2011). This work at ACF and Muir Woods adds to the network of crossdated tree-ring chronologies initiated with eight long-term research and monitoring plots in old-growth redwood forests under Save the Redwoods League's *Redwoods and Climate Change Initiative*. While tree number is low compared to many dendrochronological studies, each tree adds value given the often complex nature of crossdating this species. More trees will be sampled at ACF in 2017 to enhance this chronology and study the impact of tree crown manipulations on growth (Sillett, unpublished data). We are actively adding chronologies for investigations of growth and climate at new locations, including redwoods sampled in old-growth forests on the eastern extent of the redwood distribution in Napa County, at the Redwood Experimental Forest near the mouth of the Klamath River, and in the Santa Cruz Mountains (Carroll and Sillett, unpublished data).

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Ninety-Two Years of Tree Growth and Death in a Second-Growth Redwood Forest[1]

Benjamin G. Iberle,[2] Stephen C. Sillett,[2] Robert Van Pelt,[2] and Mark Andre[3]

Extended Abstract

Mature second-growth redwood (*Sequoia sempervirens* (D. Don) Endl.) forests are an important and uncommon resource in the redwood region. Development of second-growth redwood forests beyond rotation age is not well understood. Continuous long-term data are especially lacking, considering that the maximum possible age of second-growth stands is now over 150 years. Two observation plots established in 1923 in the Arcata Community Forest provide a unique opportunity to examine the long-term development of second-growth redwood forest.

Dr. Woodbridge Metcalf of UC Berkeley established two 0.4 ha (1 ac) plots on land that was logged in approximately the 1880s. Metcalf and associates tagged and measured all trees every 10 years from 1923 to 1963. Dr. Rudolf Becking of Humboldt State University spearheaded another remeasurement in 1990, but no publication ever resulted from Metcalf or Becking's work. The Metcalf and Becking surveys collected diameter at breast height (DBH; 1.37 m, 4.5 ft) and species for all trees, and height for only some trees. We surveyed the plots using modern methods (precise stem mapping confirmed with LiDAR data, height measurements with laser rangefinders and LiDAR data, crown volume measurements, multiple diameter measurements of buttressed tree bases, and others) and assembled a complete dataset from 1923 to 2015. We utilized new allometric models for Sitka spruce (*Picea sitchensis* (Bong.) Carrière), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), and second-growth coast redwood to predict tree-level quantities such as total biomass and leaf area from functional DBH, diameter at top of buttress, height, and crown volume (Sillett and Iberle, unpublished). The hierarchical sampling methods used to obtain the datasets for these models and the model construction are similar to methods described in Sillett et al. (2015) and Van Pelt et al. (2016).

Both plots approximately doubled in total basal area over the study period, ending at 124 $m^2\ ha^{-1}$ (538 $ft^2\ ac^{-1}$) in Plot 1 and 142 $m^2\ ha^{-1}$ (624 $ft^2\ ac^{-1}$) in Plot 2. Redwood is becoming more dominant in the plots, especially beginning around 80 years in stand age. This trend is apparent for proportion of basal area (fig. 1, bottom) and is also reflected in estimated variables (total mass, wood volume, and others). Stem density decreased over the study period, from 435 to 282 trees per hectare (TPH) (176 to 114 trees per acre, TPA) in Plot 1 and 596 to 356 TPH (240 to 144 TPA) in Plot 2. The non-redwood species are slowly dropping out of the plots, while redwood numbers stabilized in the last 50 years due to ingrowth (fig. 1, top). Causes of mortality were not recorded during surveys, other than notable windthrow events between 1943 and 1963 (stand age of 63 and 83 years). These disturbances dominate the net growth trajectory in the twenty-year period, as seen in the plateau in total basal area (fig. 1).

The only similar long-term dataset for second-growth redwood forest is Dr. Emanuel Fritz's "Wonder Plot" in Mendocino County, famous for a high rate of tree growth (Allen et al. 1996, Fritz 1945, Gerhart 2006). Established by a collaborator of Metcalf, also in 1923, this pure redwood forest on an alluvial plain provides an interesting comparison with this study's mixed-species forests located further north in Humboldt County. Stem density in the Wonder Plot is higher than the Metcalf plots around 60 years in stand age, but steeply drops to similar levels by 130 years (fig. 1, top). Tree growth in the Wonder Plot outpaced the Metcalf plots by a large margin, with approximately one-

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and-a-half times the basal area at equivalent stand ages. However, a windthrow event in 1998 all but erased that margin. The most recent Wonder Plot survey found a total basal area that will be nearly equivalent to Metcalf plot totals if their growth trend continues (fig. 1, bottom). Nevertheless, the higher growth rate of the Wonder Plot is unsurprising considering the highly productive alluvial site chosen by Fritz.

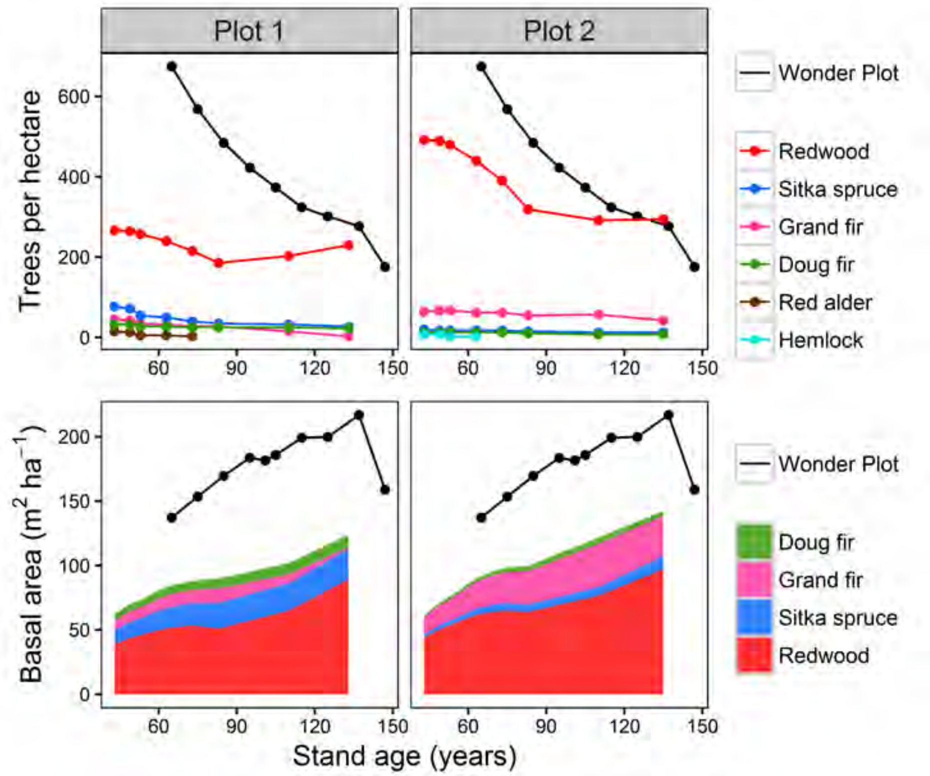

Figure 1—Basal area and stem density over time in the Metcalf plots in Arcata, California. The Wonder Plot in Mendocino, California, a pure redwood stand, is presented for comparison (data from Gerhart 2006). All three plots are 0.4 ha (1 ac).

Our application of allometric models predicting total mass and leaf area also allow for comparison to old-growth coast redwood plots described in detail by Van Pelt et al. (2016). We selected four plots from Van Pelt et al. (2016) for comparison, two in Redwood National Park and two in Prairie Creek Redwoods State Park, which are the closest geographically and ecologically to the Metcalf plots. At approximately 135 years of stand age, both Metcalf plots are remarkably close to their old-growth counterparts in leaf area index of trees alone. Unsurprisingly, the Metcalf plots are far behind the old-growth plots in total aboveground tree mass, with the old-growth plots being between three and four times heavier (fig. 2). While the second-growth forests have similar photosynthetic capacity after 135 years of growth, the trees in old-growth forests have been applying that capacity for many centuries and storing the energy in decay-resistant heartwood.

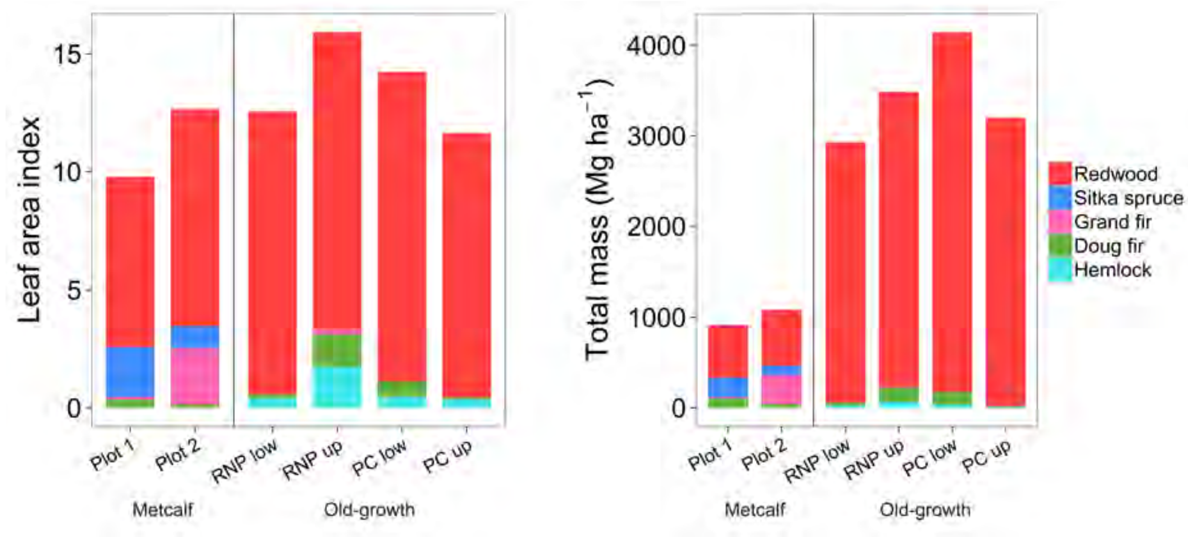

Figure 2—Total tree mass and leaf area index of the Metcalf plots (Arcata, CA) at ~135 years of stand age compared with two lowland and two upland plots in old-growth forest from Van Pelt et al. (2016) in Redwood National Park (RNP) and Prairie Creek Redwoods State Park (PC).

The Metcalf plots have shown strong growth over the study period, although not approaching the rates achieved by Fritz's Wonder Plot. This is despite heavy human use within and around the plots, even predating the establishment of the city park in 1955, according to survey notes. The undergrowth is clearly reduced by formal and social trails in the plots, particularly in the upper plot, and soil compaction is likely. Nevertheless, the Metcalf plots are a rare long-term example of northern second-growth coast redwood forest dynamics and can serve as a benchmark for comparison. Restoration of old-growth forest attributes in second-growth forests through silvicultural manipulation is of increasing interest, as seen in the Silviculture section of these proceedings. Our results from relatively unmanaged conditions can be compared to such studies, especially as treated stands move beyond rotation age. The large non-redwood component occupying growing space in the Metcalf plots, as well as the exceptional example of growth in the Wonder Plot, indicate opportunities for acceleration of second-growth redwood forests toward old-growth characteristics.

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Predicting Redwood Productivity Using Biophysical Data, Spatial Statistics and Site Quality Indices[1]

John-Pascal Berrill,[2] Kevin L. O'Hara,[3] and Shawn Headley[4]

Abstract

Coast redwood (*Sequoia sempervirens* (D. Don) Endl.) height growth and basal area growth are sensitive to variations in site quality. Site factors known to be correlated with redwood stand growth and yield include topographic variables such as position on slope, exposure, and the composite variable: topographic relative moisture index. Species composition is also a key driver of redwood stand growth and yield. We studied spatial patterns in species composition in terms of percent hardwood, and spatial patterns in topographic relative moisture index values across 109 ha (270 ac) of coast redwood forest on Jackson Demonstration State Forest in Mendocino County, California. We also examined how redwood height growth (in terms of site index) and basal area productivity varied across the study area. We performed Ordinary Kriging in ArcGIS to interpolate between plots. These continuous raster data sets were used to create contour maps to assess redwood productivity in the study area. These example applications demonstrate a potential framework and method to estimate forest growth, yield, and carbon stocks in natural forests along gradients of productivity.

Keywords: GIS, interpolation, Kriging, semi-variance analysis, *Sequoia sempervirens*, uneven-aged, variogram

Introduction

Combining forest inventory and remotely-sensed data offers opportunities to improve estimates of forest productivity and reduce sampling intensity and cost. To date, the potential for interpolation of measured and derived variables across forested land has not been widely studied or applied in regenerating mixed stands dominated by coast redwood (*Sequoia sempervirens* (D. Don) Endl.) in north coastal California.

Redwood is shade tolerant and long lived. Redwood height growth is sensitive to overhead shade (O'Hara and Berrill 2010). Unknowingly, foresters may sample dominant redwood trees for site index estimation that have undergone a period of height growth suppression. Site index estimates are also sensitive to measurement error, height growth prediction error, sample size, and method of calculating average height for the larger trees in the stand (Garcia 1998). Redwood site index can vary widely within stands; within a 109 ha (270 ac) stand the range of plot estimates for redwood site index diverged by a factor of two (i.e., range between 20.4 to 40.5 m (67 ft to 133 ft) at base age 50 years; Berrill and O'Hara 2014). Stand basal area and volume increment also vary in space, in large part due to variations in species composition. Focusing on one dominant species, we also find that redwood component basal area and volume increment also vary spatially according to differences in site quality not explained by site index (Berrill and O'Hara 2016). This inherent variability creates challenges for sampling and inventory, especially in managed stands further complicated by harvesting, growth, and regeneration which alter stand density, stand structure, and spatial patterns of tree locations.

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Sampling efficiency has been studied in redwood and other forest types. Sample precision increased as more redwood trees were sampled in fixed area plots (Berrill and O'Hara 2012). Markedly different sample sizes may be needed to achieve a specified level of accuracy or precision for different attributes or in stands with different tree sizes (Gray 2003). Berrill and O'Hara (2015) showed that redwood site index was more variable over short distances (<198 m or <650 ft) than other redwood productivity index variables. Their semi-variance analysis indicated that all indices of productivity were spatially auto-correlated but variable between nearby sample plots (i.e., at smaller spatial scales). Dominant redwood height growth lacked spatial continuity beyond 198 m (650 ft, 1/8 mile), indicating that estimates of site index from plots closer than 198 m apart would be spatially autocorrelated (i.e., plots were not independent samples). Basal area development in plots was spatially autocorrelated over greater distances within a study area with variable terrain and species composition (Berrill and O'Hara 2015). These findings suggest that estimates of redwood site index demand greater sampling intensity than sampling to index basal area or volume increment per hectare.

Understanding how productivity and associated biophysical variables vary in space will support design of efficient forest inventory and allow for improved calculation of stand totals for forest cover estimation and forest valuation for wood products or carbon offsets. The objective of our applied study was to demonstrate how sample estimates for key drivers of redwood stand growth and yield, and productivity estimates, varied spatially by mapping these variables in space by interpolation between known points on a sample grid.

Methods

Our 109 ha (270 ac) study area at Railroad Gulch (latitude 39^o 19′ 47″ N, longitude123^o 41′ 50″ W) near the southwestern edge of Jackson Demonstration State Forest (JDSF), Mendocino, California, is covered by a grid of 234 0.04 ha (1/10 ac) sample plots. Tree measurements were summarized to give stand data, productivity, and site quality indices: redwood site index (Wensel and Krumland 1986) and redwood basal area index (BA Index; Berrill and O'Hara 2014). Species composition in each plot was calculated as percent of total BA (all species in plot) in the regenerating stand prior to partial harvesting in 1982. Redwood site index was calculated by averaging site index for four redwood trees of dominant or codominant crown class in each plot. The site index for each tree was calculated using tree height data and breast height age taken from bark-to-pith increment cores (Wensel and Krumland 1986). Redwood BA Index was calculated for each tenth-acre plot by comparing the redwood component BA (per unit area) to the average redwood component BA in all plots for that density (Berrill and O'Hara 2014). Biophysical data were collected in the field or derived from a 10 m digital elevation model (DEM) for each sample plot at Railroad Gulch (Berrill and O'Hara 2016). Topographic relative moisture index (TRMI) was calculated for each plot using field-based measurements of aspect, slope profile curvature, slope steepness, and position on slope (Parker 1982).

We developed semi-variograms depicting covariance between pairs of plots for estimates of percent hardwood, TRMI, redwood site index, and redwood BA Index (Isaaks and Srivastava 1989). Next we fitted spherical variogram models to each semi-variogram, and used the models for Ordinary Kriging interpolation of each variable across the entire study area. The interpolated data (i.e., Kriging means) were displayed as color contour maps with higher and lower levels of each variable depicting spatial patterns and variability.

Results

Mapping the Kriging interpolation of percent hardwood indicated that hardwoods were more common on some, but not all, south-facing slopes and were concentrated in small- and large-sized patches (fig. 1). Hardwood-dominated sample plots that were surrounded by plots with a much lower hardwood component are depicted as a dark spot on the map of interpolated values. Conversely, a light-colored circle formed around plots without hardwood located in areas where most other plots had a hardwood component. Adjacent to these locations of unusually high or low hardwood percent, the interpolated values of hardwood percent changed rapidly over short distances from these plots with locally-dissimilar species composition.

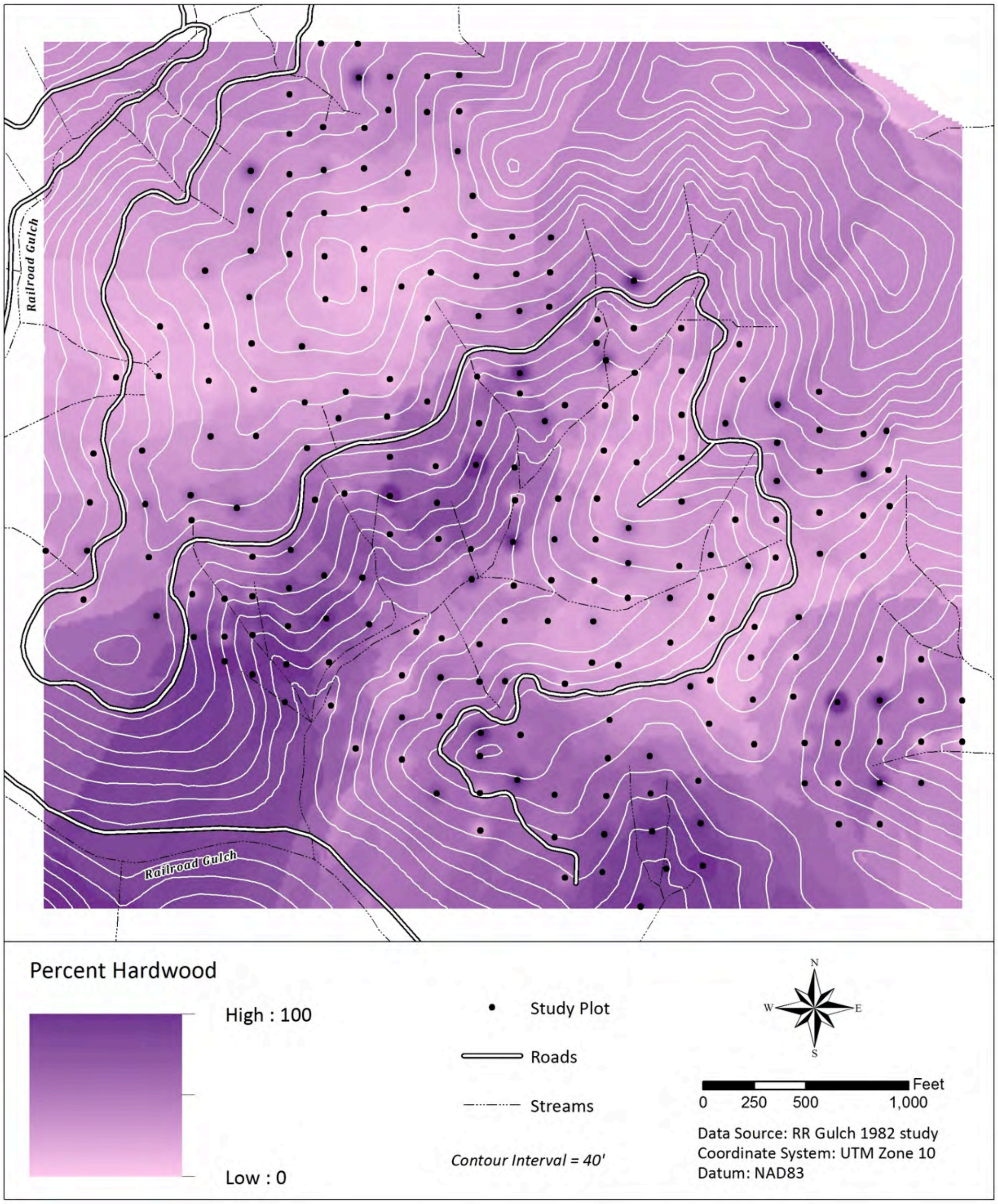

Figure 1—Kriging interpolation of species composition in terms of percent hardwood basal area in sample plots immediately prior to partial harvesting in 1982 at Railroad Gulch, Jackson Demonstration State Forest, Mendocino County, California.

Mapping the Kriging interpolation of plot estimates for TRMI showed how relative moisture related to topography across the Railroad Gulch study area. Topographic relative moisture index (TRMI) calculated for each plot using field-based measurements of aspect, slope profile curvature, slope steepness, and position on slope varied widely throughout the site, but did not exhibit high variability at smaller spatial scales (fig. 2).

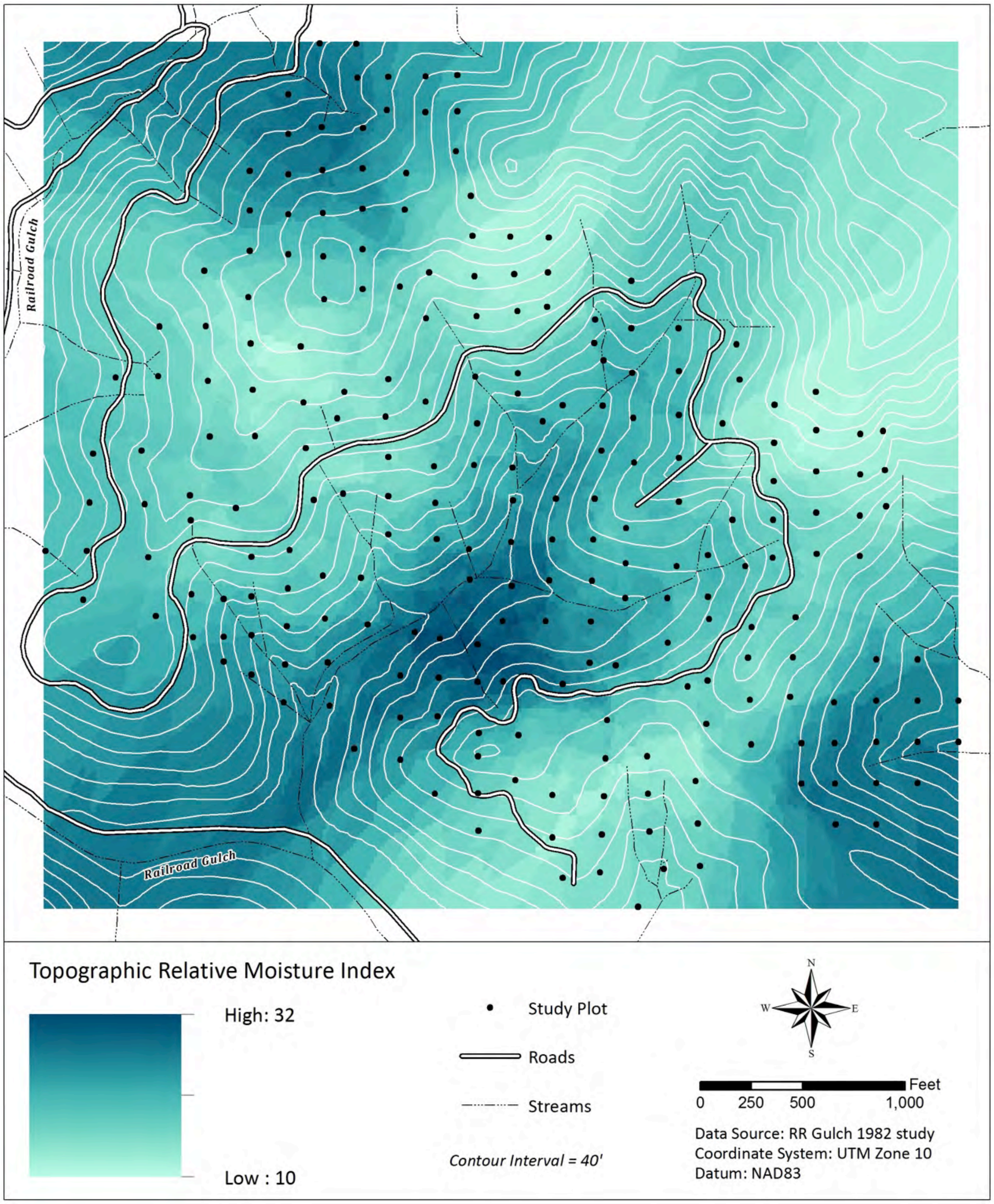

Figure 2—Kriging interpolation of topographic relative moisture index (TRMI) calculated for each plot at Railroad Gulch using field measurements of aspect, slope profile curvature, slope steepness, and position on slope (Parker 1982).

Redwood site index was strongly tied to topography, in particular the position on slope (i.e., gradient of ridge-upper slope-midslope-lower slope-gully). Site index was much lower in exposed ridgetop locations (fig. 3). Redwood BA development in each plot prior to partial harvesting in 1982 was indexed relative to the site-wide average BA per unit area for the redwood component in each sample plot. Kriging interpolation of the index values for BA Index in each plot (fig. 4) revealed that BA productivity did not necessarily coincide with high site index – itself a measure of redwood height development.

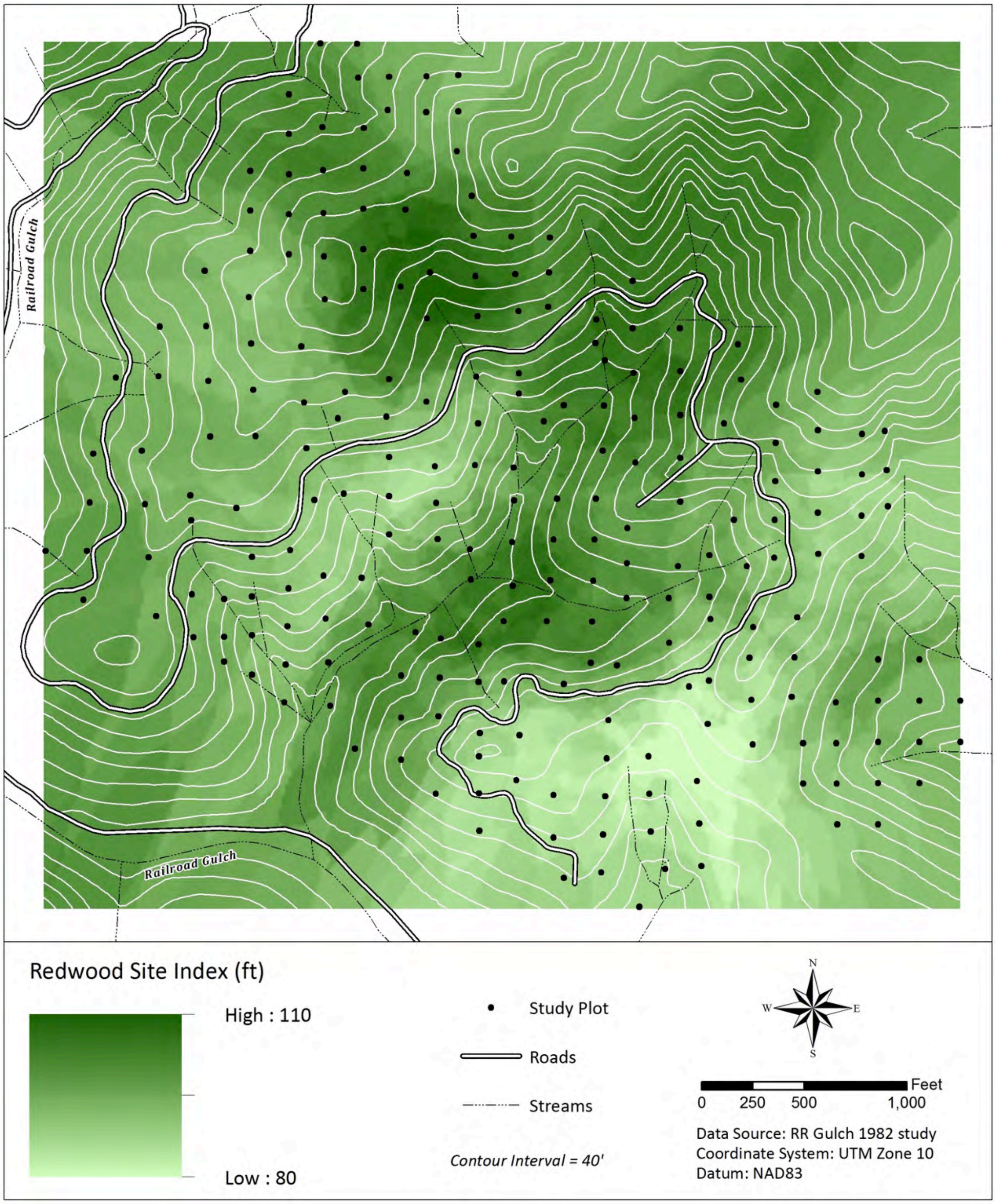

Figure 3—Kriging interpolation of redwood site index (base age 50 years) in sample plots at Railroad Gulch.

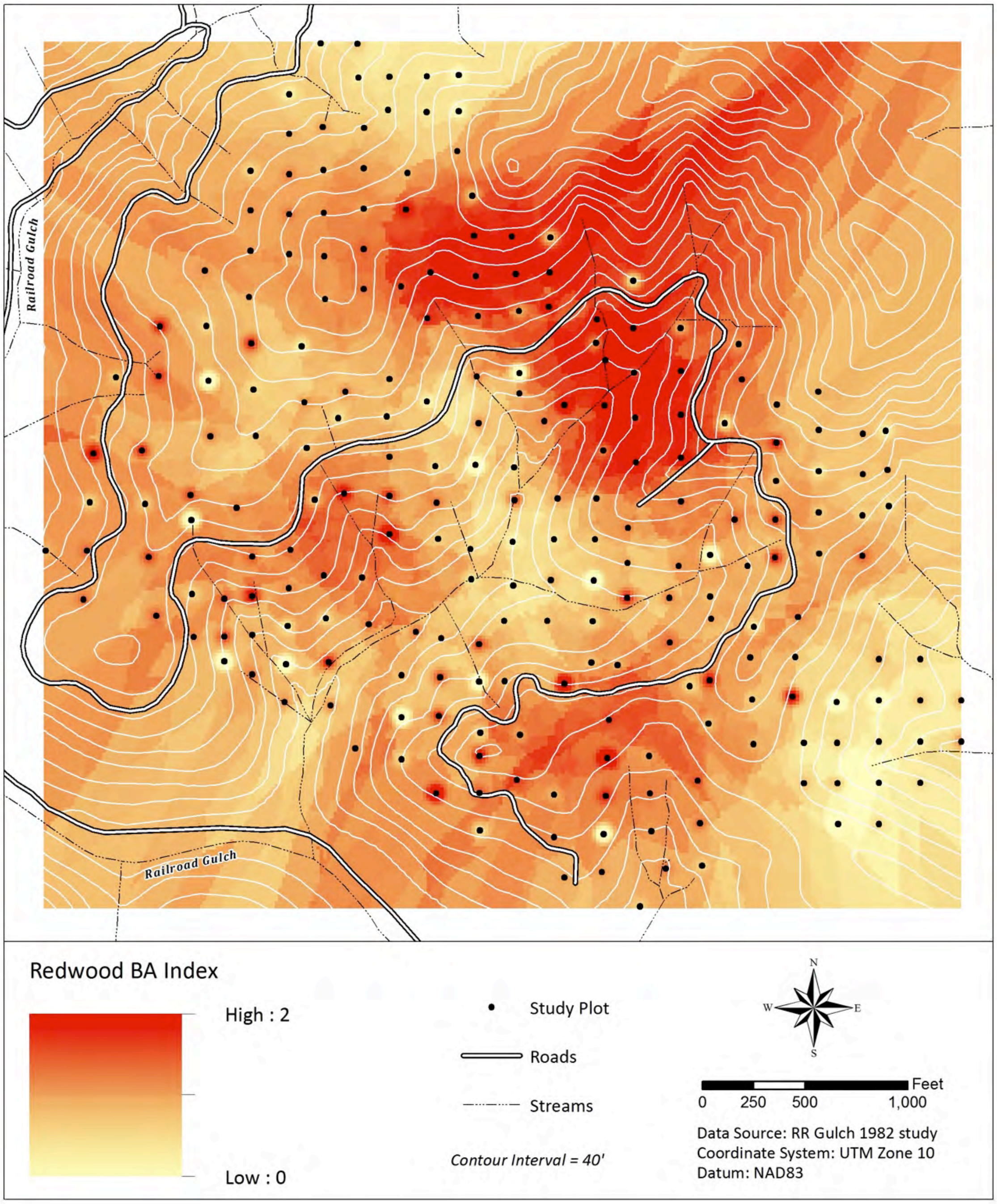

Figure 4—Kriging interpolation of redwood productivity in terms of pre-harvest age-60 redwood basal area index, the SDI-adjusted redwood basal area in sample plots immediately prior to partial harvesting at Railroad Gulch in 1982 relative the site-wide average redwood BA.

Discussion

Variations in species composition (see fig. 1) were identified as the main source of variation in stand productivity at Railroad Gulch; areas with more hardwood were less productive, and areas with more Douglas-fir were more productive on a per-acre basis (Berrill and O'Hara 2016). Interpolation and mapping of species composition variables such as percent hardwood allows the forester to readily observe patterns across the landscape, stratify inventories, and design silvicultural prescriptions best suited to meet market demand and objectives of management in mixed stands.

The Kriging interpolation of percent hardwood revealed small patches of hardwood associated with a single sample plot, and larger patches of similar hardwood composition encompassing several neighboring plots (fig. 1). We were able to show these patterns and conclude that species composition is spatially heterogeneous because of our high density of sample plots. If one applied this interpolation technique between plots spaced further apart, larger patches of hardwood would be detected but smaller patches occurring outside sample plots would not be detected or shown. Basal area at age 60 was also spatially heterogeneous, varying widely between some adjacent plots (fig. 4). Presumably this related to many factors including species composition and site quality differences affecting BA growth, or insufficient plot size for sampling clumped tree spatial patterns (Berrill and O'Hara 2012), but may also be a reflection of localized disturbances and different rates and patterns of establishment affecting development of the regenerating second-growth stand.

Topographic relative moisture index (TRMI) was another important variable correlating with stand growth and per-acre growth of the redwood component at Railroad Gulch (Berrill and O'Hara 2016). The TRMI can be calculated using field-based measurements and assessments of slope variables and aspect, and then interpolated between known points. A more cost-efficient approach is to derive relative moisture indices directly from digital elevation models, especially for inaccessible/remote study areas and where high-quality DEM coverage is available. The TRMI is a composite index that integrates key topographic variables, such as position on slope, known to be correlated with redwood height growth and BA growth (Berrill and O'Hara 2016).

Other important corollaries with redwood growth can be derived from DEMs (e.g., topographic exposure; Quine and White 1998), remotely sensed via multispectral imagery (e.g., species composition), or taken from GIS map layers (e.g., geology and soil properties). Some of these variables correlate with either or both redwood height growth and redwood BA growth, which in turn vary widely at small spatial scales (Berrill and O'Hara 2014, 2015) and proceed somewhat independently presenting challenges for forest inventory and modeling (Berrill and O'Hara 2016).

Conclusion

Kriging interpolation of key drivers of productivity and/or productivity index values gives a meaningful basis for estimating and summing stand growth and yield across any area with DEM coverage and interspersed with sample plots (or other data sources) giving species composition, tree- and stand data. Redwood forest managers and researchers are cautioned that forest growth and yield is highly variable at small spatial scales and that redwood site index cannot be expected to correlate well with stand BA and volume or the growth and yield of the redwood stand component.

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On the Variation of Inventory Estimates for Redwood Stands[1]

Daniel Opalach[2]

Abstract

There is a tremendous amount of variation in the second growth redwood (*Sequoia sempervirens* (D. Don) Endl.) forests found along the North Coast of California. In order to make prudent management decisions about these forests, foresters and other resource professionals often conduct timber cruises to collect information on tree species, diameters, tree heights, crown ratios, bole taper, and other variables of interest. When designing point sample timber cruises to collect these data, it is important not only to consider sample size (the number of point samples), but to also consider how frequently to subsample trees for volume measurements. The purpose of this paper is to report on a study where Green Diamond simultaneously assessed the effects of both of these key timber cruise design elements on confidence intervals for board foot volume estimates.

Keywords: confidence interval, point sampling, timber cruise, volume estimate

Introduction

Second growth redwood (*Sequoia sempervirens* (D. Don) Endl.) stands commonly found along the North Coast of California can exhibit a tremendous amount of variation under certain stand and site conditions. These stands were usually naturally regenerated following the harvest of the old growth timber and, as a result, typically contain a diverse mix of conifer and hardwood species. Many of these second growth stands were then subjected to partial harvesting operations on multiple occasions to remove overstory trees depending on log markets and the owner's objectives. Natural conifer regeneration following such entries was inconsistent and often hindered by brush and hardwood competition. Further complicating the situation is the presence of sprouting species, such as redwood and tanoak, which lead to a great deal of variation within the stand with respect to the spatial arrangement of the trees. As a result, it can be a challenging exercise coming up with reliable cost-effective inventory estimates.

In this paper, the focus is on a redwood/Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) stand that has all the challenges mentioned above. The stand is located in Humboldt County approximately 14.5 km (9 miles) east of the city of Eureka, California, at 40°47'17"N 123°59'5"W. It was naturally regenerated following the removal of the old growth resource. Then, the second growth resource was partially removed in multiple harvesting operations. Today the stand has a diverse range of tree species, size classes, and age classes haphazardly scattered over the landscape. It is an inventory forester's nightmare.

In order to manage the stand for timber production, Green Diamond requires current inventory information. There is a need to have information on species composition, volume by diameter class, tree ages, and site index. To collect these data, foresters rely on point sample timber cruising. However, given the heterogeneity of this stand, how should a timber cruise be designed? How many point sample plots should be included in the cruise so that acceptable confidence intervals can be constructed? Is it appropriate to consider using a mix of volume plots and count plots and, if so, what should be the ratio between volume plots and count plots? Alternatively, should volume trees be subsampled on each sample point and, if so, to what extent should subsampling occur? The objective of this study was to address such questions by examining confidence intervals and the margin of error

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for board foot (bd ft) volume estimates and how these statistics are affected by sample size and subsampling of volume trees.

Data Collection

The stand selected for this study was 64.7 ha (160 ac) in size. One of the objectives was to examine timber cruise statistics over a wide range of sampling intensities. A 76.2 m x 76.2 m (250 ft x 250 ft) grid was overlaid on the stand that resulted in 93 sample points (see fig. 1).[3] A sample size of 93 is extremely large for a 67.4 ha (160 ac) stand, and is more than enough to examine how volume estimates and their confidence intervals “stabilize” as sample size increases.

To estimate basal area per acre, the timber cruise was conducted using a Basal Area Factor[4] (BAF) of 62.5.[5] With a 62.5 BAF, each “in” tree at a sample point represents 62.5 sq ft of basal area. Every “in” tree was measured for height and diameter so its bd ft volume could be calculated using equations in Krumland and Wensel (1978) for conifers or Naccarini et al. (1979) for hardwoods.

Then, to study the impact of subsampling volume trees, the timber cruiser then determined which trees would be “in” using three additional, bigger BAFs: 184.29, 250, and 360. The mathematics of point sampling is such that as the BAF is increased the probability of a tree being “in” is decreased. Thus, the number of volume trees goes down as the BAF increases from 184.29 to 360. Using a bigger BAF to subsample volume trees is known as the “big BAF method” (Marshall et al. 2004). The big BAF method is used extensively throughout the Pacific Northwest when designing and conducting timber cruises. Under some circumstances, the method can be more efficient than cruise designs that call for volume measurements on 100 percent of the “in” trees (Marshall et al. 2004).

Another popular method for subsampling volume trees is to skip the volume measurements on some of the sample points. For example, the cruise design might specify that the timber cruiser only measure volume trees on every other plot, every third plot, or even every fourth plot—so called ‘volume plots’. With the data set collected as specified above for this study, such an analysis is easily conducted by simply ignoring the volume measurements on some of the plots and analyzing them as ‘count plots’.

[3] Volume information was desired for that portion of the stand outside of watercourse buffers so sample points were intentionally omitted in those areas.

[4] Point sampling utilizes devices to project critical angles that determine which trees are “in” or “out”. The critical angles are selected such that each “in” tree represents a predetermined basal area per acre, which is known as the Basal Area Factor (BAF). Readers interested in reviewing point sampling basics should consult a textbook such as Avery and Burkhart (2002).

[5] Using a 62.5 BAF for this project resulted in an average of 4.4 trees per sample point. This number is acceptable but it is at the low end of the target range of four to eight trees per sample point.

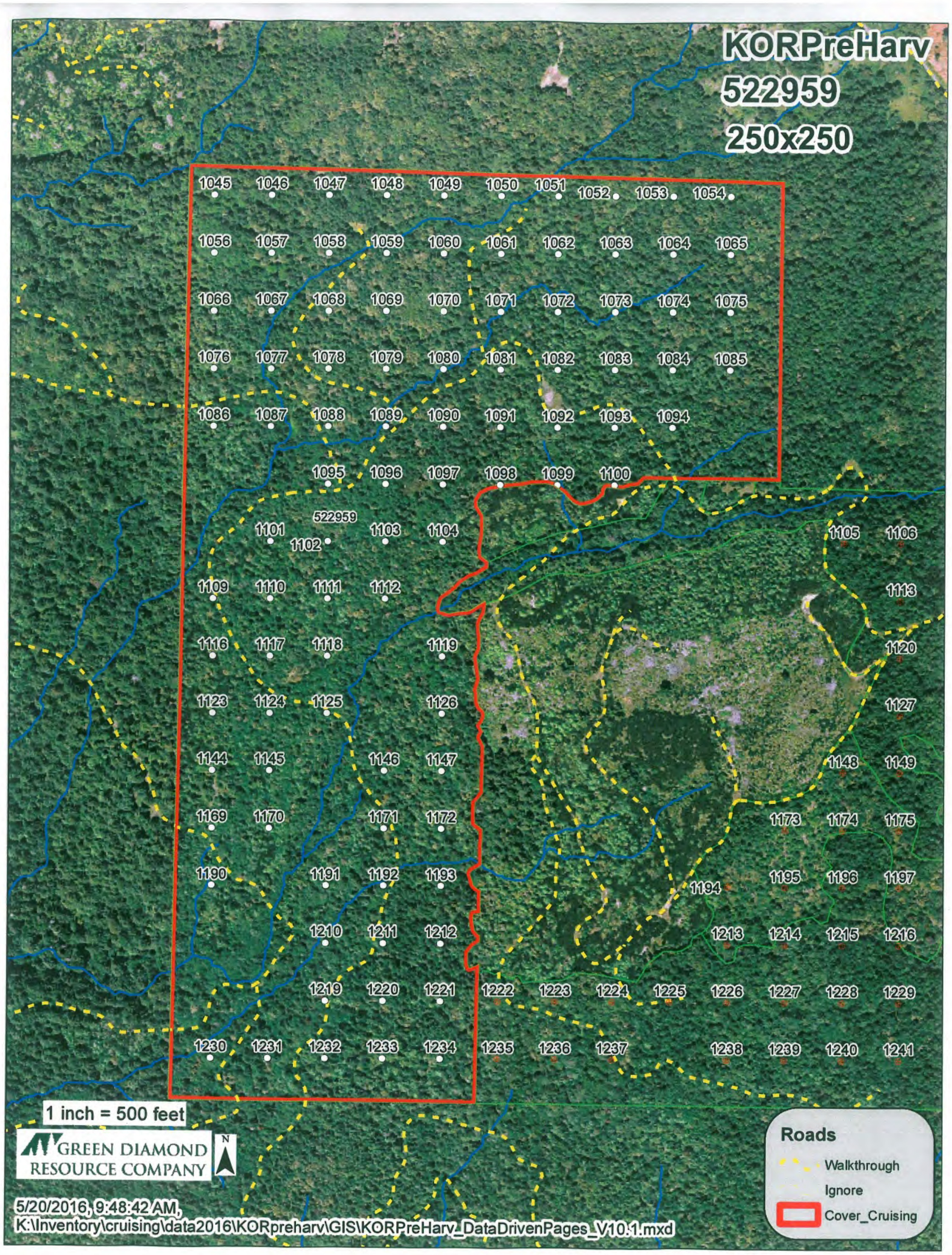

Figure 1—Aerial photograph showing the stand boundary (red lines) and the location of the cruise sample points.

Data Analysis

The timber cruise data were compiled using the variable probability sampling procedures described in Dilworth and Bell (1982). Briefly, the volume per acre is estimated as the average volume to basal area ratio (VBAR) times the average basal area (BA) per acre. That is:

$$\text{Volume per acre} = \text{VBAR} \times \text{BA per acre}$$

Dilworth and Bell (1982) also describe how to calculate standard errors (a) when all plots are volume plots (i.e., all trees on all sample points are measured for volume), and (b) when both count plots and volume plots are specified in the cruise design. Marshall et al. (2004) explain that the same formulas in Dilworth and Bell (1982) can be used to estimate volume per acre and calculate standard errors for the big BAF method.

Results

All Trees Measured for Volume

A unique way to display the results for this timber cruise (or any timber cruise for that matter) is to show what happens to the estimate of volume per acre and the confidence limits as sample size is increased from 2 plots to 93 (fig. 2). A confidence limit (CL) is either of the extreme values of a confidence interval.

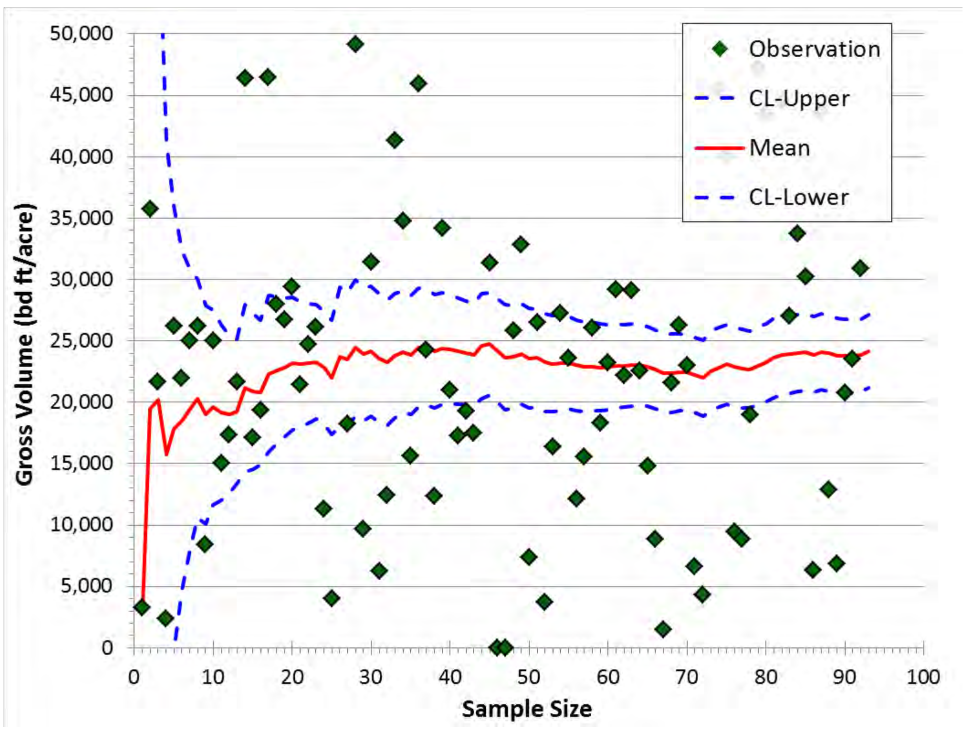

Figure 2—Estimates of mean bd ft volume per acre (red line) and associated upper and lower confidence limits (blue dashed lines) as sample size is increased from 2 plots to 93 plots. For each sample point all the trees where regarded as volume trees and used to calculate the mean and confidence limits. Note the tremendous variation associated with the plot data (green diamonds). Volume per acre varies from 0 bd ft per acre up to 50,000 bd ft per acre.

Rather than plotting the mean and confidence limits, another way to look at these same data is to plot the margin of error. The margin of error is

$$\text{margin of error} = \frac{\text{t value} \times \text{standard error}}{\text{mean}}$$

The t value x standard error is ½ the width of a confidence interval. As you can see, the margin of error is a percentage of the mean. Figure 3 is a chart showing the margin of error for the data shown in fig. 2.

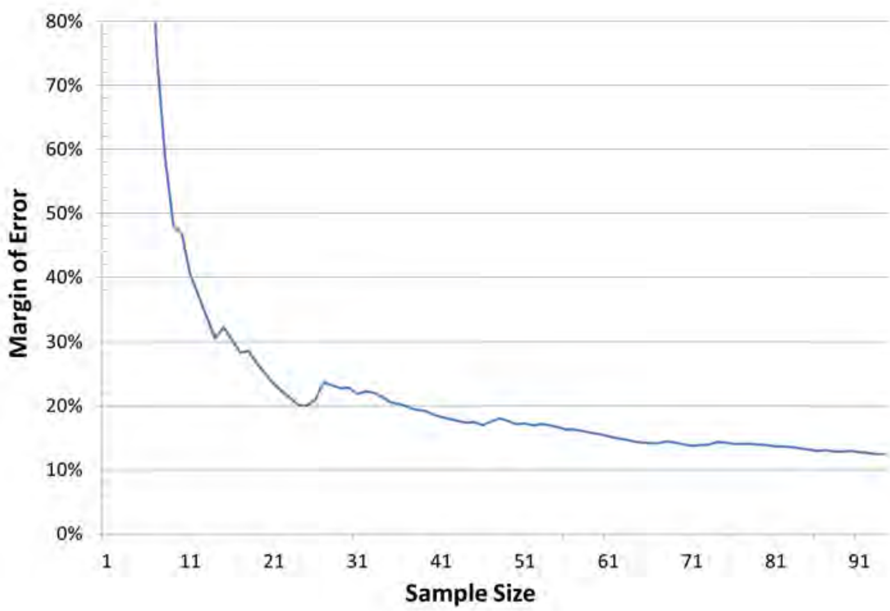

Figure 3—Illustration of how the margin of error decreases as sample size increases. The margin of error is calculated by dividing ½ the width of the confidence interval by the mean bd ft volume per acre. At the risk of being redundant, this chart is just a different way of looking at the same data shown in fig. 2.

The margin of error decreases rapidly as sample size increases from 2 plots to 30 plots. After 30 plots, the margin of error decreases gradually. This pattern is not unexpected. The margin of error is a function of the t value and the standard error, and both of these values decrease as sample size increases.[6]

To recap, figs. 2 and 3 were constructed using all the trees on the plots that were determined to be "in" using a BAF of 62.5 as volume trees. Next, I am going to look at what happens to the margin of error when trees are subsampled for volume measurements. Subsampling should increase the margin of error because fewer volume trees are used in the computation of the standard error. For specifics on calculating standard errors for point sample data, the interested reader should consult Dilworth and Bell (1984) pages 56 to 62.

[6] If you inspect a t table, you will see that t values strictly decrease as the degrees of freedom increases. (Note that degrees of freedom = sample size – 1.) Standard errors, on the other hand, generally decrease as sample size increases.

Impact of Subsampling Volume Trees by Utilizing Count Plots

To assess the impact of utilizing count plots on the margin of error, the 93 plots were segregated into volume plots and counts plots. Four scenarios were analyzed:

Scenario label	Volume plots	Volume plots (%)	No. of volume trees
BAF 62 VOL25%	1 out of 4	25%	126
BAF 62 VOL50%	2 out of 4	50%	228
BAF 62 VOL75%	3 out of 4	75%	313
BAF 62 VOL100%	4 out of 4	100%	412

Margin of error curves for each of these scenarios are shown in fig. 4. It should be noted that the 4th scenario (BAF 62 VOL 100 percent) was already presented in fig. 3. It is included in fig. 4 so the impact of subsampling can readily be compared to the scenario where all the trees are used in the calculation of volume per acre means and standard errors.

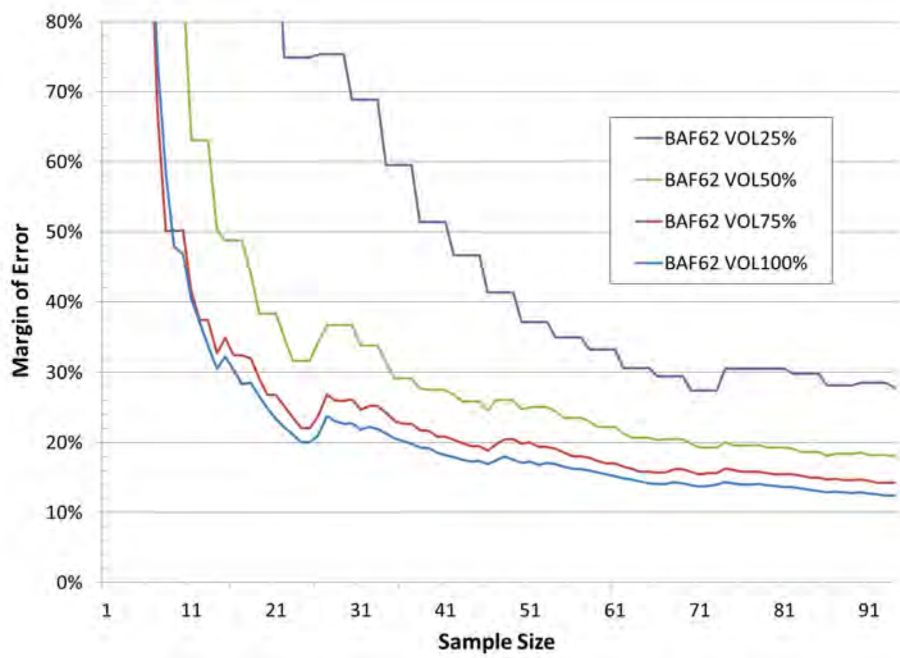

Figure 4—This chart shows the impact of utilizing count plots on the margin of error. The BAF62 VOL100 percent curve is based on all volume plots and the other curves are based on different ratios of volume plots to count plots.

The impact of subsampling volume trees by utilizing count plots in the cruise design is readily observable in fig. 4 for the redwood/Douglas-fir stand featured in this study. For example, if the goal is to obtain a margin of error that is +/– 20 percent of the mean, then that level of precision cannot be achieved with a sample size of 93 plots utilizing a sampling design where one in four plots is a volume plot. On the other hand, +/– 20 percent can be achieved with approximately 30 plots if every plot is a volume plot. If two out of four are volume plots, then the +/– 20 percent margin of error can be achieved with approximately 65 sample plots.

Impact of Subsampling Volume Trees by Utilizing the Big BAF Method

To assess the impact of utilizing the big BAF method on the margin of error, data from the 93 plots were analyzed using a selection of big BAFs to determine which trees to measure for volume. Four scenarios were analyzed:

Scenario label	Big BAF	Basal area BAF	No. of volume trees
BAF 360	360	62.5	81
BAF 250	250	62.5	119
BAF 184	184.29	62.5	162
BAF 62	62.5	62.5	412

Margin of error curves for each of these scenarios are shown in fig. 5. It should be noted that the 4th scenario (BAF 62) was already presented in fig. 3. It is included in fig. 5 so the impact of subsampling (on the margin of error) using the big BAF method can readily be compared to the scenario where subsampling is not utilized.

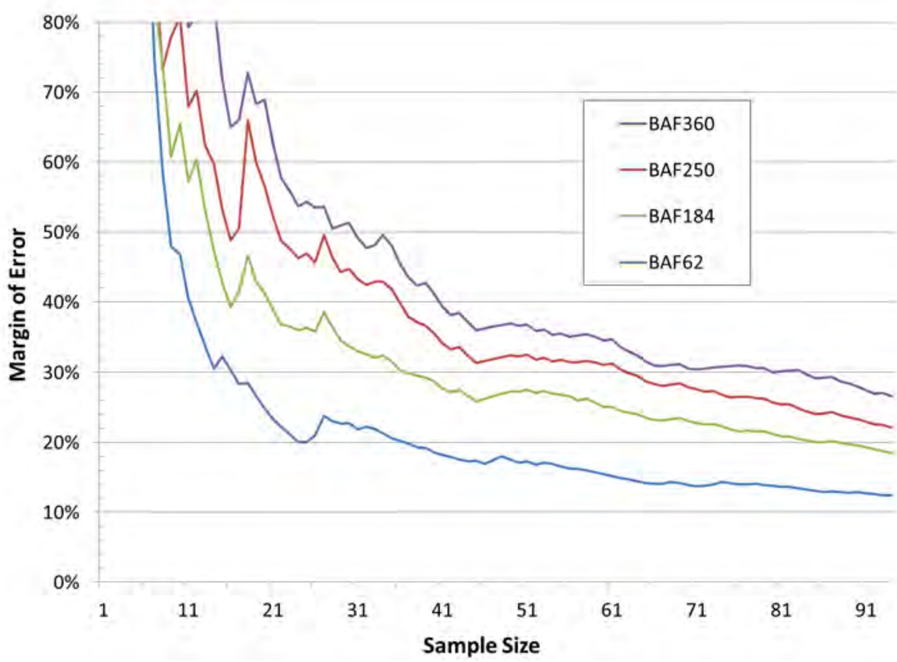

Figure 5—The chart shows the impact of utilizing the big BAF method to subsample volume trees on the margin of error. The BAF62 curve considers all trees as volume trees whereas the other curves subsample volume trees by using a BAF larger than 62.5.

The impact of subsampling volume trees by utilizing the big BAF method in the cruise design is readily observable in fig. 5 for the redwood/Douglas-fir stand featured in this study. For example, if the goal is to obtain a margin of error that is +/– 20 percent of the mean, then that level of precision cannot be achieved with a sample size of 93 plots utilizing a big BAF of 250 or 360. Utilizing a big BAF of 184.29, it takes a sample size of approximately 85 plots to achieve a precision of +/– 20 percent. On the other hand, if every tree is measured for volume +/– 20 percent can be achieved with approximately 30 plots.

Conclusion

Subsampling volume trees can save a forester a lot of time and money under certain stand and site conditions. Two examples are given in Marshall et al. (2004): a Douglas-fir stand near Corvallis, Oregon, and a Jeffrey pine (*Pinus jeffreyi* Balf.) stand near Lake Tahoe, California. In those examples, the population of volume trees exhibited very little variation in cu ft volume to basal area ratios (VBARs). For the redwood/Douglas-fir stand featured in this study, the bd ft VBARs varied by species and within species[7] and as a result, subsampling volume trees had a huge impact on standard errors and confidence intervals.

Based on the results obtained in this study, it is recommended that inventory foresters take the time and effort to understand the impact of subsampling volume trees on the standard errors and confidence intervals for the various stand and site conditions under their management. This is particularly true for foresters who work in the naturally regenerated second growth stands commonly found along the North Coast of California. Under certain stand and site conditions, it may be more efficient to take volume measurements on every tree but put in fewer sample points (see figs. 4 and 5) to achieve the desired precision. On the other hand, for redwood stands that are relatively uniform (such as intensively managed third growth) a cruise design that specifies subsampling volume trees may be a more appropriate choice. Foresters interested in subsampling the optimal number of volume trees should consult Marshall et al. (2004) for the appropriate equations and methodology.

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I want to thank my colleague Mr. Ryan Crans, Green Diamond's Timberlands Inventory Supervisor. Mr. Crans selected the study stand, designed the timber cruise, supervised the timber cruisers and check cruisers, and created an Excel spreadsheet that contained the data from the cruise. I also want to thank Mr. Brian Watson and Mr. John Vona, also Green Diamond employees, who reviewed an earlier draft of this paper and provided helpful comments.

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[7] Interested readers should contact the author for more details regarding the VBARs for the redwood/Douglas-fir stand featured in this study.

Sustainability Analysis Using FORSEE and Continuous Forest Inventory Information to Compare Volume Estimation Methods for the Valencia Coast Redwood Tract in Santa Cruz County, California[1]

Douglas D. Piirto,[2] Mitchell Haydon,[3] Steve Auten,[4] Benjamin Han,[5] Samantha Gill,[6] Wally Mark,[7] and Dale Holderman[8]

Abstract

The 1,295 ha (3,200 ac) Swanton Pacific Ranch (Swanton) and the associated Valencia Tract in Santa Cruz County have been managed by California Polytechnic State University, San Luis Obispo (Cal Poly) since 1987. Swanton's Valencia Tract is a 239 ha (591 ac) property located north of Watsonville, California. Cal Poly forest managers have conducted two harvest entries since acquiring ownership of the Valencia tract utilizing a modified BDq individual tree selection approach. A 10-year continuous forest re-inventory (CFI) was completed for the Valencia coast redwood tract to update and enhance the growth and yield analysis needed for the Non-Industrial Timber Management Plan (NTMP) sustainability analysis.

The California Growth and Yield Modeling Cooperative - Forest and Stand Evaluation Environment (FORSEE) program and 10-year CFI data were utilized to perform a sustainability analysis comparing trees per acre, basal area per acre, quadratic mean diameter, and gross volume per acre. Several tree volume estimation methods were evaluated for differences in yield reporting.

It was determined that: 1) by 2012 actual stand volume growth had completely recovered from harvest and exceeded the pre-harvest 2001 gross volume by 9.7 percent (i.e., average stand growth of 1,266 board feet per acre per year, or 3.0 percent growth rate per year); 2) the Spaulding equation appears to be a solid medial choice for Valencia Tract sustainable yield analysis; 3) sustainable uneven-age stand management resulted from setting residual stand basal area (b), maximum diameter (d) and trees per acre by diameter prescription targets (q) while leaving a few trees greater than the established maximum diameter (i.e., a modified BDq approach); 4) the project model underestimated basal area per acre growth, overestimated change in quadratic mean diameter (QMD), overestimated diameter growth of smaller trees (< 30.5 cm (12 inches) diameter at breast height (DBH; 1.37 m)); underestimated diameter growth of larger trees (> 33 cm (13 inches) DBH), underestimated height growth of larger trees (> 53.3 cm (21 inches (DBH), underestimated volume (16.3 percent lower than actual CFI volume figures). It is postulated that this difference in FORSEE model projection is either a result of the way the CFI data was processed in FORSEE, regional differences, or inherent projection inaccuracies not fully understood by FORSEE users.

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Keywords: coast redwood, continuous forest inventory, FORSEE, growth and yield model, *Sequoia sempervirens*, sustainability analysis, uneven-age forest management, volume equation comparison

Introduction

Forest management at California Polytechnic State University, San Luis Obispo (Cal Poly)'s Swanton Pacific Ranch (Swanton) and School Forest began in 1986 when owner Mr. Al Smith requested the university's assistance with management of his agricultural and forested properties. Mr. Smith bequeathed those properties to Cal Poly in 1993. Mr. Smith's long-term vision focused on Swanton Pacific Ranch and the Valencia Tract being a sustainably managed working ranch and forest with many interdisciplinary Learn by Doing activities involving students, staff, and faculty.

Swanton Pacific Ranch is located in the southern sub-district of the Coast District (Cal Fire 2016). Very strict forest practice rules were developed for this district due to citizen concerns about the extensive clearcut logging that occurred in the early 1900s to help rebuild San Francisco after the 1906 earthquake. These current, sub-district California Forest Practice Rules specify tree removal limits by diameter class, maximum permitted opening size, and Watercourse and Lake Protection Zone requirements. Sustainable management of the working forested areas of the Valencia Tract are guided by a 2001 Non-Industrial Timber Management Plan (NTMP) and a 2013 NTMP amendment (Cal Poly and Big Creek 2001, 2013). Uneven-aged forest management was implemented utilizing a modified BDq approach (Guldin 1991; Piirto et al. 1996, 2007, 2009, 2012).

The application of ecosystem management principles requires understanding of past and present conditions as desired future conditions are defined and adaptive forest management occurs (Manley et al. 1995, Piirto and Rogers 2002). Given these considerations, how effective is the current growth and yield model at estimating volume and predicting future growth at the Valencia Tract? The objectives of this observational sustainability study were to:

1. Standardize the formats of the 2001 pre-harvest, 2002 post-harvest and 2012 forest inventory data sets and conduct individual tree record quality control to validate consistency and growth of measurements.
2. Upload the data into Forest and Stand Evaluation Environment (FORSEE) databases for inventory analysis and comparison utilizing various available or project developed volume equations.
3. Grow the post-harvest 2002 dataset to 2012 and compare the results to the actual 2012 CFI inventory measurement.
4. Identify a suitable volume estimation method for future growth and yield sustainability analysis.

Methods

Project Area

The area delineated for this study was the 239 ha (591 ac) Valencia Tract of Swanton Pacific Ranch. Two management units were established largely due to topographic differences and logging systems planning needs. Inventory data acquisition was completed in 2001, 2002, and 2012. Five forest vegetation types and non-commercial landslide areas were identified within the Tract (table 1) (Cal Poly and Big Creek 2001, 2013) accounting for 228 manageable ha (563 manageable ac) out of the total 239 ha (591 ac); specific areas were excluded from management accounting for the difference.

Table 1—Valencia Tract – units, vegetation types, acres and CFI plots

Unit	Vegetation type	Total acres	Acres in analysis	Number of lots
1	Redwood	210	210	39
1	Douglas-fir	3	3	1
1	Hardwood	26	26	1
1	Brush	19	0	0
1	Landslide	1	0	0
Unit 1 subtotal	**Subtotal**	**259**	**239**	**41**
2	Redwood	259	259	32
2	Douglas-fir	0	0	0
2	Hardwood	42	42	6
2	Brush	23	23	3
2	Landslide	8	0	0
Unit 2 subtotal	**Subtotal**	**332**	**324**	**41**
Grand total		**591**	**563**	**82**

A total of 82 CFI inventory plots were measured throughout the 228 ha (563 ac) project area. The project area, vegetation types, and inventory plot locations are illustrated in fig. 1.

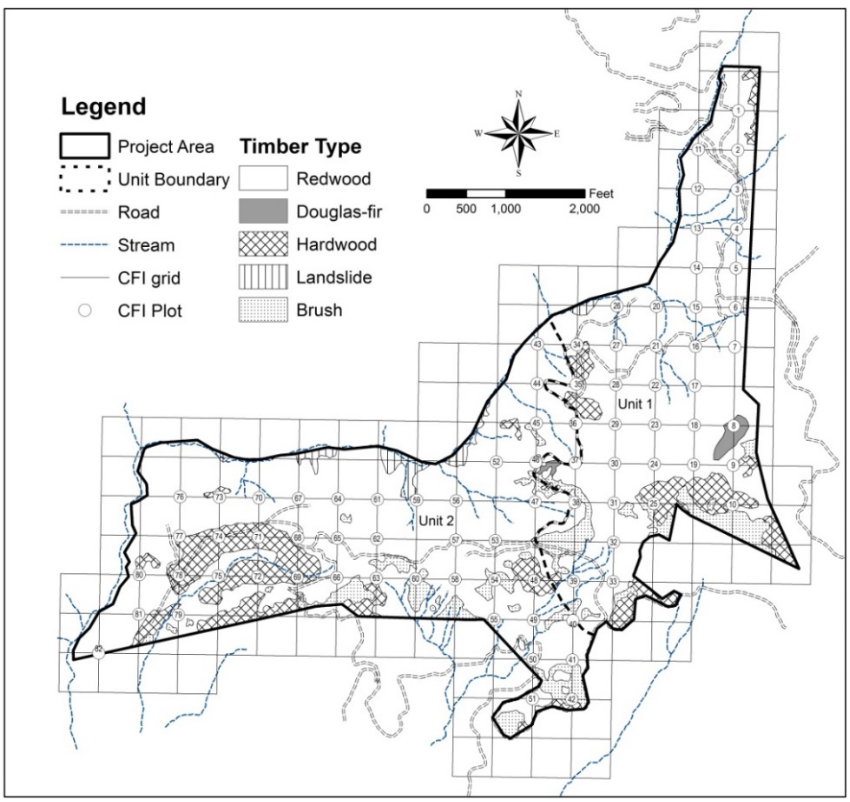

Figure 1—Swanton Pacific Ranch Valencia Tract vegetation types and CFI plots.

Continuous Forest Inventory Design, Measurement, Data Formats and Quality Control

The forest resources of the Valencia Tract were first inventoried in 2001 to support development of the Valencia NTMP (Cal Poly and Big Creek 2001). A 152.4 m (500 ft) uniform grid system was established in relation to a baseline and benchmark at the intersection of Bean Hill Road/Fern Flat Road/and Rusk Grade. Eighty-two circular 0.08 ha (0.2 ac) plots, radius 16.1 m (52.7 ft), were installed in 2001 using a systematic approach at the intersecting points of the grid to develop a

Continuous Forest Inventory (CFI) system. A few sample locations throughout the systematic grid were not measured due to accessibility constraints and other factors. The CFI plots were re-measured in 2002 and 2012. A timber harvest occurred in the project area in 2002, and a post-harvest assessment was conducted at the CFI sample locations to record which trees had been removed and/or damaged because of the harvest. A complete CFI re-measurement was conducted in 2012, 10 years after harvesting.

All inventory data were collected on paper field forms then converted into one spreadsheet template with consistent species designations, data fields, notations, and formats. Initial data set validation was conducted by Cal Poly forest managers (Steve Auten, 2012, personal communication). Inventory data inconsistencies were verified and corrected with original field notes where possible, and 2012 data inconsistencies were field verified in 2014 (Ben Han, 2014, personal communication). Additional data validation and comparison of data set measurements were conducted by Cal Poly forest managers and Environmental Resource Solutions, Inc. during data review and upload. Data field names required minor modifications to be compatible with FORSEE's naming conventions. Tree, stand, and site data went through an integrated auditing process during import into Microsoft Access and uploaded to the FORSEE software. There were some unresolved data inconsistencies.

Use and Development of Volume Estimators

Four different conifer volume estimation sources that included nine different conifer volume equations and one hardwood source that included numerous hardwood species volume equations were used for inventory compilation and volume comparison (table 2). This effort was primarily focused on analyzing various redwood volume equations to determine the differences and the perceived most accurate estimator. The various tree volume equation sources utilized were:

1. Wensel and Krumland (1983): Source 1 is the Bulletin 1907 volume equation and Bulletin 1907 taper equation as this source contains both equation types (Scribner rule) for coastal conifers. The volume equation coefficients utilized for this study are contained in the Bulletin 1907 Appendix table 1A, by species, total height to a 6-inch top diameter inside bark (dib). The Bulletin 1907 volume estimations are postulated by California mensurationists to overestimate tree volumes in the range of 10 to15 percent due to log scaling rounding protocols (Dr. Bruce Krumland, personal communication). The Bulletin 1907 taper equation coefficients utilized are contained in table 8 of that publication, by species to a 0-inch top.
2. Lennette A. and M. Lennette (1997): Source 2 is the Lennette local volume equations. These equations were developed from a sample of 70 sampled redwoods from the Valencia Tract. During recent timber harvests, Cal Poly forest managers found that these equations tend to underestimate volume by approximately 20 percent.
3. Lindquist and Palley (1963): Source 3 is the Bulletin 796 Spaulding table. Historically, Cal Poly forest managers have used this volume table to represent the redwood board foot volume Spaulding Rule to an 8-inch top inside bark. The Bulletin 796 table depicts the results of a weighted multiple regression equation; however, the equation form is not consistent with the volume model forms utilized in FORSEE. Dr. Bruce Krumland[9] converted the Bulletin 796 Table 28 Total Height Volume Table for Young Growth Redwood board foot for an 8-inch top Spaulding rule to a Wensel-Olsen model form[10] for integration with FORSEE. Dr. Krumland utilized the Statsoft Corp. Statistica, version 7 software package to develop a weighted non-linear regression model. The weighting data included 689 redwood trees from the Valencia Tract, ranging between 30.5 to 139.7 cm (12 to 55 inches) DBH that did not have visible indicators of height defects. The resulting R^2 for the model was 0.99998 and the regression standard error was 9.2 board feet.

[9] Mensurationist, author and programmer of the CRYPTOS Growth and Yield Model, and developer of the FORSEE software.
[10] The Wensel/Olsen model (1995): V=a*(DBH^b)*(HT^c)*(d^DBH).

4. Han, Ben[11]: Source 4 refers to two equations developed by Ben Han. These equations were based on fall and buck data from 104 trees collected from the Valencia Tract.
5. Pillsbury and Kirkley (1984): Source 5 is the Pillsbury PNW-414 hardwood equations. Species specific wood volume equations from table 3 in the PNW-414 publication were used for all hardwoods in the inventory.

Table 2—Volume equation sources used for each dataset compilation[a]

Report name	Redwood volume source	Douglas-fir volume source	Hardwood volume source	Conifer top dib
Bulletin 1907 Equation	1 volume equation	1 volume equation	5	6
Bulletin 1907 Taper	1 taper equation	1 taper equation	5	6
Lennette Local Equation	2	2	5	6
Bulletin 796 Spaulding	3	1 volume equation	5	8
Han Masters 6	4	1 volume equation	5	6
Han Masters 7	4	1 volume equation	5	6

[a]Volume equations available upon request.

Data Compilation and Analysis

Inventory data were compiled and analyzed using the **FOR**est and **S**tand **E**valuation **E**nvironment (FORSEE) computer program (CAGYM 2012). The CRYPTOS growth model option within FORSEE was used to model forest growth, as this model is appropriate for the coast redwood forest type (Krumland and Eng 2005, Wensel et al. 1987, Wensel and Olsen 1995). All measured site trees within the project area were utilized to establish the initial site index value for the project area. The FORSEE calculated coast redwood site index (base age 50) was 89, which is considered an acceptable estimate for this area. This average site index estimate falls into a Redwood Site Class III.

The initial 2001 inventory data was compiled to provide a baseline assessment of forest stocking. To simulate the 2002 harvest, 2001 data was grown 1 year before harvested trees were removed from the inventory, identified as the Status Code 2 field in the database, indicating the tree had been removed as a result of harvest. Harvested trees were then removed from the tree database and the inventory was recompiled and grown an additional 10 years to reflect modeled conditions as of 2012. The grown stand was saved and recompiled before reporting the modeled 2012 data statistics. The CFI re-measurement in 2012 represents the most recent inventory of forest conditions for the tract.

FORSEE compilation assumptions were:

1. No ingrowth was added to the grown 2002 data.
2. Stand BR_2 did not exist in the 2001 data, but did exist in the 2012 data for plots 42, 54, and 60. These plots were moved to the BR_2 stand in the 2001 data.
3. No site trees for stand BR_1 were taken during the inventory process. To compensate for this, site trees from stand HW_1 were duplicated. The HW_1 stand was picked because both stands exhibited a similar under-performing conifer component.
4. All volume models were set up for a 6-inch top dib, except the coast redwood Spaulding equation that was run to an 8-inch top dib due to the basis of the original volume table.
5. Only trees 12 inch DBH and greater were compiled for volume.

All inventory data sets were compiled to report forest metrics and statistics at the unit vegetation type level.

[11] Han, W.B. Comparing volume equations for young-growth redwood in Santa Cruz County. Master's thesis in progress. California Polytechnic State University, San Luis Obispo, CA.

Results

The summarized information for the 2001 pre-harvest data, 2002 post-harvest data, 2002 post-harvest data grown to 2012, and 2012 data CFI inventories are shown in tables 3 to 6, and figs. 2 through 12. The data summarizes the FORSEE modeled and measured changes of 82 CFI inventory plots over 563 timbered acres of the Valencia Tract. There may be minor summation issues due to programmatic and table display rounding. Reported values are for the total project area within the Valencia Tract for all vegetation types and units included in the analysis.

The data shows an uneven-aged forest with a reverse J-shaped diameter distribution. The current 2012 actual stand data (table 6, figs. 7 and 8) indicates the forest stocking contains approximately 304 trees per acre (TPA) of all species, has a total quadratic mean diameter (QMD) of 13.34 inches, a total basal area per acre of 295 ft^2, a conifer basal area per acre of 207 ft^2, and a conifer gross volume (Spaulding Rule) per acre of 39,367 board feet. The 2012 CFI- re-measurement data was evaluated using coast redwood gross volume equations to produce per acre estimates ranging from 29,427 board feet (Source 2) to 38,073 board feet (Source 1), which represented a 29 percent swing in stand volume. The ability of Cal Poly Forest Managers to predict growth and yield for the Swanton Pacific School Forest and the associated Valencia Tract depends on identifying a suitable volume estimation source.

The results from each dataset (2001 pre-harvest, 2002 post-harvest, 2002 grown to 2012, and 2012 actual stand re-measurement) and volume source (table 1) compilation are presented in tables 3 through 6. Stand forest metrics (TPA, QMD, BA/Ac, CO2e/Ac) are reported by species.

Table 3—Swanton Pacific Ranch Valencia Tract – FORSEE 2001 pre-harvest yield summary

Forest stocking metrics				
Species group	**TPA**	**QMD**	**BA/ac**	**CO2e/ac**
All Conifers	147.23	16.1	208.09	343.93
Redwood	131.92	16.43	194.15	320.28
Douglas-fir	15.31	12.92	13.94	23.47
Hardwoods	176.87	9.56	88.25	140.25
Totals	**324.1**	**12.95**	**296.35**	**484.37**

Gross volume per acre estimation source						
Species group	**Bulletin 1907 equation**	**Bulletin 1907 taper**	**Lennette local equation**	**Bulletin 796 Spaulding**	**Han Masters 6**	**Han Masters 7**
All conifers	38,330	36,697	30,713	36,065	33,026	33,514
Redwood	35,188	33,576	27,836	32,858	29,884	30,372
Douglas-fir	3,142	3,121	2,877	3,207	3,142	3,142
Hardwoods	2,358	2,358	2,358	2,358	2,358	2,358
Totals	**40,688**	**39,055**	**33,071**	**38,423**	**35,384**	**35,872**

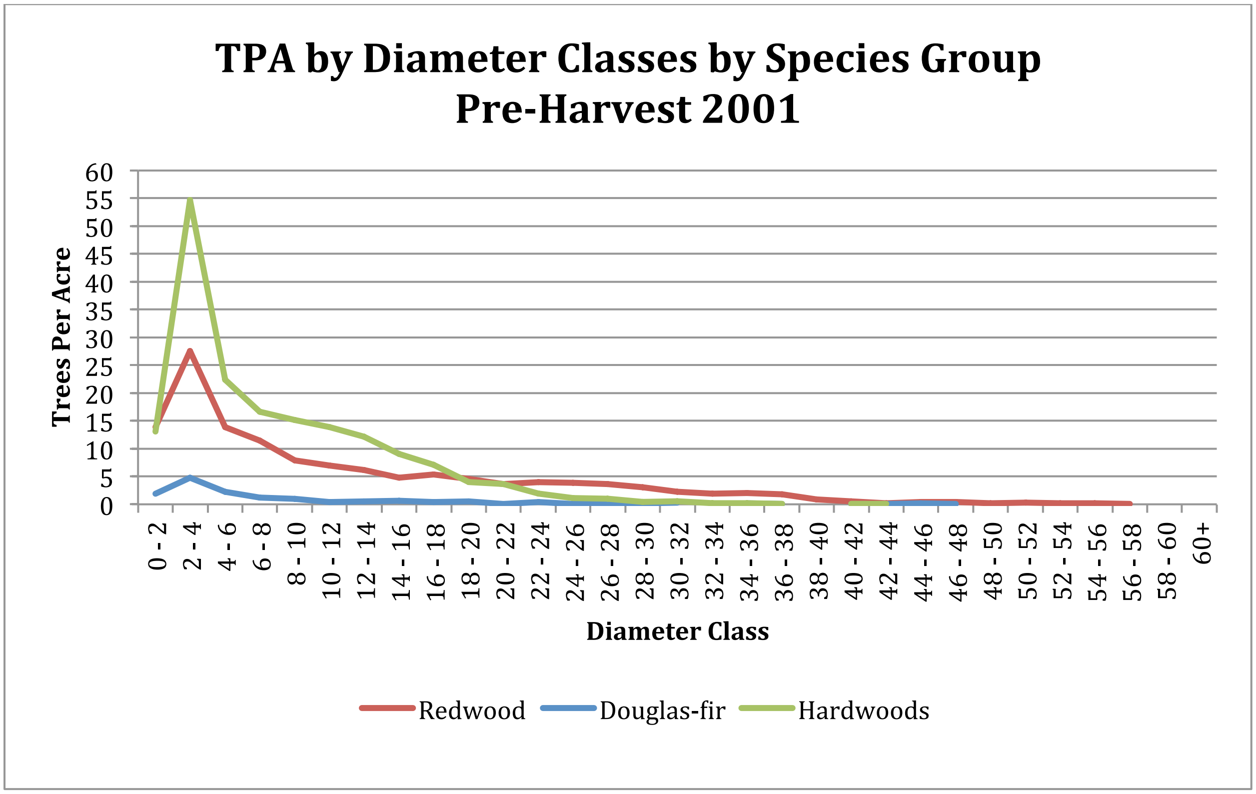

Figure 2—Swanton Pacific Ranch Valencia Tract – FORSEE 2001 pre-harvest summary. Trees per acre by diameter class and species group.

Table 4—Swanton Pacific Ranch Valencia Tract – FORSEE 2002 post-harvest yield summary

Forest stocking metrics				
Species group	**TPA**	**QMD**	**BA/ac**	**CO2e/ac**
All conifers	120.16	15.73	162.25	264.92
Redwood	108.22	15.90	149.17	242.73
Douglas-fir	11.94	14.17	13.07	22.37
Hardwoods	138.15	10.26	79.38	126.87
Totals	**258.31**	**13.10**	**241.63**	**391.97**

Gross Volume per Acre Estimation Source						
Species group	**Bulletin 1907 equation**	**Bulletin 1907 taper**	**Lennette local equation**	**Bulletin 796 Spaulding**	**Han Masters 6**	**Han Masters 7**
All conifers	29,053	27,893	22,890	27,343	25,093	25,515
Redwood	26,033	24,907	20,187	24,285	22,073	22,494
Douglas-fir	3,021	2,986	2,704	3,058	3,021	3,021
Hardwoods	2,135	2,135	2,135	2,135	2,135	2,135
Totals	**31,189**	**30,028**	**25,026**	**29,478**	**27,229**	**27,650**

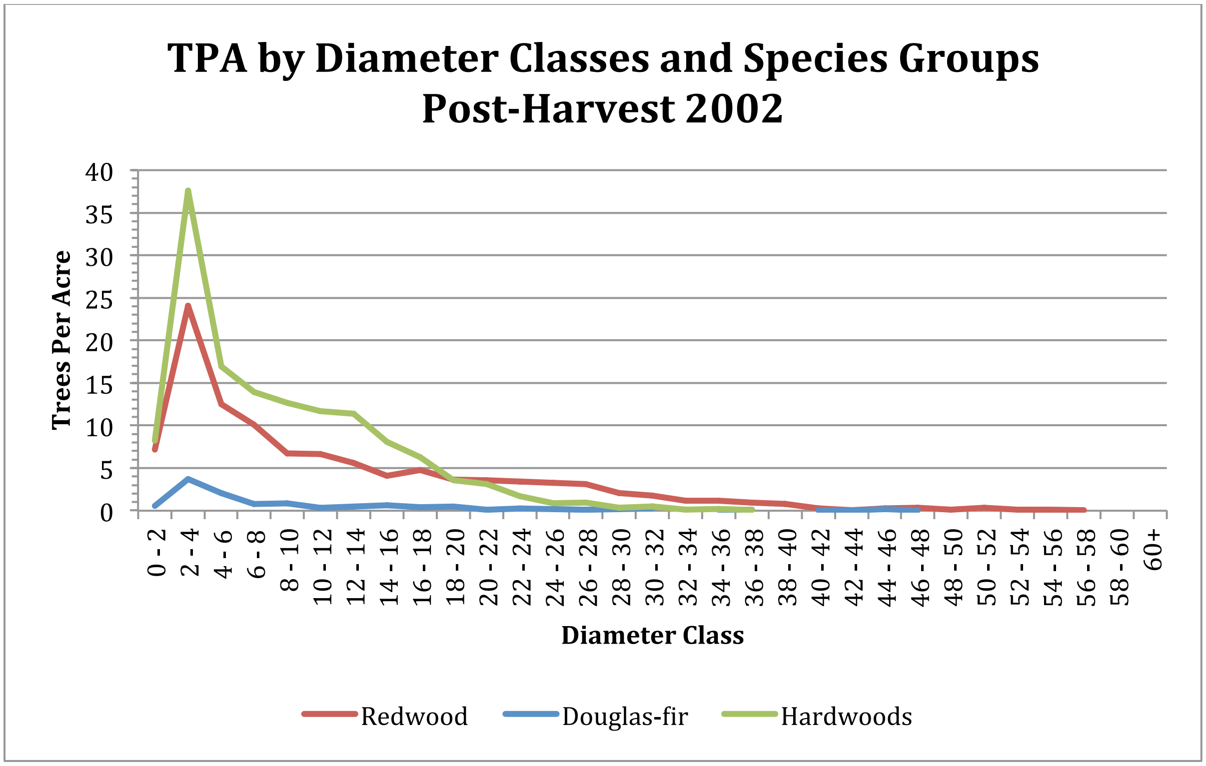

Figure 3—Swanton Pacific Ranch Valencia Tract – FORSEE 2002 post-harvest summary. Trees per acre by diameter class and species group.

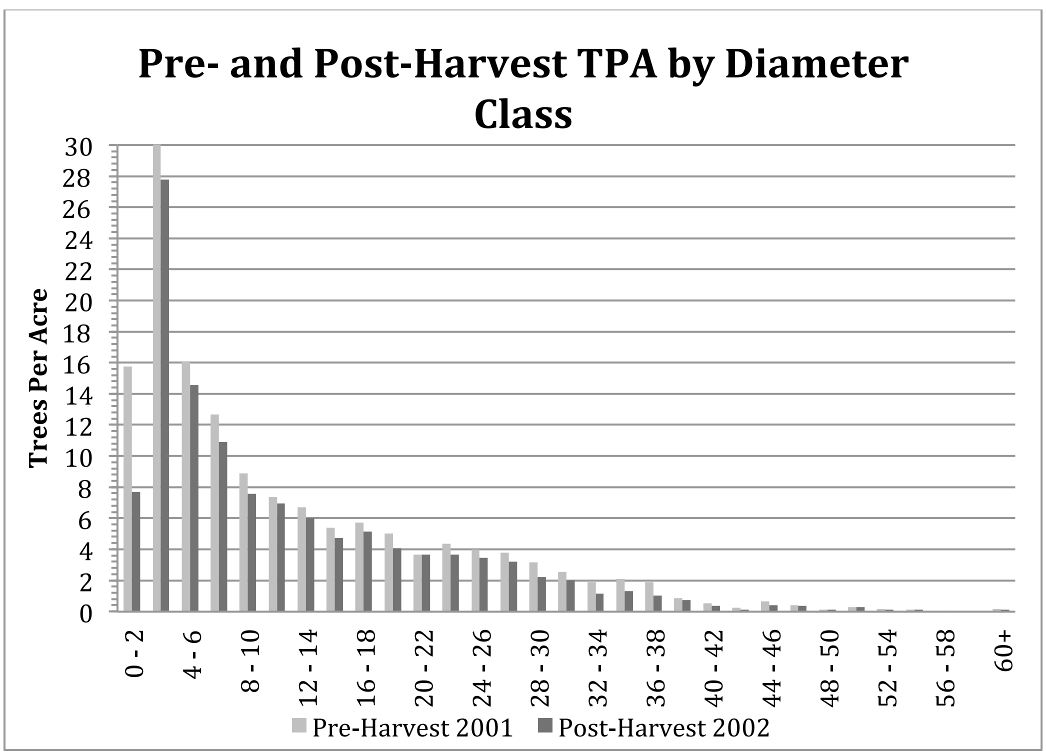

Figure 4—Swanton Pacific Ranch Valencia Tract – FORSEE 2002 post-harvest actual stand data summary. Comparison of 2001 pre-harvest and 2002 post-harvest diameter distribution.

The Valencia Tract was harvested in 2001 and 2002. Actual log scale records indicate a harvest volume of 4,106,000 for the entire Valencia Tract (467 harvested acres). The project data set indicate a harvest volume of 4,165,000 (Taper equation) and 4,330,000 (Spaulding) which is within 1.4 percent and 7.9 percent respectively of the actual volume harvested.

Table 5—Swanton Pacific Ranch Valencia Tract – FORSEE 2002 grown to 2012 yield summary

Forest stocking metrics				
Species group	**TPA**	**QMD**	**BA/ac**	**CO2e/ac**
All conifers	121.06	16.97	190.11	312.03
Redwood	108.97	17.10	173.75	283.62
Douglas-fir	12.09	15.75	16.37	28.23
Hardwoods	138.55	11.08	92.77	151.25
Totals	**259.61**	**14.13**	**282.88**	**463.10**

Gross volume per acre estimation source						
Species group	**Bulletin 1907 equation**	**Bulletin 1907 taper**	**Lennette local equation**	**Bulletin 796 Spaulding**	**Han Masters 6**	**Han Masters 7**
All conifers	34,913	33,402	26,495	32,712	30,119	30,588
Redwood	31,034	29,658	23,301	28,817	26,240	26,710
Douglas-fir	3,878	3,744	3,194	3,895	3,878	3,878
Hardwoods	2,538	2,538	2,538	2,538	2,538	2,538
Totals	**37,451**	**35,940**	**29,033**	**35,251**	**32,657**	**33,127**

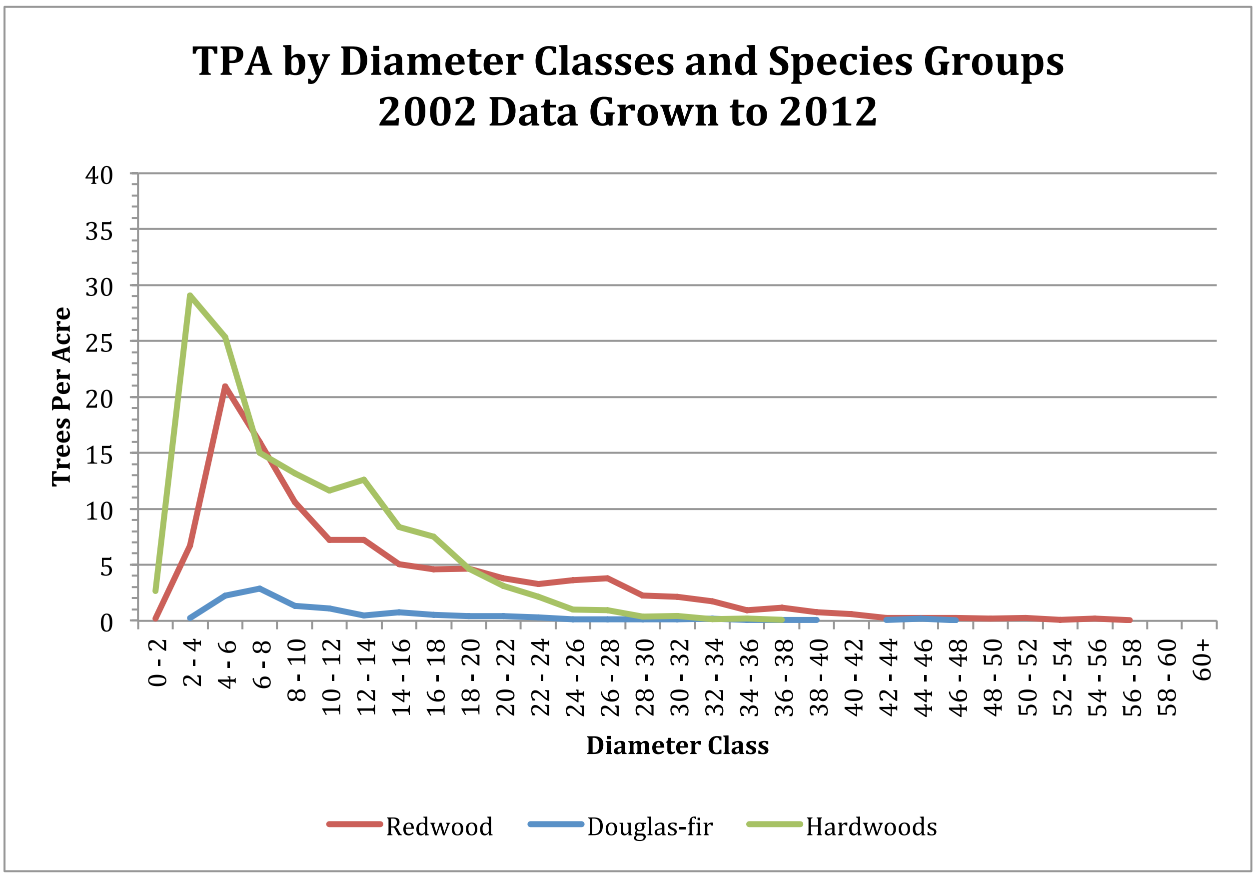

Figure 5—Swanton Pacific Ranch Valencia Tract – FORSEE 2002 grown to 2012. Post-harvest summary. Trees per acre by diameter class and species group.

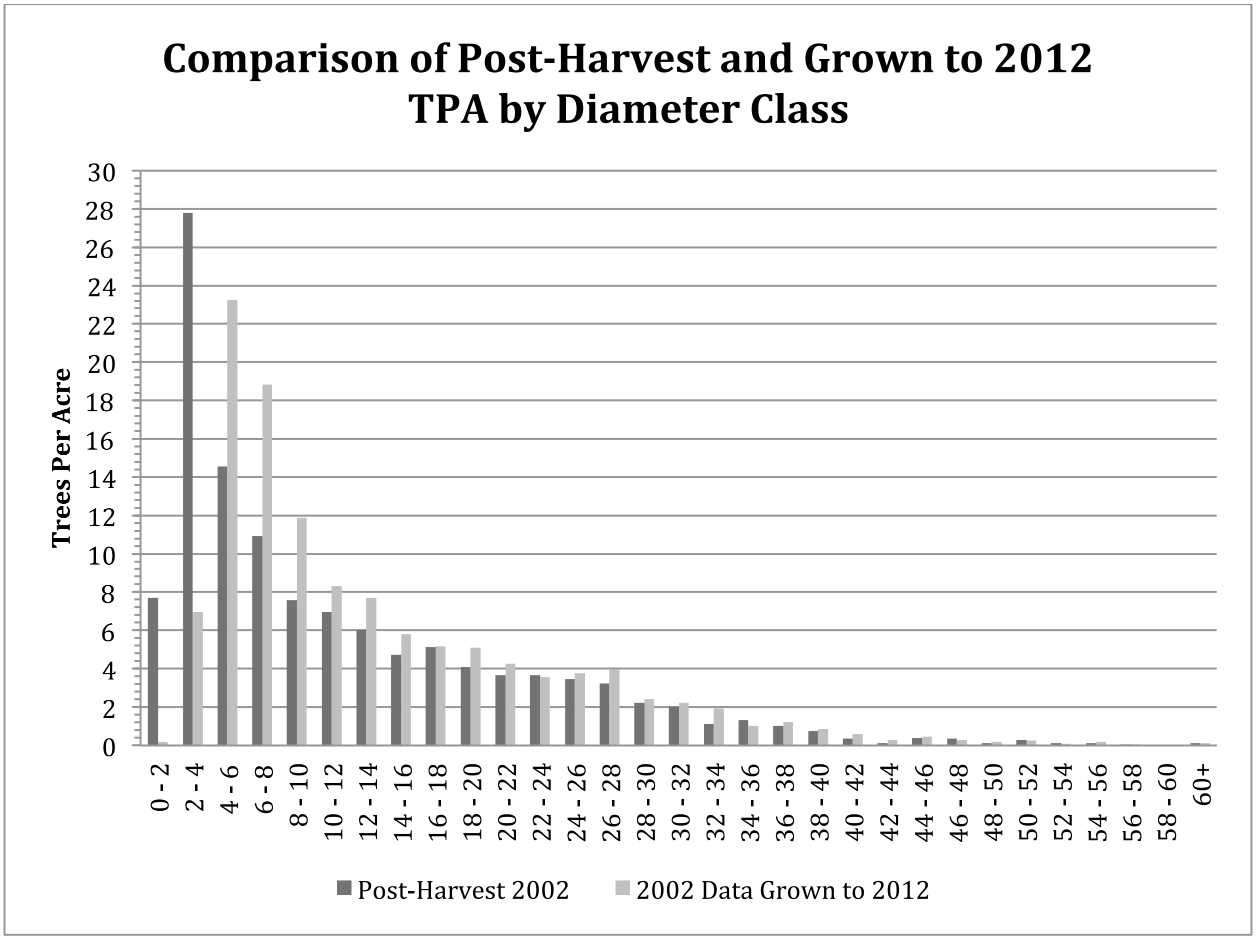

Figure 6—Swanton Pacific Ranch Valencia Tract – FORSEE 2002 grown to 2012. Post-harvest summary comparison of 2002 post-harvest and 2002 grown to 2012 diameter distribution.

Table 6—Swanton Pacific Ranch Valencia Tract – FORSEE 2012 actual stand yield summary

Forest stocking metrics				
Species group	**TPA**	**QMD**	**BA/ac**	**CO2e/ac**
All conifers	155.11	15.64	206.88	354.93
Redwood	142.99	15.64	190.73	326.15
Douglas-fir	12.12	15.63	16.16	28.78
Hardwoods	148.6	10.42	88.04	150.52
Totals	**303.71**	**13.34**	**294.92**	**505.27**

Gross volume per acre estimation source						
Species group	**Bulletin 1907 equation**	**Bulletin 1907 taper**	**Lennette local equation**	**Bulletin 796 Spaulding**	**Han Masters 6**	**Han Masters 7**
All conifers	42,227	39,784	32,994	39,367	36,361	36,527
Redwood	38,073	35,739	29,427	35,191	32,207	32,374
Douglas-fir	4,154	4,045	3,567	4,176	4,154	4,154
Hardwoods	2,773	2,773	2,773	2,773	2,773	2,773
Totals	**45,000**	**42,557**	**35,767**	**42,140**	**39,134**	**39,301**

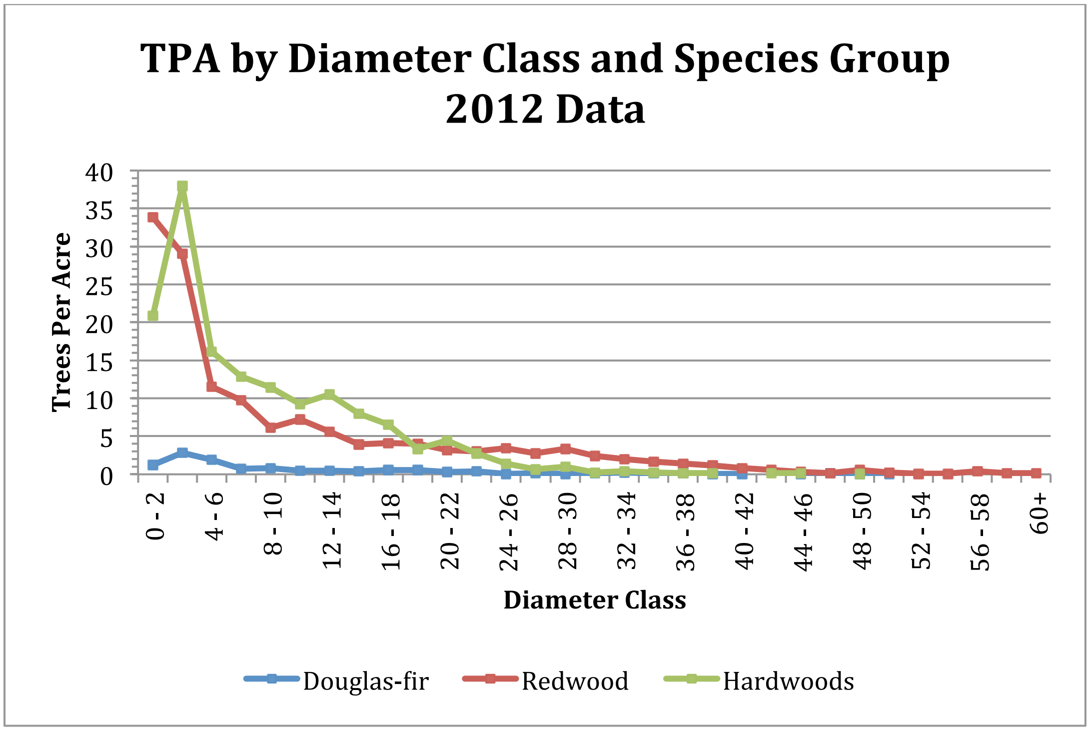

Figure 7—Swanton Pacific Ranch Valencia Tract – FORSEE 2012 actual stand summary. Trees per acre by diameter class and species group.

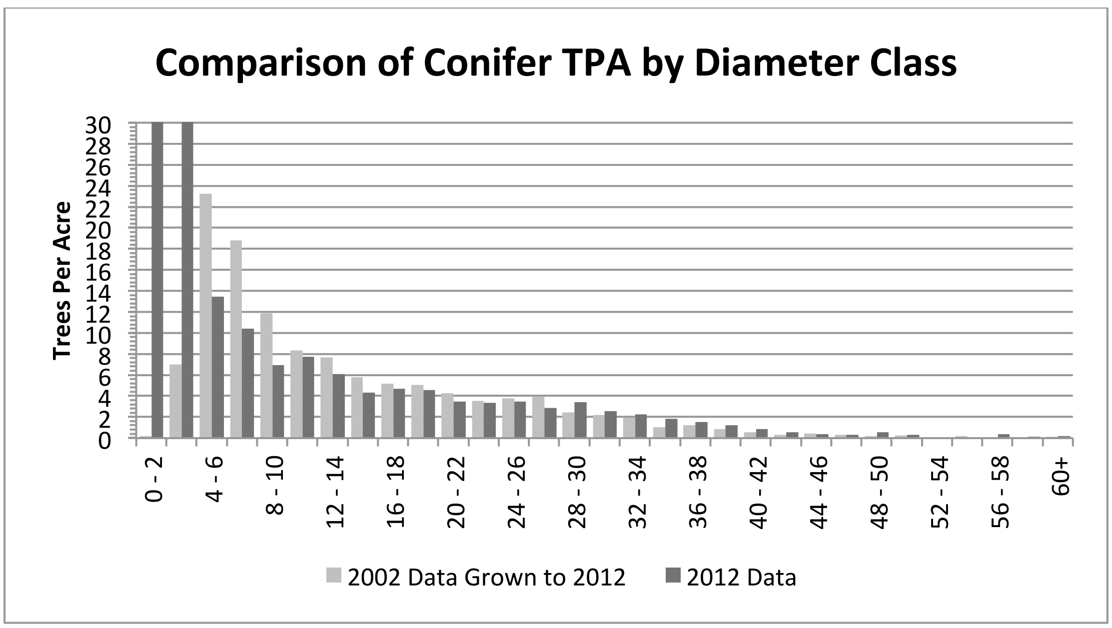

Figure 8—Swanton Pacific Ranch Valencia Tract – FORSEE 2012 CFI Summary. Comparison of 2002 grown to 2012 and 2012 CFI data.

The comparison of the volume equations (based on the 2012 CFI actual stand dataset) are listed from highest to lowest computed redwood volume in table 7.

Table 7—Redwood volume computation ranked from highest to lowest volume per acre based on 2012 CFI dataset

Equation name	Source	Redwood gross volume per acre
Bulletin 1907 equation	1	38,073
Bulletin 1907 taper	1	35,739
Spaulding rule	3	35,191
Masters 7	4	32,374
Masters 6	4	32,207
Lennette local	2	29,427

As previously indicated, the Bulletin 1907 equation is postulated to over-estimate volume by 10 to 15 percent. Likewise, Cal Poly forest managers have experienced that the Lennette Local equation under-estimates volumes by up to 20 percent. Therefore, it seems prudent to identify a reliable volume estimation source between these two extremes. Many California mensurationists are choosing the Bulletin 1907 Taper equation as the best publicly available source. The site specific Han fell and buck volume equations (see footnote 11) is 9 percent less than the Bulletin 1907 Taper estimates, and indicates lower redwood volumes per acre. This project has converted the Lindquist and Palley Bulletin 796 Spaulding Rule table to a form compatible with FORSEE and provides a slightly more conservative estimate of volume due to the 8-inch top diameter, a manufacturing specification that has been common for California mills purchasing coast redwood sawlogs.

The 11-year interval measurement (2001 pre-harvest data and 2012 measured data) indicates that the Valencia Tract was harvested in 2002 at a per acre conifer volume rate of approximately 24 percent (8,712 board feet per acre). By 2012, the Spaulding equation indicates the stand volume growth had completely recovered after harvest and exceeded the pre-harvest 2001 gross volume condition by 9.7 percent (average stand growth of 1,266 board feet per acre per year, or 3.0 percent growth rate per year). The robust growth rate may be attributable, in part, to maintaining large trees in the stand structure.

FORSEE estimates of overall volume grown to 2012 were under-estimated by 16.3 percent compared to 2012 actual measured data (fig. 9). It is postulated that this difference in FORSEE model projection is either a result of the way the CFI data was processed in FORSEE, regional differences, or inherent projection inaccuracies not fully understood by FORSEE users.

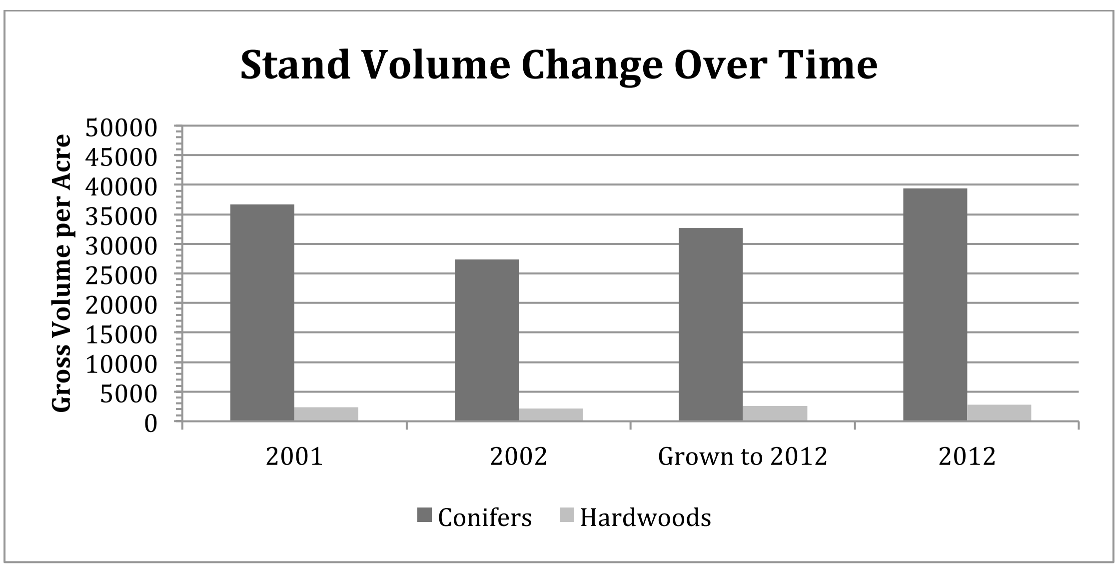

Figure 9—Stand board foot volume change over time.

The measured increase in conifer basal area from 2002 to 2012 was 44.63 square feet per acre, whereas the FORSEE model predicted an increase of 27.86, which represents an 8 percent underestimation in conifer basal area growth based on how the CFI data was processed in the FORSEE model, primarily attributed to not including ingrowth. The 2012 actual stand data indicates the stand has regrown all basal area in the last decade (2001 conifer basal area was 208 per acre; 2012 conifer basal area is 207 per acre).

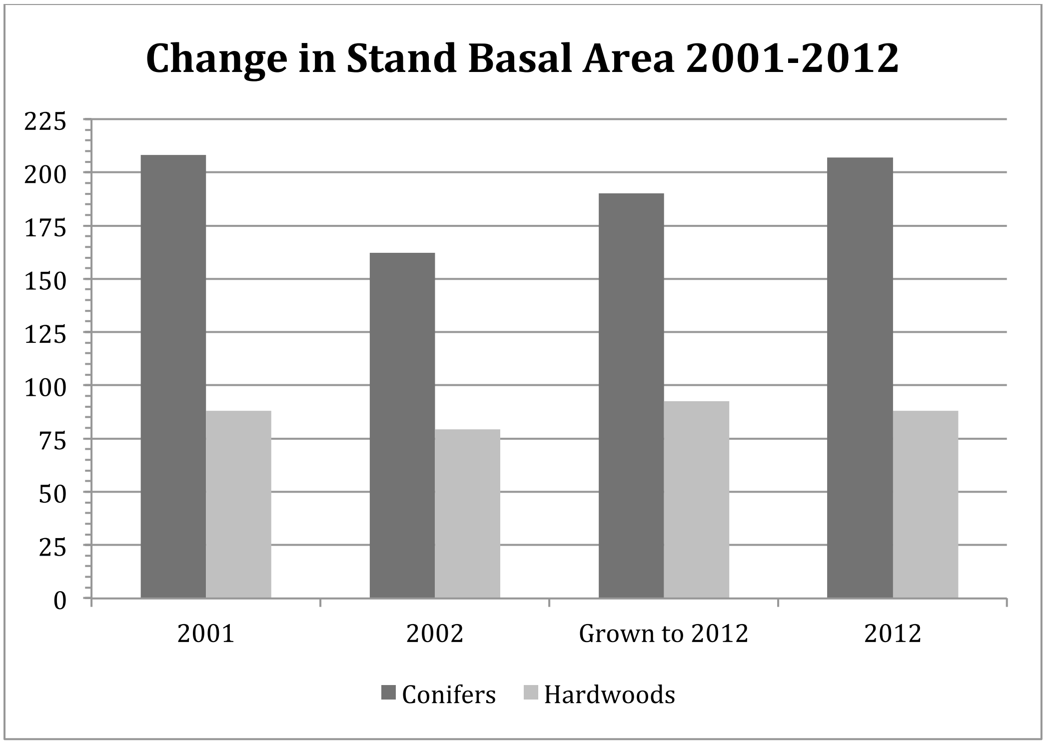

Figure 10—Change in stand basal area 2001-2012.

The FORSEE model predicted a 7.9 percent increase in conifer quadratic mean diameter (QMD); however, there was actually a 0.9 percent decrease per the 2012 data measurement. This is consistent with expectations when not including ingrowth to the FORSEE model.

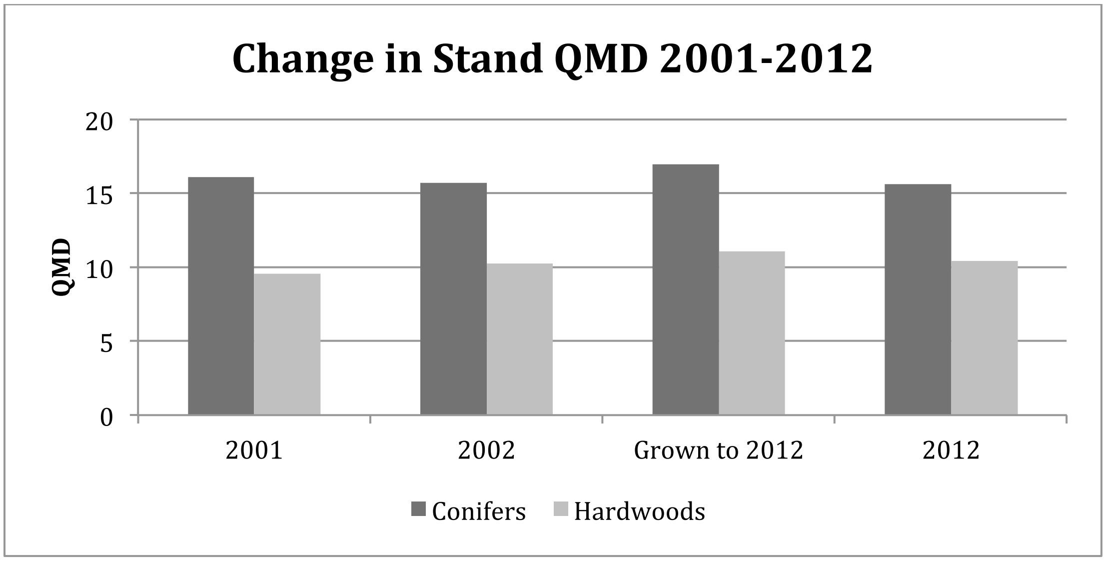

Figure 11—Change in stand QMD 2001-2012.

The FORSEE model predicted very little change in numbers of TPA. This was expected because no ingrowth was added to this growth regime. In comparison to the 2012 data, we may expect more TPA due to sprouting and new seedlings. The 2012 data indicates approximately 34 additional conifer TPA, generally consistent with expectations.

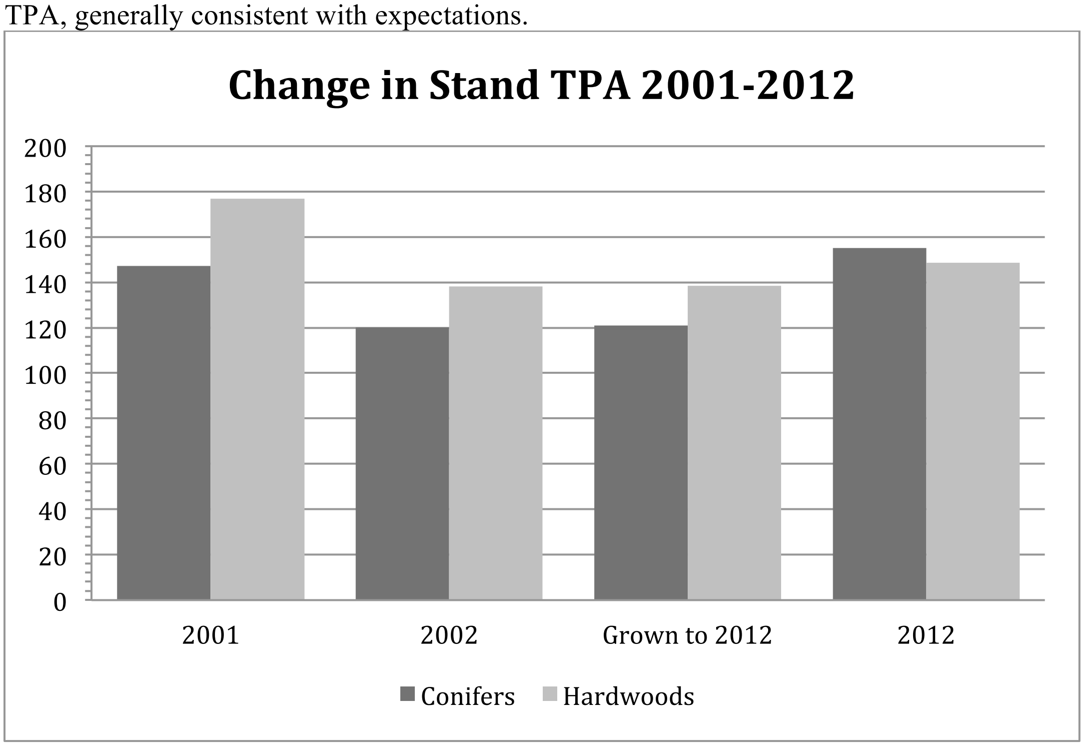

Figure 12—Change in stand TPA 2001-2012.

A detailed examination of the tree growth data revealed that the FORSEE growth model underestimated diameter growth on trees > 33 cm (13 inches) DBH, and underestimated height growth on trees > 53.3 cm (21 inches) DBH; FORSEE growth estimates below these DBH and height values were moderately overestimated. Underestimating growth on the larger trees would underestimate volume, which is consistent with the reported results; overestimating growth on smaller trees would likely have a small impact on volume in the short term, especially considering that trees less than 30.5 cm (12 inches) were not included in the volume reporting totals.

Discussion

We have completed the project objectives, learned lessons, and gained valuable insight into processes that are used to guide future forest management. In developing this comparative study, Cal Poly forest managers needed to review and provide quality control for continuity of inventory data, and while this provided some difficulty, this process will help guide future data collection quality control efforts and record keeping.

Second, Cal Poly forest managers had previously utilized a static local volume table for redwood that was based on Bulletin 796. This project allowed the development of a standardized volume equation utilizing the board foot Spaulding Rule. This fitted equation is compatible with FORSEE and can also be used in other spreadsheet programs that will allow reliable volume estimations for growth and yield updates, timber sale harvest volumes, etc.

Third, the accuracy of the FORSEE growth model for use in the southern sub-district was compared to measured data at a 10-year interval. While growth rates can be adjusted in FORSEE, and the calculated stands site index is a factor in FORSEE growth, the results from the default model settings indicate that for the Valencia Tract, the FORSEE growth model underestimated basal area per acre growth, overestimated the change in QMD, overestimated the growth potential of smaller trees < 30.5 cm (12 inches) DBH, and underestimated the growth potential of larger trees > 33 cm (13 inches) DBH within this forest. This study validates what many forest managers in California have speculated, that updated, regionally specific, growth models are necessary to provide accurate estimates for forest managers to conduct responsible forest planning and for public agencies charged with enforcing long-term sustained yield plans.

Lastly, Cal Poly managers have evaluated six different redwood volume estimation methods and shown that the medial published source (Spaulding Rule) appears reliable for growth and yield reporting for its Santa Cruz County forest tracts. The use of Spaulding Rule volumes is consistent with Cal Poly's historic reliance on this source and its perceived accuracy during previous timber harvest cutouts. Having this equation for direct inventory compilation in FORSEE will be a valuable asset during future management planning and harvesting projects.

Conclusion and Management Implications

The Valencia Tract FORSEE growth results for the Spaulding model (gross volume per acre) were approximately 16.3 percent low for the 2002 to 2012 10-year measurement interval. Based on these results, it is recommended that either a growth adjustment be applied, adjustments to site index, or development of updated, regionally specific growth models be undertaken. Growth adjustments can be difficult to calibrate and defend; updated growth models would aid in the accuracy of estimating and reporting volume growth trends for long-term sustained yield plans and other biomass estimation projects.

Successful use of CFI data and FORSEE growth and yield analysis is dependent upon several factors in addition to landowners' objectives. The collected data is most useful when identified and formatted correctly for use in the analysis. Identification of data needs to include property name, unit and/or stand designator, month and year of data collection. Data format should be consistent with inventory compiler input fields and utilize consistent and repeatable identification codes for status, damage, defect, and other attributes. If working on large properties where data collection effort spans more than one season, a geographic information system is useful to maintain historic records and effectively plan inventory effort to maximize forest management options.

Summarizing, it was determined in this Valencia Tract coast redwood sustainability analysis that: 1) consistency and validation of CFI data is paramount for reliable comparisons; 2) this analysis underestimated basal area per acre growth, overestimated change in QMD), overestimated diameter growth of smaller trees (< 30.5 cm (12 inches) DBH), underestimated diameter growth of larger trees (> 33 cm (13 inches) DBH), underestimated height growth of larger trees (> 53.3 cm (21 inches) DBH), underestimated volume (16.3 percent lower than actual CFI volume figures); 3) by 2012 stand volume growth had completely recovered and exceeded the pre-harvest 2001 gross volume by 9.7 percent (i.e., average stand growth of 1,266 board feet per acre per year, or 3.0 percent growth rate per year); 4) the Spaulding rule appears to be a medial choice for Valencia Tract sustainable yield analysis; 5) sustainable stand management resulted from setting residual stand basal area (b), maximum diameter (d) and trees per acre by diameter targets (q) while leaving a few trees greater than the established maximum diameter. FORSEE is a useful inventory, growth, and yield model that can become better with local calibration and proper use by trained professionals.

Acknowledgments

We thank Ms. Janet Webb, the McCrary Family, Big Creek Lumber Company, California Department of Forestry and Fire Protection (Cal Fire), Ms. Nadia Hamey, Dr. Brian Dietterick, and the many individuals, students, organizations, companies, and agencies who have contributed over the years to advancing Cal Poly's Learn by Doing forest management and environmental stewardship work at Cal Poly's Swanton Pacific Ranch. We also thank Environmental Resource Solutions, Inc. (ERS), Ms. Bonnie Burchill, ERS staff, Dr. Bruce Krumland, Dr. John Kliejunas, Dr. Sauli Valkonen, and Ms. Lori Ann Walters for their assistance with this project. The federal McIntire Stennis program provided funding for this observational, management study. This paper memorializes: 1.) Mr. Al Smith who bequeathed Swanton Pacific Ranch to Cal Poly so that Cal Poly could have a place for students, faculty, and staff to Learn by Doing; and 2.) Mr. Dale Holderman, a highly respected Registered Professional Forester in Santa Cruz, California, who provided ideas to the authors that were incorporated into this paper before he passed away in 2016.

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SESSION 2 – Fire Ecology and Effects

Forest Restoration at Redwood National Park: Exploring Prescribed Fire Alternatives to Second-Growth Management: a Case Study[1]

Eamon Engber,[2] Jason Teraoka,[2] and Phil van Mantgem[3]

Abstract

Almost half of Redwood National Park is comprised of second-growth forests characterized by high stand density, deficient redwood composition, and low understory biodiversity. Typical structure of young redwood stands impedes the recovery of old-growth conditions, such as dominance of redwood (*Sequoia sempervirens* (D. Don) Endl.), distinct canopy layers and diverse understory vegetation. Young forests are commonly comprised of dense, even-aged Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and redwood stump sprouts, with simple canopy structure and little understory development. Moreover, many of these young stands are believed to be vulnerable to disturbance in the form of drought, disease and fire.

Silvicultural practices are increasingly being employed by conservation agencies to restore degraded forests throughout the coast redwood range; however, prescribed fire treatments are less common and potentially under-utilized as a restoration tool. We present an early synthesis from three separate management-scale prescribed fire projects at Redwood National Park spanning 1to 7 years post-treatment. Low intensity prescribed fire had minimal effect on overstory structure, with some mortality observed in trees smaller than 30 cm diameter. Moderate to high intensity fire may be required to reduce densities of larger Douglas-fir, the primary competitor of redwood in the Park's second growth forests. Fine woody surface fuels fully recovered by 7 years post-burn, while recruitment of larger surface fuels was quite variable. Managers of coastal redwood ecosystems will benefit by having a variety of tools at their disposal for forest restoration and management.

Introduction

Redwood National and State Parks (RNSP) are best known as having the most spectacular examples of old-growth coastal redwood forests, but approximately 45 percent of RNSP is composed of second-growth forests. Typical structure of young redwood stands impedes the rapid recovery of old-growth conditions, such as dominance of redwood (*Sequoia sempervirens* (D. Don) Endl.) distinct canopy layers and diverse understory vegetation (O'Hara et al. 2010, Teraoka and Keyes 2011). Young forests are commonly comprised of dense, even-aged Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and redwood stump sprouts, with simple canopy structure and little understory development. Under these conditions the relatively shade-intolerant Douglas-fir is expected to exclude redwood from the upper canopy until large gaps are formed, a process that may take centuries (Thornburgh et al. 2000). Moreover, many of these young second growth stands are believed to be vulnerable to disturbance in the form of drought, disease and fire (fig. 1). While the presence of second-growth redwood forest presents a management challenge, it also provides an opportunity for landscape restoration, an important theme in the 1978 Redwood National Park Expansion Act (USDI 1999).

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Figure 1—Dense second growth forest conditions in Redwood National Park.

Figure 2—Fire-scarred redwood cross-section from Redwood National Park.

Wile the understanding of fires role in redwood forest development is somewhat ambiguous, tree ring evidence suggests fires were historically quite frequent resulting from a combination of Native American and lightning ignition sources (Lorimer et al. 2009; fig. 2). Fire history research in Del Norte and northern Humboldt counties has reported historical fire return intervals of about 10 years at a site east of Prairie Creek State Park (Brown and Swetnam 1994), 11 to 26 years in Del Norte Redwoods State Park (Norman 2007), and up to 500 years at humid coastal sites in RNSP (Veirs 1980). While fire surely played a role in the development of many old growth stands, second growth redwood forest structure differs drastically from old growth (e.g., smaller trees, more of them, sprouts surrounding rotten stumps), and limited information is available on the effects of fire in second growth redwood stands.

The RNSP currently uses silviculture (i.e., mechanical thinning) as the primary tool to meet the forest restoration objectives in the park, which include: decrease stand density; increase spatial heterogeneity in the forest canopy; alter tree species composition to give redwood a competitive advantage; increase understory diversity; and increase forest vigor. Silvicultural methods have a long history of success in commercial and non-commercial settings (Bauhus et al. 2009, Teraoka and Keyes 2011), however the use of silviculture in ecological restoration is generally a novel approach to park management and was first used by RNSP Ecologist Stephen Veirs in the early 1980s (but has since increased in scale and scope). Prescribed fire may have utility in the restoration of second growth redwood forests in RNSP, especially in low-accessibility stands. Prescribed fire may also offer unique benefits over mechanical treatments (e.g., fuels reduction, understory vegetation response, snag and cavity creation, nutrient cycling, etc.). In this article we present and discuss results from three prescribed fires that were conducted to help restore second-growth forests in RNSP, dating back to 2006. The three case studies demonstrate both low-intensity and moderate-intensity treatments, and we discuss management implications and long term monitoring.

Methods

We present monitoring results from three prescribed fires located in the Bald Hills area of Redwood National Park (fig. 4). The Dolason-Airstrip (DA) unit (12.1 ha, 30 ac) was treated in the fall of 2007 with a low-moderate intensity burn. Monitoring plots were installed pre-burn and re-measured 1 year post-burn and 7 years post-burn. A total of 21 plots were installed and overstory trees ≥ 20cm (7.9

inches) diameter at breast height (DBH; 1.37 m) were assessed on 1/25th ha plots while pole trees < 20cm DBH were assessed on 1/100 ha plots. One Browns Planar Intercept fuels transect was installed at each plot (Brown 1974). The Dolason Trailhead (DTH) unit (3.2 ha, 8 ac) was treated in the fall of 2006 with a moderate- high intensity burn. Eight monitoring plots were installed immediately post-burn (no true "pre" data were recorded) and assumptions were made concerning the pre-burn live/dead status of trees; plots were re-assessed 7 years post-burn. One Browns fuels transect was installed at each plot. Similar plot sizes were utilized for overstory and pole trees in the Dolason-Airstrip and Dolason Trailhead burn units. The Lower Airstrip Expansoin (LAE) unit (20.2 ha, 50 ac) was treated with low-intensity fire in the fall of 2013. Nine small plots (1/50th ha) were utilized for pole trees < 20 cm (7.9 in) DBH and fuel loading (one transect per plot), while two, 1 ha large plots were installed to capture treatment effects on overstory trees ≥ 20cm DBH. Plots were assessed pre-burn and 1 year post-burn. Both LAE and DA had a mixed composition of Douglas-fir, tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh), and redwood, while the Dolason Trailhead unit lacked redwood, but was dominated by Douglas-fir and tanoak. All stands within these units were 40 to 45 years of age at time of treatment. All units had a combination of crown volume scorch and/or bark char height assessed post-burn.

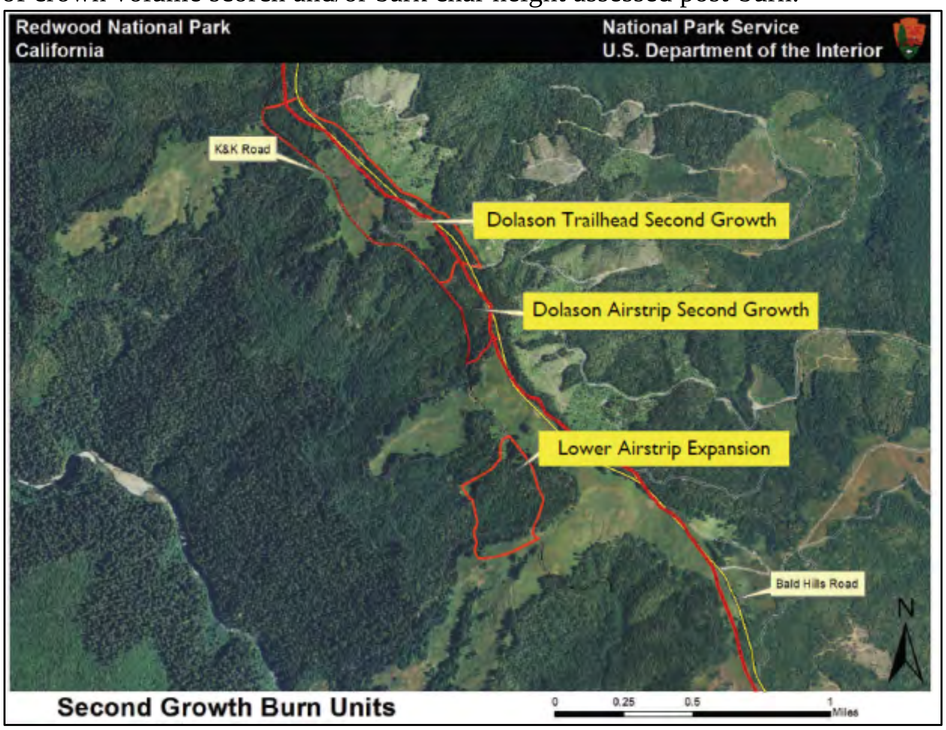

Figure 3—Second-growth burn units in the Bald Hills of Redwood National Park.

Results and Discussion

Outcomes for each of the three burns are presented individually below. Table 1 displays the primary forest attributes monitored for these projects at each measurement, including stand density (trees per hectare [TPHa] and basal area [m^2ha^{-1}, BAHa]), overstory species composition, and fuel loading (fine fuels [1-100Hr] < than 7.6 cm (3 inches) diameter; heavy fuels [1000Hr] > than 7.6 cm diameter). Measures of fire severity, including percent crown volume scorched (excluding DTH where it was not measured) and bole char height, are presented in fig. 4.

Table 1—Results for forest attributes measured for each case study (PRE = prior to fire treatment.; YR-1 and Yr-7 = 1 and 7 years post-treatment, respectively; POST = post burn same season of burn; species composition is in percent)

	Low intensity		Moderate intensity		Mod/high intensity	
Attribute	**LAE PRE**	**LAE YR-1**	**DA PRE**	**DA YR-7**	**DTH - Post**	**DTH YR-7**
TPH*a*	1533	1206	1638	948	469	143
BAHa[b]	86.8	83.5	76.4	76.3	120.4	91.5
Fine fuel TPA	5.3	3.6	4.8	5.9	1.0	7.0
Heavy fuel TPA	11.9	17.7	35.5	28.0	16.7	22.7
			Species composition (%)			
Redwood TPHa	13.7	14.1	17.4	23.5	-	-
Redwood BAHa	39.0	39.6	19.5	20.4	-	-
Douglas-fir TPHa	30.9	34.2	29.2	31.5	75.3	91.3
Douglas-fir BAHa	36.5	36.6	49.2	51.2	99.5	99.99
Tanoak TPHa	45.0	40.9	51.2	44.2	24.7	8.7
Tanoak BAHa	21.0	20.2	30.8	28.0	0.5	0.001

[a] TPHa = Trees per hectare; [b] BAHa = Basal Area, square meters per hectare.

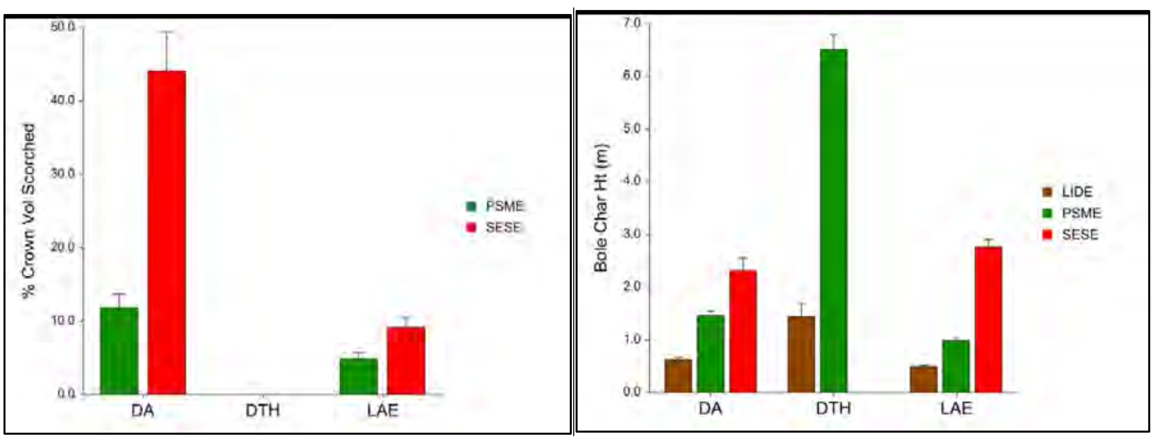

Figure 4—Crown volume scorch (%) (Left) and bole char height (meters) (Right) by species for each of the three prescribed fires. PSME = Douglas-fir, SESE = redwood, LIDE = tanoak.

Case Study #1: Dolason-Airstrip Burn Unit 2007 - Low/Moderate Intensity Prescribed Fire

The Dolason-Airstrip Burn Unit (fig. 5) is a mix of second-growth forest and conifer-encroached prairie. Dominant tree canopy cover is Douglas-fir followed by tanoak and redwood. There had been no prior management of the unit by the Park until the 2007 prescribed burn; monitoring plots were re-measured 1 year post-burn and again in 2014 (7 years post-burn). Fine fuel loading was reduced by about 40 percent immediately following the burn; however by 7 years post-burn, fine fuels recruited 23 percent above pre-burn condition (fig. 6). Heavy fuels remain 21 percent below the pre-burn condition after 7 years.

Figure 5—Fire spread in the Dolason-Airstrip Rx burn.

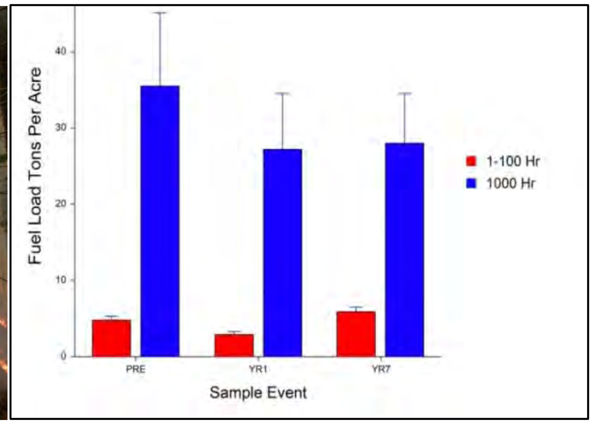

Figure 6—Fine (1-100 hr RED) and heavy (1000 hr BLUE) fuel loading pre-burn, 1 year post-burn, and 7 years post-burn.

Prior to treatment, there was an average of 1,638 trees per ha and 76.4 m^2 of basal area per hectare. Seven years after treatment, there was an average of 948 trees per hectare (42 percent decrease), and 76.3 m^2 of basal area per hectare (no substantial change). Tree mortality was concentrated in the smaller size classes (below 30 cm DBH; fig. 7). Prior to the prescribed burn, there were 68 percent more Douglas-fir trees than redwood trees, and 152 percent more Douglas-fir basal area than redwood basal area. Seven years following the burn entry, there were 34 percent more Douglas-fir trees than redwood; basal area remained unchanged (from 152 to 151 percent) (see table 1 and fig. 7).

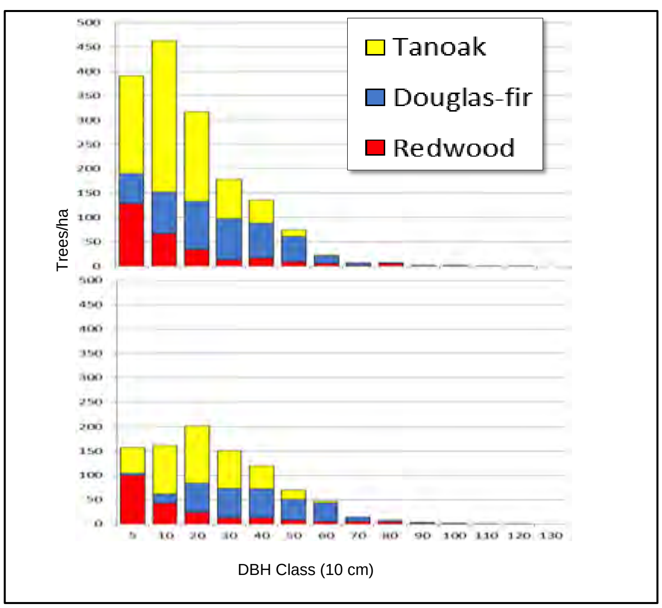

Figure 7—Diameter distribution pre-burn (Top) and 7 years post-burn (Bottom). Bars represent number of trees per hectare (by species) in each 10 cm diameter size class.

Figure 8—Photo-point depicting tree canopy (Top) and understory condition (Bottom) pre-burn (Left), immediate post-burn (Center), and 7 years post-burn (Right). Note resprouting of redwoods from the bole and crown in areas with high crown scorch.

Case Study #2: Dolason Trailhead 2006 - Moderate/High Intensity Prescribed Fire

The Dolason Trailhead burn unit demonstrates the use of moderate-high intensity prescribed fire in Douglas-fir and tanoak forest (fig. 10, fig. 13). Due to circumstances, monitoring plots were installed immediately post-burn, so no true pre-burn data are available. However, at the immediate post-burn sample, the pre-burn stand condition (e.g., living/dead status of trees) was assumed, based on presence/absence of scorched foliage. No pre-burn fuels data are presented. Dominant tree canopy cover is Douglas-fir, followed by a minor understory tanoak component. There had been no prior management of the unit by the Park until the 2006 prescribed fire; monitoring plots were re-measured in 2013.

Fine fuel loading increased by 600 percent between the immediate post-burn sample and the 7 year post-burn sample (table 1, fig. 9). Heavy fuel loading increased by 36 percent between the immediate post-burn sample and the 7 year post-burn sample.

Prior to treatment, we assumed there was an average of 469 live trees per hectare and 120.4 m^2 of basal area per hectare (assumption based on estimated live/dead tree status at the immediate post-burn sample event; table 1). Immediately post-treatment, there were 206 live trees per hectare and 103.5 m^2 of basal area per hectare. Seven years after treatment, there was an average of 143 live trees per hectare (69 percent decrease from pre-burn), and 91.5 m^2 of basal area per hectare (24 percent decrease from pre-burn). Tree mortality was concentrated in the size classes below 60 cm DBH. Tanoak dominated the small size classes (< 30 cm DBH); all but one died immediately following the burn, the single survivor perished by year 7 (fig. 11). Most top-killed tanoak currently exhibit 1 m tall basal sprouts (fig. 12).

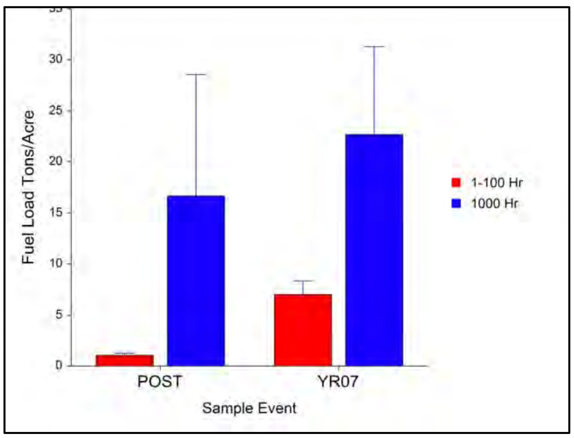

Figure 9—Fine fuel loading (1-100 Hr; Red) and heavy fuel loading (1000 Hr; Blue) immediately post-burn and at 7 years post-burn, Dolason Trailhead Prescribed burn.

Figure 10—Moderate to high intensity fire in the Dolason Trainhead Precribed Burn.

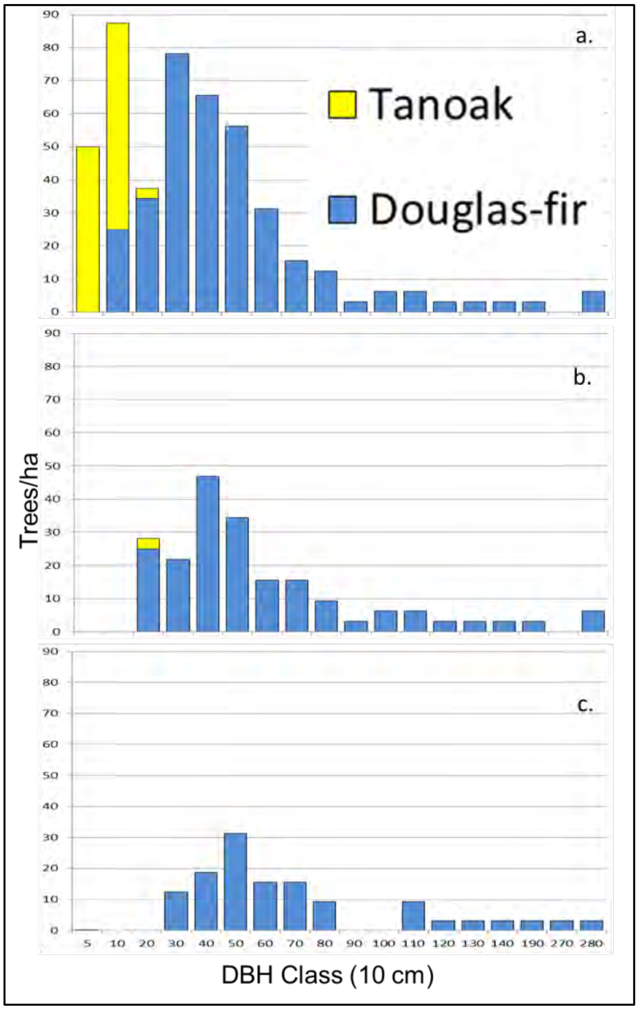

Figure 11—Diameter distribution pre-burn (a) immediately post-burn (b), and 7 years post-burn (c). Bars represent the number of trees per hectare (by species) in each 10 cm diameter size class.

Figure 12—Tanoak mortality and basal sprouting in the Dolason Trailhead Prescribed Burn.

Figure 13—Evident canopy mortality in a high-intensity patch of the Dolason Trainhead Prescribed Burn.

Case Study #3: Lower Airstrip Expansion 2013: Low Intensity Prescribed Fire

The Lower Airstrip burn unit is mostly second-growth forest dominated by Douglas-fir, with lesser amounts of tanoak, redwood, and other hardwoods. There had been minimal management to the unit by the Park until the 2013 prescribed burn. Monitoring plots were re-measured 1 year post-burn.

Fire spread was intermittent throughout the Lower Airstrip Prescribed Burn due to moist fall conditions, and mostly low intensity, with flame lengths typically less than 0.3 m (1 ft) (fig. 14), though there were small pockets (~1/20th ha) of high-intensity fire where fuels, topography, and ignition patterns aligned. 1000 hr fuel moisture was sampled pre-burn and ranged from 27 to 47 percent (mean = 34 percent). Many areas with green vegetation (ferns) or short-needle litter (Douglas-fir fuelbeds) carried fire poorly and resulted in a patchy burn pattern (fig. 15). At 1 year post-burn, fine fuels (1, 10, and 100 hr) were reduced by 33 percent, while heavy fuels (1000 hr sound and rotten) increased by 48 percent.

Figure 14—Low-intensity fire behavior in the Lower Airstrip burn, concentrated in redwood leaf litter.

Figure 15—Heterogeneous burn pattern in the Lower Airstrip Expansion unit resulting from moist fall conditions.

Almost 1400 living and dead trees were sampled in the monitoring plots pre-burn. Overall tree mortality was low at 1 year post-burn, as expected given the low-intensity fire behavior observed during the burn. For overstory trees ≥ 20 cm DBH, the 1 year post-fire mortality rate for redwood was 1.4 percent, 2.1 percent for Douglas-fir, and 2.2 percent for other species (tanoak and California laural [*Umbellularia californica* (Hook. & Arn.) Nutt.]); in other words, overstory composition didn't change much following the burn. Trees smaller than 20 cm DBH exhibited an overall mortality rate of 58 percent for redwood, 25 percent for Douglas-fir, 36 percent for tanoak, and 21 percent for laurel (fig. 18). Mortality of redwood was mostly concentrated in trees smaller than 10 cm DBH, many of which re-sprouted. The greater mortality of redwood compared to other species may be biased since areas with redwood were more likely to burn than areas without (e.g., Douglas-fir dominated patches) under the damp fall conditions.

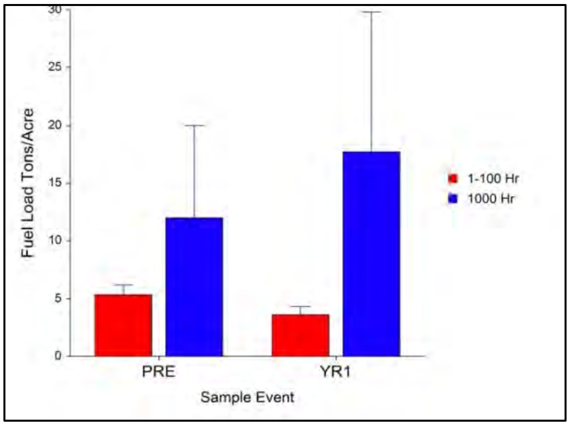

Figure 16—Fine fuel loading (1-100 Hr) and heavy fuel loading (1000 Hr) pre-burn, and at 1 year post-burn. Lower Airstrip Prescribed Burn Unit.

For overstory redwoods, 29 percent of the stems had new basal sprouts at 1 year post-burn, with a mean sprout height of 38.1 cm. Redwood poles had an even more vigorous sprouting response, with 73 percent re-sprouting from the base (mean sprout height of 35.6 cm) and 27 percent from the bole (epicormic). Areas with higher fire intensity appeared to have a more vigorous sprouting response (fig. 17). Portions of the stand with more redwood litter (surface fuel) burned with greater fuel consumption and continuous fire spread due to the flammable redwood leaf litter.

Figure 17—Basal and epicormic sprouting on redwood in the Lower Airstrip Expansion Burn Unit.

If tree mortality and crown scorch data for the DA and LAE burn units are pooled together (both of which had redwood in the overstory), redwood appear to have a greater tolerance for crown scorch than Douglas-fir (fig. 19). For example, Douglas-fir trees killed by fire had an average crown volume scorch close to 40 percent, while redwood killed by fire had an average crown volume scorched over 80 percent. Additionally, there were 52 individual redwoods that suffered 90 percent crown scorch or more; out of these 52 trees, only 26 died, and all but one of those were 15 cm (6 in) DBH or smaller.

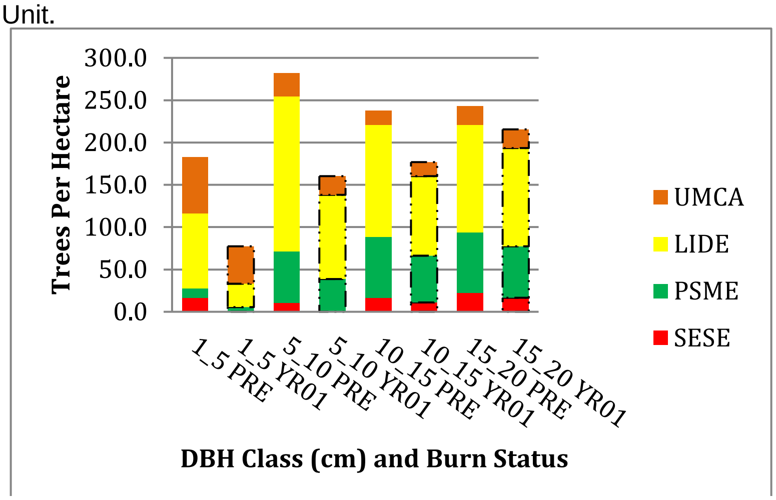

Figure 18—Small tree (< 20 cm DBH) diameter distribution by 5 cm size classes and paired pre-burn and YR-1 condition. Colors represent species. UMCA = California Bay, LIDE = tanoak, PSME = Douglas-fir, SESE = Redwood.

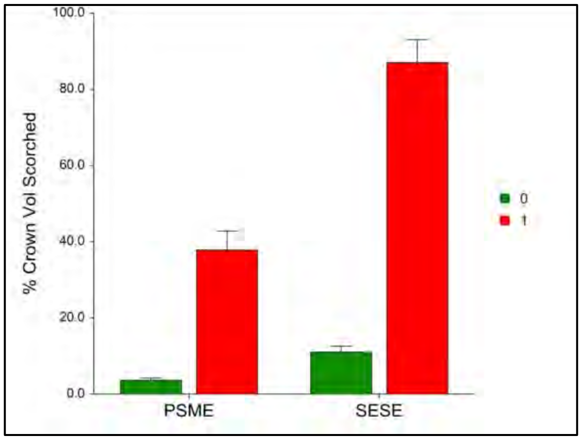

Figure 19—Crown volume scorch (%) and mortality status (0 = living, 1 = dead) for Douglas-fir (PSME) and redwood (SESE) trees in the DA and LAE burn units.

Conclusions

Forest Structure

The low-intensity prescribed fire resulted in relatively small changes to overstory forest structure, with some mortality of smaller trees concentrated below 20 to 30 cm DBH. Moderate to high intensity fire may increase mortality in Douglas-fir up to 60 cm DBH, and top-kill understory tanoak (e.g., DTH burn). Redwoods appear to have a higher tolerance for crown scorch, and with the ability to resprout from the crown and bole, may have a competitive advantage in high-intensity patches. Redwood's main competition—Douglas-fir in the larger classes—may not be affected by low-moderate intensity fire and may require follow-up mechanical thinning to achieve forest restoration objectives. Burning initiates a vigorous sprouting response (basal and epicormic) on pole and overstory redwoods and tanoak, which will result in new cohorts of trees.

Fuels and Fire Behavior

Redwood leaf litter is quite flammable and areas dominated by redwood were observed to burn at a higher intensity and with greater fuel consumption than areas dominated by Douglas-fir. While small redwoods may be susceptible to mortality, competitors are likely to suffer greater injury, and the ability of redwood to resprout may give the species an advantage over Douglas-fir. Fine fuels fully recover, even exceed pre-burn load by 7 years post-burn, suggesting re-entry may be required to maintain low fuel loading. Heavy fuels may remain unchanged (low intensity burns) or increase or decrease (moderate-high intensity burns) by 7 years post-burn.

Management Implications

Prescribed fire may be a viable second growth management tool, with some notable differences to standard silvicultural methods. The application of prescribed fire:

- returns a key process to fire-adapted landscapes and may have unique effects;
- may promote redwood, but with less control over outcomes;
- may not always achieve large Douglas-fir mortality goals while remaining in prescription, requiring mechanical thinning to remove large trees.

The use of prescribed fire in second growth forests warrants further study to assess patterns in tree mortality under hotter burn prescriptions, the role of redwood stump smoldering and consumption in redwood mortality, and understory vegetation recovery following fire.

Acknowledgments

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Influence of Compounding Fires on Coast Redwood Regeneration and Stand Structure[1]

Matthew R Brousil[2] and Sarah Bisbing[2]

Abstract

Disturbance is fundamental to forest ecosystem function, but climate change will continue to increase both disturbance frequency and intensity in the future. Forests subject to increasingly frequent and intense disturbances are more likely to experience overlapping (compounding) disturbance effects. Compounding disturbances may exert unpredicted, non-additive stresses on ecosystems, leading to novel conditions that may exceed the capacity for local species to survive and regenerate. We further hypothesize that compounding disturbances could alter physical and chemical growing conditions in forest soils in ways that disadvantage tree species poorly adapted to the impacts of novel disturbance regimes. Forest composition, structure, and function could shift following increased pressure on one or more species. A better understanding of these remnant effects will be essential to managing and conserving coast redwood (*Sequoia sempervirens* (D. Don) Endl.) forests, which are projected to see increased frequency of fire under future climate scenarios. Our objectives in this study were to quantify the effects of time-since-fire and single vs. compounding disturbances on coast redwood forest structure, composition, and regeneration dynamics, and to evaluate the effects of abiotic soil qualities on post-fire regeneration. We stem mapped and field sampled coast redwood forests burned in 1985, both 1985 and 1999, 2008, and 2013 and modeled observed regeneration counts as a function of single vs. compounding fires, understory light, and post-fire nutrient levels. Forest structure, composition, and regeneration following compounding disturbance were most similar to the redwood-dominated forest of the recent 2013 burn. There were no unique effects of compounding disturbance on soil nutrient levels, although calcium peaked following the most recent fire. None of the predictors in our hypothesized model were significant, showing that soil legacies and nutrients may not be highly influential in regeneration processes following fire and that the number of fires in an area may have a complex influence on regeneration dynamics. However, this study underlines the need for further research into additional compounding fire disturbances in coast redwood forests to confirm whether the observed homogenized forest conditions were the result of compounding fire disturbance. Such outcomes would have negative implications for ecosystem services and overall function if compounding disturbances are more frequent in the future.

Keywords: climate change, coast redwood, compounding disturbance, disturbance interactions, fire, *Sequoia sempervirens*

Introduction

Forests provide invaluable ecosystem services and sociocultural resources, and are adapted to specific historical disturbance regimes. Disturbances are important to forest ecosystem function, but climate change will continue increasing the frequency and intensity of forest disturbances in the future (Dale et al. 2001, Millar and Stephenson 2015). Although ecosystems are often able to maintain function following a disturbance to which they are adapted, altered disturbance conditions can lead to lowered ecosystem tolerance to change (Johnstone et al. 2016). Therefore, increased frequency and intensity in disturbance regimes may exert non-additive impacts on ecosystems, leading to novel conditions that exceed an ecosystem's capacity to survive and regenerate (Buma and Wessman 2011, Metz et al. 2013). Understanding the effects of novel disturbance conditions requires an emphasis on forest resilience, the ability to absorb and adapt to ongoing change (Walker et al. 2004), and the

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mechanisms that confer resilience (Johnstone et al. 2016). However, long-term ecosystem function is at increased risk when the mechanisms that confer ecological resilience have not yet recovered from previous disturbances before a subsequent event occurs (Buma 2015). Such compounding disturbances occur when multiple ecosystem perturbations are either concurrent (e.g., disease and fire) or occur closely enough in time that recovery from the first disturbance is not complete before additional disturbances occur (e.g., two fires in rapid succession; Paine et al. 1998). For example, coast redwood (*Sequoia sempervirens* (D. Don) Endl.) forests can experience unexpectedly high mortality following compounding disturbances (Metz et al. 2013). Research into resilience over a sequence of time since disturbance can inform management decisions under regimes of increasingly frequent disturbance and higher mortality.

Regeneration is a key mechanism of post-disturbance forest resilience, but can decrease markedly following changes in disturbance characteristics (Buma and Wessman 2011). Soil legacies, the functional modifications to soil that linger after disturbances, can exert a strong influence over regeneration density (Vacchiano et al. 2014). The frequency and severity of disturbances influences the production of soil legacies (Certini 2005), including both surface and belowground properties like mineral soil exposure, aggregate stability, reductions in exchangeable K and Mg, cation exchange capacity (CEC), total N and long-term increases in organic carbon (Johnson and Curtis 2001, Vacchiano et al. 2014). A better understanding of soil legacies and their influences is especially relevant in coast redwood forests, which are experiencing increased pressure from multiple types of disturbance, including fire (Metz et al. 2013, Ramage et al. 2010). It follows that novel disturbance regimes could alter physical and chemical growing conditions in forest soils and affect regeneration patterns for years after disturbance events, leading to changes in forest composition and structure.

The objectives of this research were to better understand how single and repeated fires exert controls on coast redwood regeneration, stand structure, and composition by: 1) assessing differences in stand structure and composition over varying times since fire disturbance and between single and repeat burns, 2) quantifying the effect of time since fire and single vs. compounding disturbance on nutrient levels in coast redwood forests and 3) by modeling coast redwood seedling and sprout regeneration as a function of soil nutrient levels, single vs. repeated fires, and understory light.

Materials and Methods

Site Descriptions

Our primary goal in site selection was to identify locations with varying, but evenly spaced amounts of time since last fire disturbance in the Big Sur ecoregion of California. Additionally, we selected sites with north or northeast aspect, 45 to 300 m in elevation, and similar soil types (described below). Our final site selections included two surface fire events at Landels-Hill Big Creek Reserve (“Big Creek”) and two surface fire events at Pfeiffer Big Sur State Park (“Pfeiffer”). Big Creek provided our two oldest burn locations: one area burned in 1985 and a second burned both in 1985 and 1999 (Department of Forestry and Fire Protection 2015; Mark Readdie, personal communication). Burned areas in Pfeiffer comprised our two most burn events: one burned in 2008 and a second burned in 2013 (Department of Forestry and Fire Protection 2015). The 1985 Gorda-Rat fire consumed 22,662 ha (U.S. Department of Agriculture Forest Service 1986), burning the majority of the Big Creek property. We used this fire as the earliest fire event in order to create a timeline from 1985 to present. The second burn site at Big Creek was located in an area last burned in the 1999 Kirk fire but also burned in the 1985 Gorda-Rat fire, making this a compounding burn site. The Kirk fire consumed 35,086 ha and burned more than half of the Big Creek property (National Interagency Fire Center, n.d.). Burned areas in Pfeiffer comprised our two most recent burn events: one area burned in 2008 and a second burned in 2013 (Department of Forestry and Fire Protection 2015). The 2008 Basin-Complex fire burned 65,890 ha, including much of the northwestern portion of Pfeiffer (Department of Forestry and Fire Protection 2015, InciWeb 2008), whereas the 2013 Pfeiffer fire burned 371 ha in a small portion of the park's southwestern edge (InciWeb 2013). Fire perimeter data (Department of Forestry and Fire Protection 2015) for both properties show that our study sites experienced fire infrequently, with a gap of 68 years prior to the 1985 Gorda-Rat fire at Big Creek and no previous fire recorded at Pfeiffer since perimeter records began in the late 1800s. The 1985, 2008, and 2013 burn histories corresponded to one soil type in the NRCS Web Soil Survey (Soil Survey Staff 2014): Gamboa-Sur complex (Gamboa: Haploxerolls, Sur: Haploxerolls; Soil Survey Staff 2003). The 1985/1999 burn histories were from the McCoy series and rock outcrop-Xerorthent association. Like Gamboa and Sur, the McCoy series is a moderately deep and well-drained Mollisol (Argixerolls) but with a higher clay content (Ludington et al. 2005, Soil Survey Staff 2003). Soil pits at each burn site confirmed that underlying soils were very similar across burn histories.

Table 1—Description of soil type, soil series, lithology, and climate factors for the four burn histories used in this study

Site	Burn year	Soil type[a]	Soil series[a]	1st and 2nd most common lithology[b]	Mean annual temperature	Mean annual precipitation
Big Creek Reserve	1985	Very gravelly very fine sandy loam; Stony sandy loam	Gamboa; Sur	Sandstone, mudstone	13.54 °C[c]	67.61 cm[c]
	1985 & 1999	Loam	McCoy	Sandstone, mudstone		
Pfeiffer Big Sur State Park	2008	Very gravelly very fine sandy loam; Stony sandy loam	Gamboa; Sur	Sandstone, mudstone	14.28 °C[d]	99.29 cm[d]
	2013	Very gravelly very fine sandy loam; Stony sandy loam	Gamboa; Sur	Sandstone, mudstone		

[a] Soil Survey Staff 2003.

[b] Ludington et al. 2005.

[c] 12/2005 – 2/2016 (Whale Point Station, Western Regional Climate Center 2016a).

[d] 7/2001 – 2/2016 (Big Sur Station, Western Regional Climate Center 2016b).

Design of the Long-Term Plot System

Beginning in June 2015, we installed plots at Big Creek using randomly selected sampling locations in ArcMap (version 10.2.2, Esri, Redlands, CA, 2014). We selected two locations from each fire with comparable aspect, elevation, slope, and coarse soil designations. In each location per burn, we randomly placed ten 0.01 ha square sampling plots (10 plots x 2 locations x 2 burn years = 40 plots). In the summer of 2016, we continued this sampling in the two additional burn histories at Pfeiffer Big Sur State Park following the same protocol. The purpose of these plots was to allow sampling for soils and coast redwood regeneration and for long-term monitoring of forest community and composition as a result of altered disturbance regimes.

Sampling Overview

Within each 0.01 ha plot we counted, mapped, and measured the diameter of all overstory trees and saplings at 1.37 m (diameter at breast height, DBH) above the ground. All overstory trees were tagged, along with a subset (10 percent) of clumped sprouts and seedlings. We defined sprouts and seedlings as individuals < 1.37 m in height and ≤ 2.5 cm diameter at base. Of individuals meeting these criteria, individuals that were also ≥ 1 m away from a mature tree were categorized as seedlings.

We also quantified abiotic variables relevant to tree regeneration, including understory light availability and soil nutrient levels. At the center of each 0.01 ha subplot, we took a hemispherical photograph using a Nikon Coolpix 5000 35 mm digital camera and Nikon FC-E8 0.21x lens (Nikon, Tokyo, Japan) mounted 1 m above the ground and positioned with the top side of the camera facing north (Pastur et al. 2012). A tripod with bubble level was used to correct for uneven terrain below the camera. We used Gap Light Analyzer (version 2.0, Frazer et al. 1999) to process the hemispherical photos and calculate percent total light at each subplot center. Soil sampling for chemical analysis and detection of soil legacies from fire occurred on all 80 plots in May 2016. Soil samples were collected from the A horizon (0 to 10 cm depth) of each site. Litter and duff layers were first discarded from the soil surface. Twelve equally sized soil subsamples were then collected in a stratified sampling design across each subplot and mixed for a composite sample. We air dried soil samples to a constant weight prior to chemical analysis, sieved them (using a 2 mm screen), and sent them to A&L Western Agricultural Labs (Modesto, CA, USA) for processing for macronutrient content. Nutrients of interest (NO_3^-, NH_4^+, PO_4^{3-}, and Ca^{2+}) for analysis were selected based on previous studies indicating their importance in the Cupressaceae family (of which coast redwood is a member) and in other forest ecosystems (Hawkins and Robbins 2014, Rao and Rains 1976, Trant et al. 2016).

Data Analysis

We tested for changes in soil nutrient status following burn events and then modeled coast redwood regeneration as a function of soil qualities and abiotic site variables. We used ANOVA to compare soil nutrient levels between burn histories and employed Tukey's HSD test to distinguish nutrient levels between burn years for instances where an ANOVA global *F* test was significant. We then used a generalized linear model (GLM) with a negative binomial distribution and log link function to model counts of coast redwood seedlings in each random subplot as a function of burn year, levels of soil nutrients, and percent available light (Bolker et al. 2009). We tested a full model including burn event, understory light, NO_3, NH_4, P, and Ca^{2+} to determine whether nutrient levels across burn years were drivers of redwood regeneration. Statistical analyses were run in *R* (R Core Team 2015), with the MASS package employed for GLM analysis (Venables and Ripley 2002).

Results and Discussion

Diameter distribution among the four burn histories showed distinct differences (figs. 1 and 2). In all burn histories, the average stand density (trees per hectare) was highest in small sizes classes (0 to 2.5 cm and 2.6 to 10.0 cm DBH). However, in the two oldest single fire areas (1985 and 2008), the diameter distributions show patterns typical of uneven-aged stands, compared with more even-aged stand characteristics in the diameter distributions of the compounding fire disturbance (1985/1999) and in the most recent single fire disturbance (2013). The compounding fire disturbance was also most similar to the 2013 single fire disturbance event in the amount of regeneration (0 to 2.5 cm DBH stems). These results indicate that the compounding fire disturbance in these coast redwood forests may have resulted in an environment that remains similar to more recently burned areas, even 16 years following disturbance. Moreover, the compounding disturbance plot is noticeably more homogenous, with coast redwood overwhelmingly dominant, than either the 1985 or 2008 single disturbances, though it is comparable to the makeup of the 2013 burn. This may indicate that the compounding disturbance drove a shift to a forest environment dominated by a more fire-adapted species. These results would be consistent with the hypothesis that fire reduces tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh) dominance compared with coast redwood in recently burned areas due to higher coast redwood survival in the overstory (Lazzeri-Aerts and Russell 2014, Ramage et al. 2010), higher basal sprout density (Lazzeri-Aerts and Russell 2014), and reduced tanoak sprout dominance in burned compared with unburned areas (Ramage et al. 2010). Additionally, fire perimeter records beginning in the late 1800s show that these areas of forest have re-burned at longer intervals than the 16 years between fires on our sites (Department of Forestry and Fire Protection 2015). However, data are not available on the pre-fire composition of these stands, so a component of these differences may also be related to site-specific variables.

Figure 1—Distribution of stem diameter classes across four burn histories. Distribution of overstory (> 5 cm diameter at 1.37 m height) tree diameter size classes across four burn histories by average trees per hectare (TPH). The 1985/1999 burn was burned twice and is considered a compounding disturbance. UMCA = *Umbellularia californica*, SESE = *Sequoia sempervirens*, QUWI = *Quercus wislizeni*, LIDE = *Lithocarpus densiflorus*.

Some soil nutrients examined (Ca^{2+}, PO_4^{3-}, NO_3^-, and NH_4^+) showed differences among burn histories. Comparisons between burn years for Ca^{2+} showed significantly higher levels in the 2013 burn ($F = 6.2395$; $P = 0.0014$; fig. 3). Phosphorus levels were significantly higher in the 2008 burn, and lowest in the twice-burned 1985/1999 burn ($F = 8.6996$; $P = 0.0002$; fig. 4). For nitrogen, NO_3^- levels did not significantly differ between burn years ($F = 2.0433$; $P = 0.1235$) but NH_4^+ levels were significantly higher in the area burned once in 1985 ($F = 4.2612$; $P = 0.0106$; Figures 5 and 6). Our results support the findings of previous work, in which Ca^{2+}concentrations increased following slash burning in eucalyptus forest and remained high for 2 years (Tomkins et al. 1991). Other studies have noted similar patterns with some variation: for example, increased Ca^{2+} following fire in a jack pine (*Pinus banksiana* Lamb.) barren for 1 year (Smith 1970), or up to 21 years following wildfire in a boreal forest in Quebec (Simard et al. 2001). However, our work contrasts with other studies documenting pulses in Ca^{2+} lasting shorter (< 2 years, Khanna and Raison 1986) periods of time. Compounding fire disturbances did not result in significantly higher long-term Ca^{2+} levels in our study. Phosphorus availability often increases within the first year following a fire (Romanyà et al. 1994), but it can be followed by a quicker, though more variable decline than that of Ca^{2+} (Macadam 1987). In our study, phosphorus was highest eight years post-fire and lowest in the compounding 1985/1999 burn, but the 2013 burn and 1985 single burn were indistinguishable from one another.

This is contrary to what previous work suggests about post-fire phosphorus dynamics. Our sampling may have missed an early peak for the most recently burned sites and additional variation could reflect site-specific characteristics. Nitrogen fluctuations in our burned areas generally followed the pattern of past research. Previous studies have documented pulses in NH_4^+ for a year or more following fire with a lagged increase in NO_3, but increases in one or both forms of inorganic nitrogen may be gone within one to five years (Covington et al. 1991, Covington and Sackett 1992, Grogan et al. 2000). In this study, NH_4^+ concentrations were significantly higher in the single 1985 burn compared with any other burn, but there were no other significant differences between nitrogen levels and burn year. This could be an indication that initial pulses in one or both of these nutrients did occur but dissipated by the time of our sampling, though.

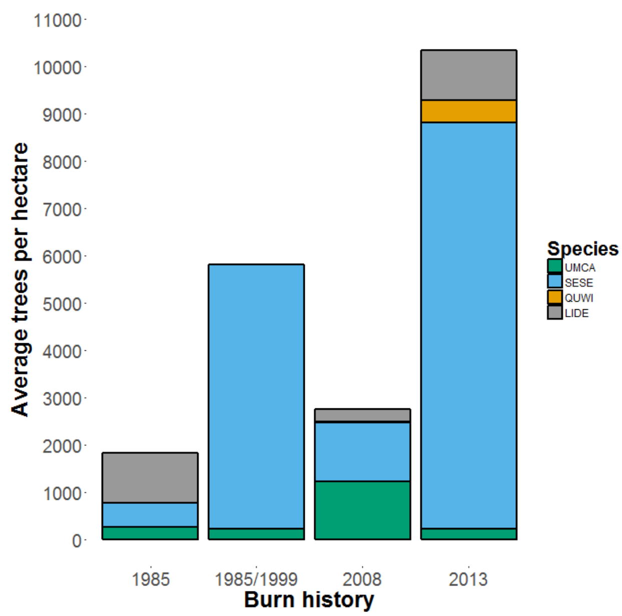

Figure 2—Species makeup for regeneration in the 0 to 2.5 cm diameter class (diameter at 1.37 m height) across four burn histories by average trees per hectare (TPH). The 1985/1999 burn was burned twice and is considered a compounding disturbance. UMCA = *Umbellularia californica*, SESE = *Sequoia sempervirens*, QUWI = *Quercus wislizeni*, LIDE = *Lithocarpus densiflorus*.

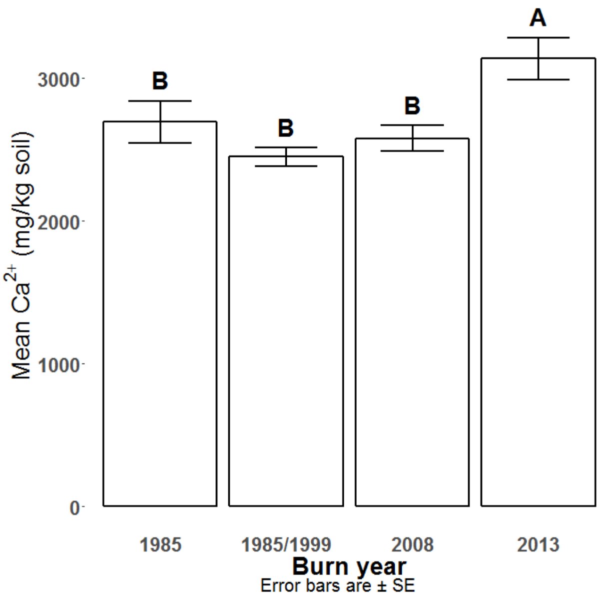

Figure 3—Ca^{2+} levels across four burn histories. Bars not connected by the same letter are statistically different from one another ($p < 0.05$).

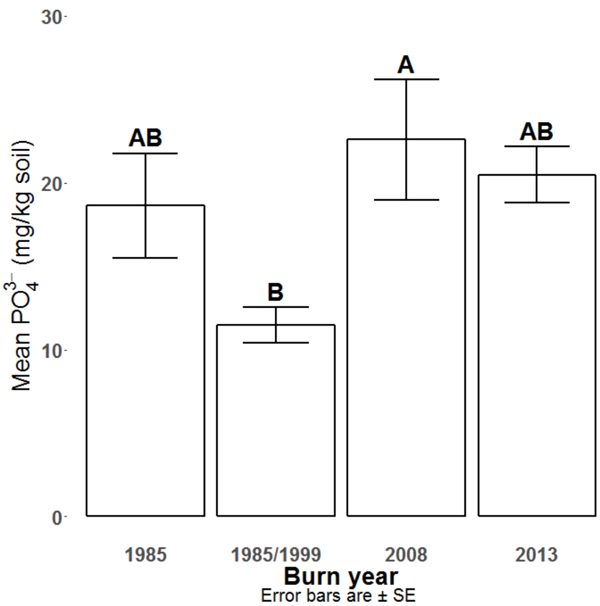

Figure 4—PO_4^{3-} levels across four burn histories. Bars not connected by the same letter are statistically different from one another ($p < 0.05$).

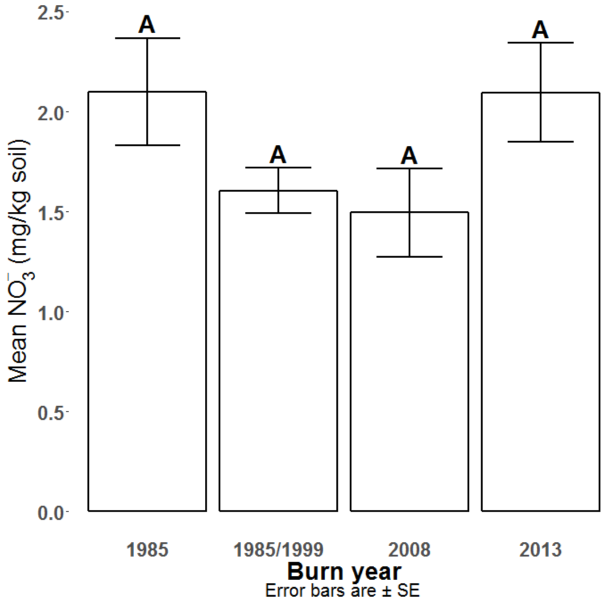

Figure 5—NO_3^- levels across four burn histories. Bars not connected by the same letter are statistically different from one another (p < 0.05).

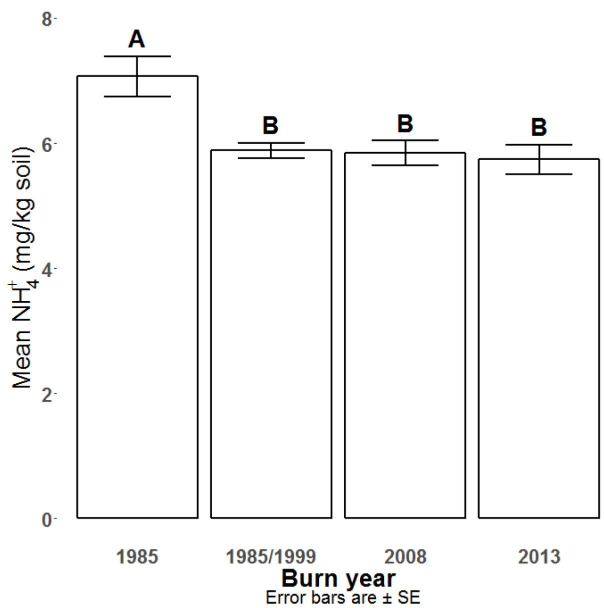

Figure 6—NH_4^+ levels across four burn histories. Bars not connected by the same letter are statistically different from one another (p < 0.05).

We used a generalized linear model to test the hypothesis that Ca^{2+}, PO_4^{3-}, NO_3^-, and NH_4^+ levels would be important predictors of coast redwood regeneration in addition to burn year and percent total understory light in our study sites. However, none of the variables was significant in predicting the amount of coast redwood regeneration (table 2). These results contrast findings from Vacchiano et al. (2014), which showed that decreases in nutrient levels from fire were associated with increased Scots pine (*Pinus sylvestris* L.) regeneration. Our findings do not indicate preference for post-fire establishment conditions in coast redwood forests both in terms of belowground soil conditions and aboveground light availability.

Table 2—Output of generalized linear model (GLM) analysis for selected predictor variables modeling observed numbers of coast redwood regeneration at the subplot level

Variable	Coefficient	*p* value
Percent total light	0.0057	0.8301
Burn year 1985/1999	0.7070	0.0517
Burn year 2013	0.3046	0.4320
NO_3	0.1834	0.2038
NH_4	-0.0995	0.4160
Ca	0.0001	0.6172
P	-0.0100	0.4603

The results of this study must be considered alongside several limitations. Due to restraints in site access and resources we were unable to conduct research on more than one coast redwood forest area that experienced compounding fire disturbance and we did not locate an unburned (control) forest. Additionally, we did not model fire behavior as predictor of changes in stand structure and composition, and pre-fire stand composition and structure data were not available for any of our research locations. Thus, since we did not compare multiple compounding fire events, some of the observed change in stand structure and composition following compounding disturbance may be remnants of pre-disturbance forest characteristics or of variable fire behavior. Changes in soil nutrient availability may be partially related to site effects since we do not have undisturbed areas as reference, although the lack of replication of specific amounts of time since burn and numbers of burns prevents us from drawing firm conclusions.

Conclusion

In this study, a forest experiencing two fires 16 years prior to sampling showed similarities in stand structure and composition to a forest that experienced a single burn event 2 ½ years prior to sampling. Compounding and recently disturbed forests were similar in stand structure and increased coast redwood dominance. However, compounding fire disturbance did not leave a detectable legacy in a selection of soil nutrients that we expected to be influential for coast redwood regeneration. Despite differences in nutrient and stand structure between burn years and previously described relationships between light availability and redwood regeneration (O'Hara and Berrill 2010), these factors were not significant predictors of coast redwood regeneration. Taken together, these results may indicate that changes in coast redwood forests toward more redwood-dominated stand composition may be related to compounding fire disturbance. However, because we did not have access to multiple compounding burn sites and pre-fire stand composition and structure data, additional research is needed in order to confirm the effects of compounding disturbance in coast redwood forests. For example, current stand composition may reflect pre-fire coast redwood dominance in addition to effects of compounding fire disturbance. If future work in this ecosystem confirms the relative declines of species like tanoak that we observed following compounding disturbance, it will be important to assess what effects local declines of these species would have on ecosystem services provided by these and other associated species (Bowcutt 2014). Soil legacies specific to compounding fire disturbances do not appear to be present, and nutrient levels in our study did not seem to be limiting for coast redwood regeneration in these forests. Influences of soil conditions on stand structure and composition seen in other research (Johnstone et al. 2010, Vacchiano et al. 2014) are not confirmed in our study, showing that these feedbacks can be indirect and ecosystem specific. Our research supports previous evidence that coast redwood benefits from fire disturbance (Lazzeri-Aerts and Russell 2014), but we show that a greater understanding of compounding disturbance impacts on resilience adaptations is needed in this and other forest ecosystems. Research in other forest ecosystems shows that compounding disturbances can disrupt resilience adaptations (Buma and Wessman 2011, Gower et al. 2015). Future studies should seek to confirm whether the effects seen here can be isolated from site-specific or pre-fire stand conditions, as shifts in composition may be indicative of increased ecosystem vulnerability to change (Johnstone et al. 2016). Therefore, greater knowledge of the traits that confer ecological resilience and function in these forests will be important for understanding the impacts of altered disturbance regimes in the future.

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Coast Redwood Seedling Regeneration Following Fire in a Southern Coast Redwood Forest[1]

Rachel Lazzeri-Aerts[2] and Will Russell[2]

It has been hypothesized that individuals adapted to conditions near the species' range edge, may increase the likelihood that the species will persevere under changing climatic conditions (Rehm et al. 2015). The southern coast redwood (*Sequoia sempervirens* (D. Don) Endl.) forests vary from more northern redwood forests in terms of stand size, genetics, forest associates, and have less annual precipitation (Noss 2000). Redwoods regenerate through both seed and basal sprouting, with sprouting being the dominate regeneration method (Douhovnikoff et al. 2004). A study of three recent fires in the Santa Cruz Mountains, California found that one site had significantly more redwood seedlings 1 year post-fire than the other two sites (Lazzeri-Aerts and Russell 2014). Our objectives were to: 1) quantify seedling germination and size, and 2) characterize environmental conditions on this site 8 years post-fire. Prolific seedling recruitment and growth may help the species persist under changing climate conditions. As conditions become drier and warmer in the southern part of the redwood range, the increased ability to regenerate through seed germination may be an advantage.

The Whitehurst Fire burned 103 ha (256 ac) within the Mt. Madonna County Park, Santa Clara County, California in June 2008 within a mixed coast redwood forest. Twenty-six plots were sampled in 2009 and 2016 for number of seedlings in three size classes (small: < 0.05 m (< 1.64 ft), medium: 0.5 to 1.0 m (1.64 to 3.28 ft), and large: > 1.0 m (> 3.28 ft)), height of tallest seedling, canopy cover, shrub cover, duff depth, soil moisture, slope, and aspect.

On average, more redwood seedlings were found than seedlings of three other common forest associates—madrone (*Arbutus menziesii* Pursh), tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh), and mixed oak (*Quercus agrifolia* Née, *Q. chrysolepis* Liebm., *Q. wislizeni* A. DC.)—in both 2009 and 2016. There were fewer redwood and madrone seedlings per square meter in 2016 than 2009, but more tanoak and oak seedlings per square meter in 2016 (fig. 1). The difference in mean number of seedlings was significantly lower for redwood ($p = 0.03$) and significantly higher for tanoak ($p = 0.0004$).

[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.

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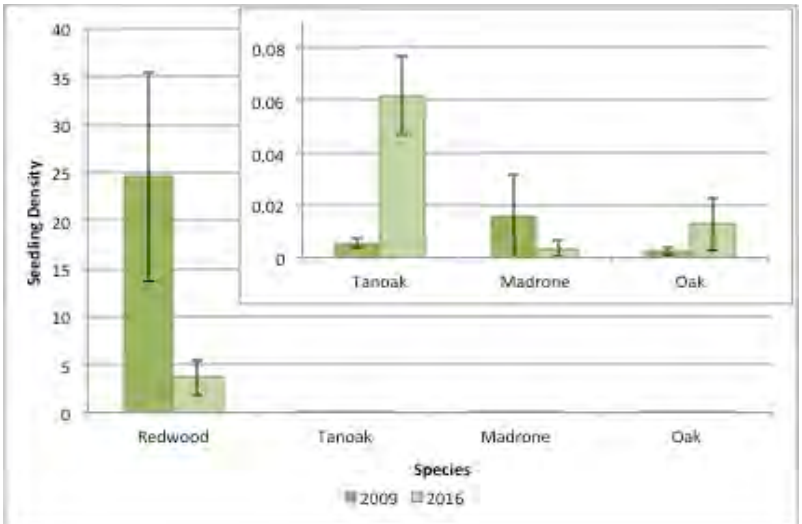

Figure 1—Mean (+/- 1 SE) seedling density (per m^2) 1 and 8 years after fire in a mixed coast redwood forest in the Santa Cruz Mountains, California. Inset figure highlights changes in seedling density for low density tree species. Units of inset figure are the same as main figure.

Between 2009 and 2016, most plots had fewer redwood seedlings and mortality ranged from 22 percent to 100 percent. However, six plots had more redwood seedlings in 2016 compared to 2009. The largest increase was from 75 to 3,623 total seedlings. The difference in mean number of seedlings on these six plots between 2009 and 2016 was significant ($p < 0.001$). These results indicate that new seedlings continued to germinate up to 8 years post-fire.

In 2009, all measured seedlings were in the small size class except one oak in the medium size class. The tallest redwood seedling was 0.28 m (0.92 ft). By 2016, 2.45 percent of redwood seedlings were in the large size class, 13.06 percent were in the medium size class, and 84.49 percent were in the small size class (fig. 2). The tallest redwood seedling measured was 3.0 m (9.8 ft), and the tallest seedling of any other species was a madrone at 1.6 m (5.2 ft).

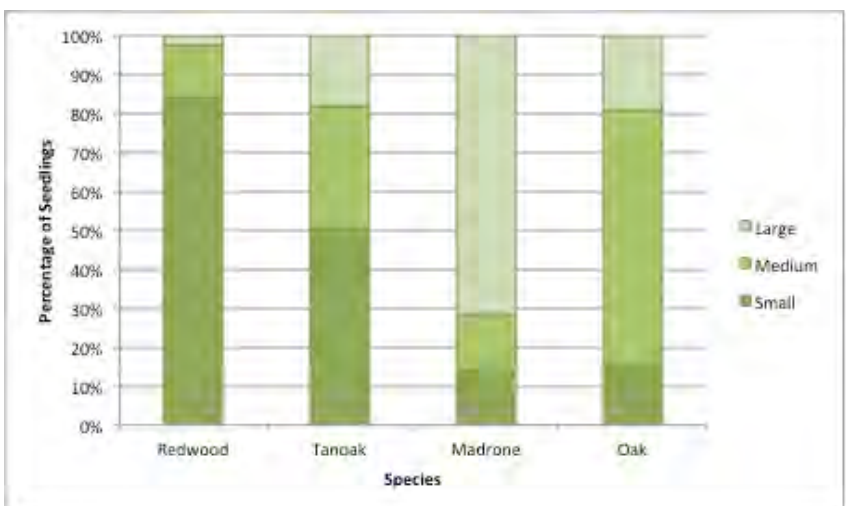

Figure 2—Percentage of seedlings in three size classes (small: < 0.05 m, medium: 0.5 to 1.0 m, and large: > 1.0 m) 8 years after fire in a mixed coat redwood forest in the Santa Cruz Mountains, California.

Percent canopy cover was measured with a spherical densitometer and we found a significant increase in the percent canopy cover ($p = 0.0002$) from 2009 to 2016. The mean canopy cover was

68.7 percent and 80.0 percent in 2009 and 2016, respectively. However, we found no correlation between canopy cover and number of seedlings. Mean duff depth was 3.4 cm (1.3 in) and 4.2 cm (1.6 in) in 2009 and 2016, respectively. The difference between duff depths was not statistically significant, and we found no correlation between duff depth and number of seedlings for either year. While shrub cover was not measured in 2009, we observed few shrubs in and around the plots. In 2016, we found a 16 percent mean shrub cover, but there was no correlation between percent shrub cover and number of seedlings. Additionally, slope and aspect were not significantly correlated with the number of redwood seedlings in either 2009 or 2016, and soil moisture was not significantly correlated with the number of redwood seedlings in 2016.

Although the overall number and density of redwood seedlings declined between 2009 and 2016, redwood seedlings are continuing to germinate as evidenced by the six plots with more seedlings in 2016 than 2009. Additionally, many of the seedlings increased in size over the same time span. This result was unexpected as seedlings typically have low vigor (Olson et al. 1990). While we measured no redwood seedlings in the medium or large size classes in 2009, a combined 15.51 percent of the seedlings we measured in 2016 were large or medium. This shows that some of the redwood seedlings are thriving. Olson et al. (1990) suggest that seedlings need high soil moisture levels and full sunlight to grow best after germination. However, we found no statistically significant relationship between seedling density or seedling height and soil moisture or canopy cover (as a proxy for light availability). Combined with the lack of relationship to duff depth, shrub cover, slope, and aspect, this leaves the cause of continuing seedling recruitment and increasing vigor on this site unexplained. As abundant seedling germination and growth may be advantageous with continuing climatic changes, we suggest ongoing investigations of southern coast redwood regeneration. These studies should focus on genetics, soil, and past land use and management.

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SESSION 3 – Watersheds and Aquatic Ecology

Watershed-Scale Evaluation of Humboldt Redwood Company's Habitat Conservation Plan Timber Harvest Best Management Practices, Railroad Gulch, Elk River, California[1]

Andrew Stubblefield,[2] Shane Beach,[3] Nicolas M. Harrison,[3] and Michelle Haskins[2]

Abstract

The objective of this study is to test the effectiveness of California Forest Practice Rules and additional best management practices implemented as part of Humboldt Redwood Company's Habitat Conservation Plan and Watershed Analysis prescriptions, in limiting the delivery of management-derived sediment. A paired watershed study format is being utilized to evaluate sediment sources: road surfaces, watercourse crossings, landslides, channel incision and bank erosion, and tributary channel head-cutting. The study compares the West Branch (1.48 km^2, 365 ac) and the East Branch (1.28 km^2, 314 ac) of Railroad Gulch, a tributary to the Elk River (152 km^2) which flows into Humboldt Bay just south of Eureka, California. The watershed has been intensely logged in the past and is underlain by two erodible geologic terranes, the Pleistocene age Hookton Formation and the late Miocene to middle Quaternary age undifferentiated Wildcat Group.

Forty seven percent of the East Branch was logged in the summer of 2016 under Timber Harvest Plan 1-12-110 HUM, with 32 ha (80 ac) of single tree selection, 18 ha (45 ac) of group selection, and 10 ha (24 ac) of no-cut zone left as buffer strips along Class I and II watercourses. A new native surfaced seasonal road was constructed in the summer of 2015. Several existing roadways were reopened during the same time period. All of these roads are appurtenant to the plan and were utilized for hauling throughout the summer of 2016. Cable yarding systems were used.

No timber operations will occur within the West Branch, which will serve as the study control. Methods used to evaluate prescription performance include: pre- and post-construction road inventory and characterization, turbidity synoptic sampling during storm events, landslide inventories, channel cross-section surveys, pebble counts, continuous turbidity, stage, and rainfall monitoring, peak flow analysis, and an analysis of Beryllium-10 (Be-10) isotopes to estimate for long-term (e.g., 1000 to 10,000 year) erosion rates.

Data collection for the study began in earnest in 2014, with limited streamside landslide data collected in 2013. Sediment loads were well correlated between the two branches during an extremely dry water year (WY) and a below average water year. In WY 2014 the total annual suspended sediment load equaled 49 Mg (metric tons) km^{-2} (0.22 t ac^{-1}) in the East Branch and 38 Mg km^{-2} (.017 t ac^{-1}) in the West Branch. Loads were tenfold higher in WY 2015 in conjunction with a single large storm event which triggered several debris torrents in addition to several streamside failures. The WY 2015 total annual sediment load equaled 861 Mg km^{-2} (3.8 t ac^{-1}) in the East Branch and 716 Mg km^{-2} (3.2 t ac^{-1}) in the West Branch. Historically active debris slides and earthflows cover approximately 6 percent of the study basin. Five active upland failures have been detected during the project period; two in WY 2014 and three in WY 2015. Several of these active landslides are hydrologically connected and at selected sites were observed to strongly influence downstream turbidity. Cross-sectional surveys indicated that channel banks remained stable with limited thalweg scour between 2014 and 2015. Post-harvest monitoring will continue through 2019.

[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.

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Introduction

Improved harvesting techniques such as selective harvesting, cable yarding and protection of soil with slash placement, combined with better road construction and maintenance practices, have been implemented by California timber companies in recent years in compliance with California Forest Practice Rules. The rule changes have come in response to declining salmonid populations and the 303d listing of many North Coast rivers for impairment of beneficial uses by sedimentation. In addition, Humboldt Redwood Company (HRC) has implemented best management practices as part of a Habitat Conservation Plan (HCP) and Watershed Analysis prescriptions. There is a need to investigate whether these new practices are having the desired effect in preventing discharge of sediment into rivers and minimizing the activation of erosion sources on forested landscapes.

The approach of this project is to implement a paired watershed study with a control and treatment basin that retain similar geologic, geomorphic and hydrologic traits and to collect monitoring data before, during and after road-building and harvest activities occur. The main goal of this study is to quantify the effects of land use on stream suspended sediment loads and turbidity levels, and through detailed monitoring of stream, road and hillslope condition, be able to identify the sources of any observed elevated sediment. The study will also evaluate changes to peak flow magnitude and timing resulting from timber operations, and evaluate long-term basin wide erosion denudation rates. These latter analyses and road condition surveys will be reported separately. This article reports on suspended sediment loads and sediment sources for the first two years of the study before road building and harvest activities occurred.

Methods

Study Site

The Railroad Gulch drainage, a tributary to the Elk River, Humboldt County, California, was selected for study. It contains a West Branch and East Branch of approximately equal size (1.48 km^2 and 1.28 km^2 respectively), aspect (N-facing), drainage networks, slopes and geology. Railroad Gulch is underlain by sediments associated with the Middle to Late Pleistocene aged Hookton Formation and the Miocene to Late Pliocene aged Undifferentiated Wildcat sediments. These bedrock types are highly erodible and subject to both shallow and deep-seated mass movements. The subject basins are within Humboldt Redwoods Company’s ownership.

Following initial clear-cutting and railroad harvesting in the early 1900s, this basin became restocked with dense second-growth stands comprised of various types of conifer and hardwoods. Stands are currently composed of approximately 85 percent redwood (*Sequoia sempervirens* (D. Don) Endl.), 12 percent Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), 2 percent grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), and Sitka spruce (*Picea sitchensis* (Bong.) Carrière, and 1 percent hardwoods (primarily red alder, *Alnus rubra* Bong.). Selection and even-aged silvicultural practices were applied to the second growth stands within the study area between 1987 and 2002. Between 2001 and 2008, 25.9 ha (64 ac), distributed between both branches, were clearcut. Current stands are primarily even aged.

Forty seven percent of the East Branch was initially entered in the summer of 2015 to open existing roadways and to construct new road. These roads are natural surfaced seasonal haul roads that were constructed and storm-proofed in accordance with HRC’s HCP prescriptions. Approximately 500 m of this haul road network does pass through the lower reaches of the West Branch basin (control). Road surfaces along this segment that fall within Riparian Management Zone (RMZ) were rocked and erosion-control wattles (rolls of absorbent coconut fiber batting) were installed across all water bars, diversion potential dips, and rolling dip outfalls that directed surface water toward the West Branch to discourage road-derived sediment from entering the control basin.

In the summer of 2016, timber in the East Branch was harvested under Timber Harvest Plan 1-12-110 HUM during. Silvicultural prescriptions consist of 32.4 ha (80 ac) of single tree selection, (18.2

ha (45 ac) of group selection, and 9.7 ha (24 ac) of Class I and II watercourse RMZs. These RMZs act as buffer strips along the larger order waterway and were designated as "no-cut" areas per HRC's HCP and watershed analysis prescriptions. Cable yarding systems were used.

New seasonal roads were constructed in the summer of 2015, and existing roadways reopened. All new and re-opened roadways were utilized for hauling during 2016 logging operations.

Procedures

Stream and Road Crossing Turbidity

Continuous suspended sediment concentration (SSC) data are obtained during the wet season (October through May). Data is obtained using in situ turbidometers and autosampling equipment and turbidity threshold sampling methods as described in Lewis and Eads (2001). Auto-samplers installed at gaging stations at the outlet of each subbasin, in conjunction with a staff plate and stage logger. Autosampling is triggered by field technicians in advance of storms expected to exceed 2.5 cm of precipitation in a 24 hour period. Manually triggered sampling was found to be more reliable than turbidity threshold sampling for this watershed. Water samples collected during storm events are processed for SSC using gravimetric methods (Gray et al. 2000). Annual suspended sediment yields are calculated by multiplying stream discharge times the storm-specific turbidity-suspended sediment relationships. Peak flow data will be analyzed in subsequent reports to explore any potential changes due to harvest. Bedload was not measured for this study. Samples of channel alluvium and hillslope sediment were taken for estimation of long-term denudation rates using Be-10 dating methods (Balco et al. 2013, Ferrier et al. 2005). However results from that analysis are not yet available.

Synoptic samples are collected during storm events that surpass the 2.5 cm rainfall threshold. Storms of this magnitude occur an average of nine times per year. Synoptic samples are also collected at specified locations along the main stems of both basins. Beginning at the confluence at the same time, team members walk upstream in each branch and collect water samples from the mainstem, tributary channels. Samples are also collected above and below newly constructed, newly reconstructed, and existing road crossings. Refer to fig. 1 for synoptic sample locations. Samples are then processed for turbidity level using a benchtop turbidimeter (Hach Instruments, Loveland, CO).

Streamside Landslide and Bank Erosion

A survey is conducted annually along an approximately 792.5 m (2,600 ft) reach of both the West and East Branches to identify active streamside failures. Landslide sediment contributions are determined by measuring the void created by the streamside mass movement. Movement depths are ascertained by visually reconstructing the pre-slide slope configuration and estimating the maximum thickness of the material lost. Percent delivery is based on ocular estimates and the amount of debris remaining at the toe of the subject failures. All active slides are photographed, flagged, labeled, and painted for identification for subsequent site visits.

Landslide Inventory

Five sets of stereo-paired aerial photographs dated between 1948 and 2010 were used to identify recently active landslides for each photo year. Numerous deep-seated, compound failures and shallow debris slides were noted and subsequently verified through field surveys. During the verification surveys numerous small scale failures, not identified through aerial photograph analysis, were encountered and mapped. A compilation map containing all those failures identified during recent and past investigations as dormant-historic in age (equal to or less than 100 years in age per Keaton and DeGraff 1996) are shown on fig. 2. Annual resurveys are conducted to detect new or reactivated landslides.

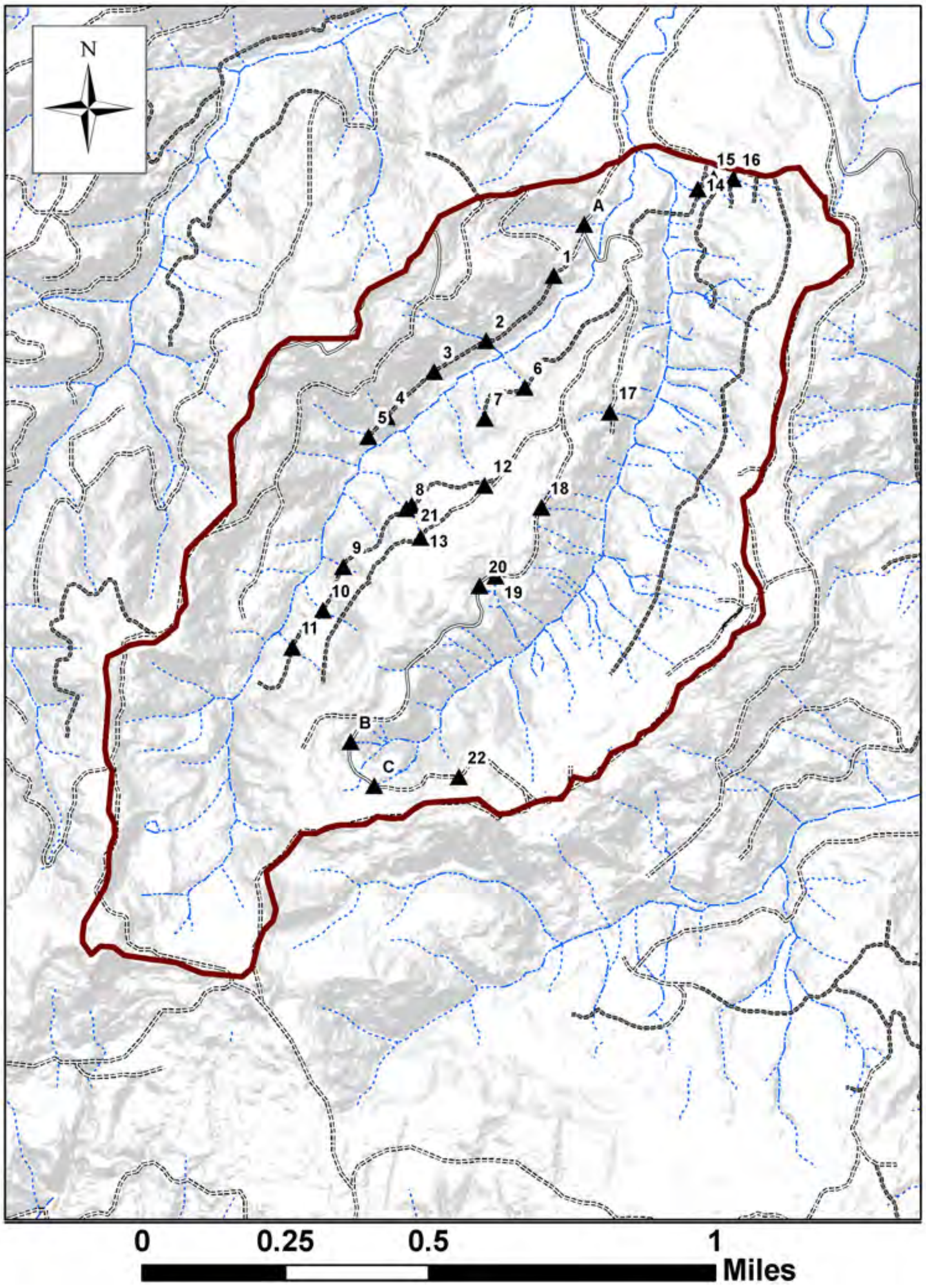

Figure 1—Synoptic sampling locations at road construction sites, Railroad Gulch, Elk River, California. Brown line: project boundary; blue lines: watercourses; dashed black lines: roads; black triangles: synoptic sampling locations at road construction sites.

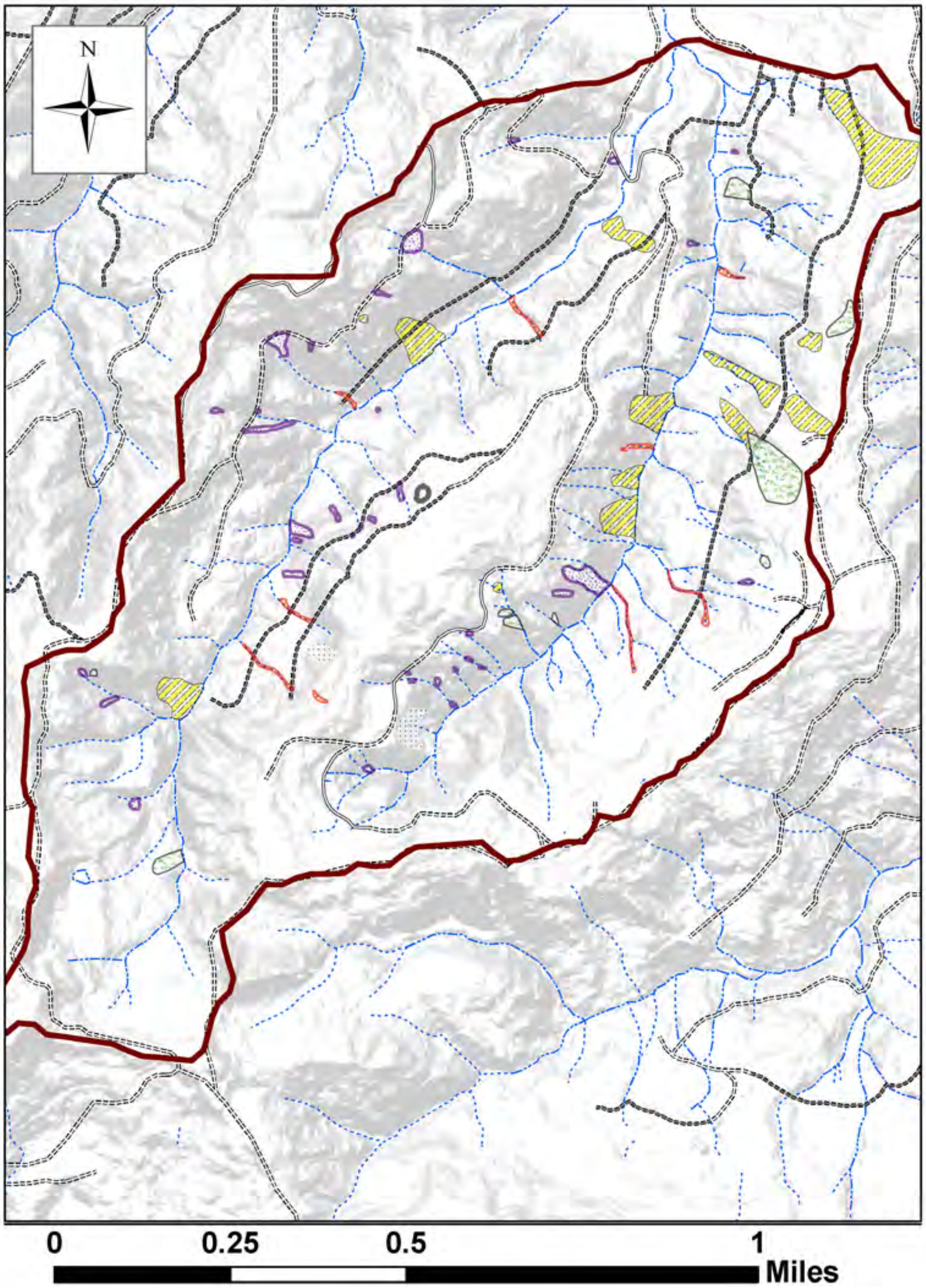

Figure 2—Landslide locations, Railroad Gulch, Elk River, California. Brown line: project boundary; blue lines: watercourses; dashed black lines: roads; red outlines: debris flows; thick grey outlines: disrupted ground; polka dots with purple outlines: debris slides; polka dots (no outline): debris slide slopes; yellow patterns: earthflows; green patterns: trans/rotational landslides.

Channel Head Migration

Channel head migration is monitored by annually assessing rebar monuments established at the upper most point of recognizable erosion (knickpoint, rill, soil pipe, etc.) on first order watercourses. Thirty-eight sites were geo-referenced, characterized, and flagged; 20 new plots were added in the summer of 2015. Several of the plots installed in 2013 were found during the 2015 survey to terminate in some type of hardscape (woody debris, bedrock, etc.) or were not located at the upper most point of erosion. This study reports on the findings relating to the summer 2015 inspection of the initial thirty eight plots.

Road Surface Erosion

Road surveys were conducted for control, pre-treatment and post-harvest road segments to estimate rill and gully erosion occurrence and potential for delivery to the stream network. Road condition and grade were recorded for each road segment using methodology similar to MacDonald and Ramos-Scharrón (2005). Road segments were defined as lengths between naturally occurring (i.e., ridgetops) and structural (i.e., water bars) hydrologic divides. Road survey analysis will be reported in subsequent articles.

Stream Channel Stability and Composition

Sets of stream channel cross-sections were surveyed at low, middle and upper sections of the main watercourse within each branch of Railroad Gulch (total = 13 cross-sections per branch). Annual surveys allow for the detection of trends in channel incision and aggradation. Pebble count measurements are conducted annually to detect trends in streambed particle-size distribution. Two hundred sediment particles are measured at pre-specified locations across 20 transects placed within the general location of each set of cross-sectional survey locations. Transects span the bankfull width of the channel. Pebble counts span habitat features (pools and riffles) as they are not easily delineated due to the incised nature of the channels and quantity of large wood.

Results and Discussion

Sediment Yield

We report on the first 2 water years (WY) of the study. WY 2014 was an extremely dry year as mean and peak streamflow values were 1/4 to 1/3 those values recorded for WY 2015, respectfully (table 1). Suspended sediment yields in WY 2014 were only 1/17th the size of the WY 2015 yields (table 1). The percent of time that turbidity exceeded 25 nephlometric turbidity units (NTU) doubled in WY 2015 for both basins. The turbidity, autosampling and stage recording equipment were operated without incident in both monitoring periods. Examples of monitoring period discharge, turbidity and SSC from stream samples are shown in figs 3 and 4. In both years, a high percentage of the sediment yield resulted from one or two large storm events. For WY 2014, 77 percent of the measured sediment load was delivered on March 28, 2014. For WY 2015, 40 percent was delivered on February 15th, 2015, and 30 percent on April 13, 2015. During both dry and wet years, discharge, turbidity and sediment yields in control and treatment watersheds closely tracked each other (table 1). This result is not surprising as the basins are quite similar in size, vegetation, geology and land use history. It establishes a firm basis of comparison for evaluating impacts from the subsequent road and harvest activities. Both basins produce extremely high sediment loads and high levels of chronic turbidity. Along with neighboring Tom’s Gulch, these are the highest measured loads in the Elk River watershed. The lower mainstem of the Elk River averages 250 Mg/km^2 annual suspended sediment load.

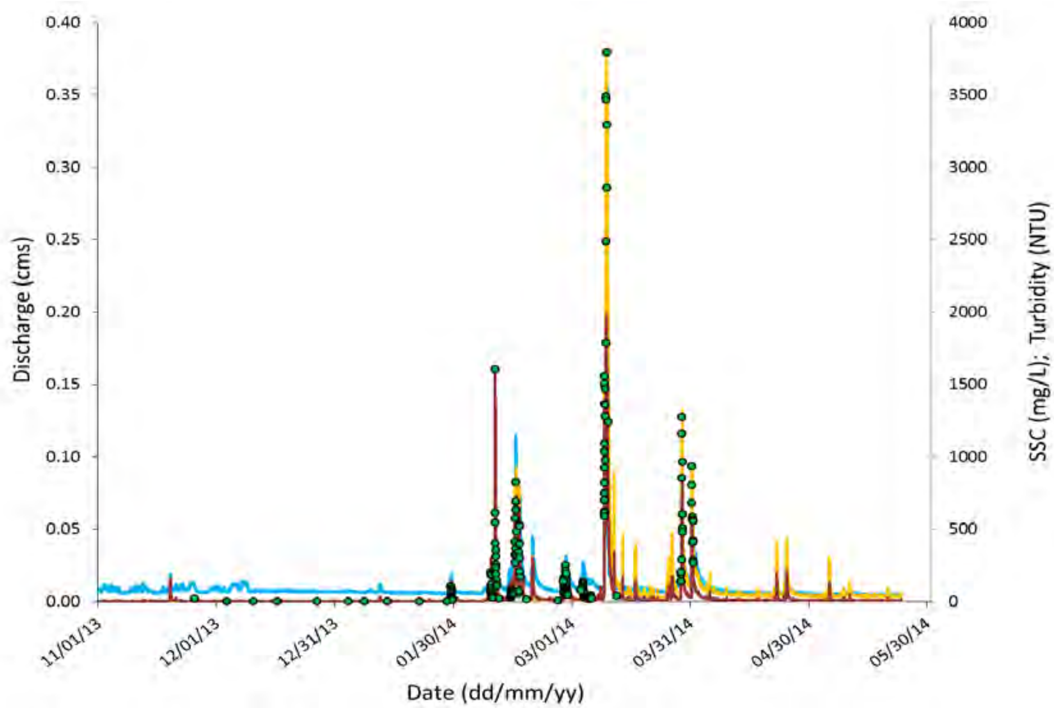

Figure 3—Annual streamflow and turbidity, WY 2014, West Branch Railroad Gulch, Elk River, California. Blue line: discharge (cms); tan line: SSC (mg/L); green circles: SSC measured in collected stream samples.

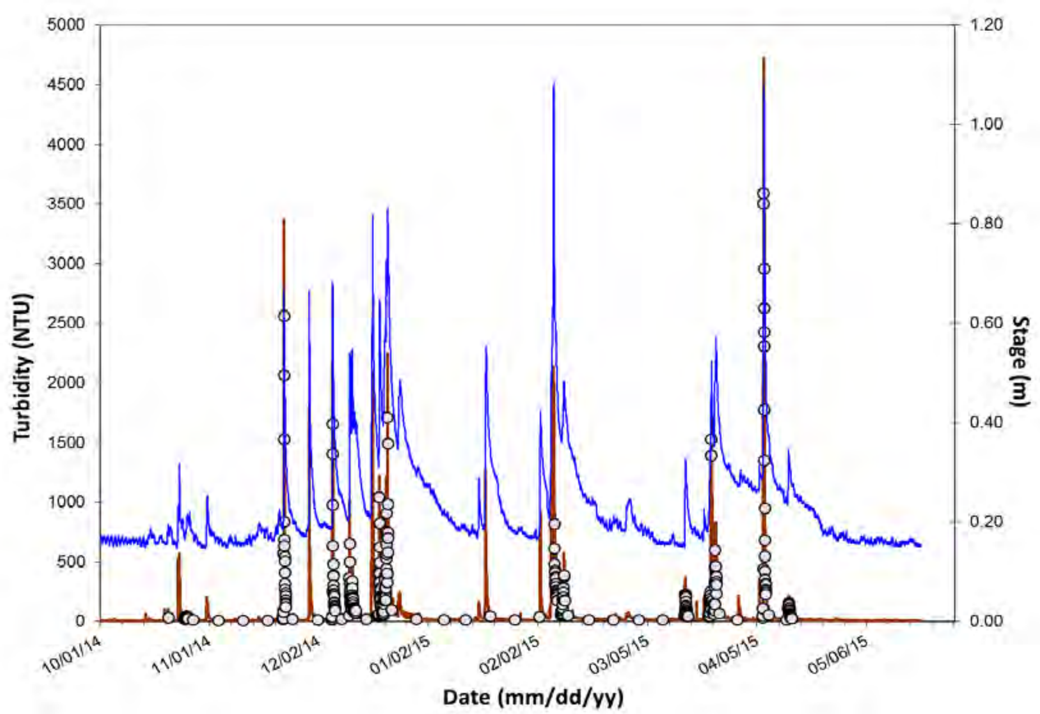

Figure 4—Annual streamflow and turbidity, WY 2015, East Branch Railroad Gulch, Elk River, California. Blue line: stage (m); brown line: turbidity (NTU); white circles: turbidity (NTU) measured in collected stream samples.

Table 1—Summary data for East Branch (treatment) and West Branch (control), Water Year 2014 and 2015, Railroad Gulch, Elk River, California

Time period	basin	Basin area (km^2)	Sediment load (Mg)	Sediment yield (Mg/km^2)	% time turbidity > 25 NTU	Mean discharge (m^3/s)	Peak discharge (m^3/s)	Peak discharge ($m^3/s/km^2$)
Nov. 1, 2013 – May 23, 2014	East Branch	1.28	63	49	16%	0.01	0.42	0.33
	West Branch	1.48	57	38	15%	0.01	0.36	0.24
Oct. 1, 2014 - May 21, 2015	East Branch	1.28	1102	861	28%	0.03	1.6	1.2
	West Branch	1.48	1060	716	30%	0.04	2.0	1.3

Stream and Road Crossing Turbidity

Synoptic samples were collected for five events in WY 2014 and eight events in WY 2015. Turbidities were higher below road crossings than above for numerous locations and events. Differences ranged from zero to 110 NTU. These results would suggest chronic sediment loading from road crossings is occurring. A debris torrent, triggered by the February 2, 2015 storm, came down the tributary stream crossed by site 11 (fig. 1). The debris torrent discharged woody debris and sediment onto and across the crossing. Synoptic samples showed very high turbidity above the crossing resulting from the torrent, ~16,000 NTU on February 6, 2015 and ~26,000 NTU on February 7, 2015. Turbidity prior to the event was consistent with other sites at 40 to 120 NTU. These results support literature findings regarding the importance of roads (McCashion and Rice 1983) and landslides and debris torrents (Kelsey 1980) as sediment sources in this region. As reconstruction and regrading activities took place in summer 2015, results from monitoring of these activities at sites A, B, C, 17, 18, 19, 20, and 22 (fig. 1) will be reported in subsequent articles. The turbidity samples do not allow for quantification of sediment loads as the samples were not concurrently analyzed for SSC. As such they provide an indication of sediment sources within the watershed, but do not inform a quantitative sediment budget.

Streamside Landslides and Bank Erosion

Over 50 independent slides and points of scour were identified between WY 2013 and 2015. Streamside sediment sources are currently dominated by relatively shallow bank slumps and zones of channel scour. Bank slumps are a product of observed stream bank undercutting and not a result of soil saturation from rainfall. These failures are typically discrete landslides that occurred in association with impinging flow in response to flow deflection by large woody debris. Scour tends to occur in areas where streamside slopes are composed of fine grain relatively stiff soils. These erosional features are commonly very shallow (30 to 80 cm) and less than 4.5 m wide. Scouring was observed along straight reaches as well as in reaches with high curvature. Woody debris also appears to play role in the location of these erosion points.

Streamside landslide displacement and delivery volumes for WY 2013, 2014, and 2015 are provide in table 2. Over the study period the West Branch delivery volume per year are nearly double those recorded in the East Branch. This general relationship tracks consistently through the moderate, dry and wet years of WY 2013, 2014 and 2015, respectively. The roughly uniform nature of the pre-treatment data should help in identifying variations (if any) in future landslide rates.

Table 2—Sediment (yd^3) displaced and delivered from streamside landslides and bank erosion, for East Branch (treatment) and West Branch (control), Water Years 2013, 2014, and 2015, Railroad Gulch, Elk River, California

	East Branch		West Branch	
Water year	**Displacement (m^3)**	**Delivery (m^3)**	**Displacement (m^3)**	**Delivery (m^3)**
2013	4.6	4.6	8.7	8.6
2014	0.2	0.2	0.5	0.5
2015	6.7	6.7	15.7	9.8

Landslide Inventory

Landslide activity in the project area appears to be concentrated along roadways and the sidewalls/headwalls of the more deeply incised watercourses. Movement is dominated by landslide mechanisms affiliated with translational failures, therefore the majority of the slides on fig. 2 have been classified as debris slides/flows. Although smaller in number, deep-seated compound failures (earthflows, trans/rotational, etc.) were found to be significantly larger in magnitude compared to their debris slide counterparts. These larger deep-seated events are commonly confined to toeslopes along the valley walls of the study basins.

Seventy-one historically active landslides were identified within the project area (West Branch = 34; East Branch = 37). The size of the landslides is variable, ranging from very small (80 m^2) to covering multiple acres (6500+ m^2). Landslides, in total, cover nearly 18 ha which represents approximately 6 percent of the study area (5 percent West Branch; 7 percent East Branch). About 13 percent of the study area is underlain by landslide deposits that were identified by previous work as dormant-young or older age. Several of these queried landforms have not, to date, been substantiated.

Five active failures have been mapped; two in WY 2014 and three in WY 2015. All of these features are confined to slopes within the control basin with none observed in the East Branch during this time period. Only two of these features delivered sediment to a watercourse, while debris associated with the other slides was captured by roadways or forested hillslopes. Each failure was classified as debris slides/flows with three of them being categorized as reactivations.

The largest of these failures is estimated to have discharge about 40 m^3 into the adjoining watercourse. This slide occurred following periods of heavy rainfall between February 5 and 6, 2015. The rain gage position in the headwater of the West Branch recorded 9 cm of precipitation between the 5th and 6th. Two smaller slides (7 m^3 and 15 m^3) also occurred in response to this same rain event.

Channel Head Movement

The thirty-eight plots installed in WY 2013 were revisited subsequent to the conclusion of WY 2015. The post-WY 2015 surveys found no evidence of channel erosion or headward migration. It was also noted that a majority of the subject watercourse channels (regardless of basin) did not appear to have passed much if any surface flow during WY 2015.

Stream Channel Stability and Composition

The lower (downstream) section of the East Branch appears to have been relatively stable from WY 2014 to WY 2015 with limited scour and channel widening occurring at one cross-section (4) at the top of the lower section. This was the only cross-section throughout the East Branch where notable channel widening was documented. Overall stream channel conditions in both the East and West Branch appear to be heavily dominated by the presence of in-stream wood. When large wood is present in narrow channels with unconsolidated beds and banks it can obstruct and re-direct streamflow during storm events resulting bank under-cutting and bed scour. The middle section of the East Branch appears to have remained largely static. The uppermost (upstream) reaches experienced some thalweg scour. However, the channel is more incised in these sections as the stream transitions from a Class I to a Class II (non-fish bearing) watercourse between cross-sections 9 and 10. Channel conditions at the East Branch gaging station were relatively stable between WY 2014 and WY 2015 with slight thalweg aggradation and scour along the left bank. Stability is important at this location for developing discharge rating curves.

Overall scour was minimal and banks were generally stable within the West Branch. The lower (downstream) to middle sections of the West Branch appear to have been very stable both in terms of thalweg elevation and bank stability from WY 2014 to WY 2015. Some thalweg scour was observed in the lower portion of the most upstream section. This was not observed in the most upper reaches which remained very stable. Like the East Branch the West Branch transitions from Class I to II between cross-sections 9 and 10. This is largely due to the presence of a large landslide with the channel becoming extremely incised above cross-section nine. The West Branch gaging station profile suggests some bank erosion with minimal thalweg scour. This level of erosion is not particularly large (nor unexpected) given the poorly indurated nature of the underlying bedrock and soils.

Pebble counts were first conducted in fall 2015 prior to WY 2016. East Branch pebble count surveys depicted channel reaches that primarily consisted of medium sand (< 2 mm size category). West Branch pebble count surveys depicted channel reaches that were also predominantly made up of coarse sand, however there was a small component (~12 percent) of fine gravels (2 to 4 mm) and coarse gravels (4 to 64 mm).

Channel substrate observed in the subject reach appears to be influenced by the composition of the underlying geology. The geologic map being developed for the project area indicates that the Hookton Formation sequence underlying the West Branch has a larger constituent of coarse sands and fine gravel than that in the East Branch. This is reflected in the larger grain sized recorded in the West Branch than the East Branch.

The cross-sectional survey indicates that streambeds in the two basins are not systematically incising nor aggrading during the time period of this portion of the study. More erosion would be expected in wetter years. We cannot evaluate whether the size composition of the substrate is changing at this time, however the data provides a baseline for future comparisons, establishes the similarity between the control and treatment watersheds before the treatment was applied, and suggests that that underlying geology does appear to directly influence stream bed particle sizes.

Conclusions

The control and treatment basins of Railroad Gulch appear to behave similarly during dry and moderate conditions over the first 2 years of the study, before roads or timber harvest were completed. Discharge, sediment loads and turbidity, volumes of streamside landsliding, and bank and bed scour were consistently similar. Sediment loads on both basins were tenfold higher in the second year of the study (WY 2015) as a result of a large storm event which triggered debris torrents and streamside failures. Both basins produce extremely high sediment loads and turbidity. In both years, a high proportion of total annual sediment loads occurred in one or two storm events. Turbidities below road crossings in both watersheds were elevated from turbidity recorded above, indicating chronic sediment inputs from this source. Historically active landslides covered 6 percent of the watershed area, with five active slides identified, two of which delivered sediment to watercourses. Channel head migration was not observed on either basin during this time period. Minimal bed scour was observed. Pebble counts indicated predominantly medium sand in both basins with a small component of fine gravels on the West Branch. In summary, the data collected establishes a strong basis for detailed comparisons and documentation of any changes resulting from the East Branch timber harvest.

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Fast Response to Fast-Forwarding Nature: Instream Large Wood Habitat Restoration[1]

Cheryl A. Hayhurst[2,3] and William R. Short[2]

Abstract

How quickly and in what way does a channel bed respond when large wood elements are introduced in a way that imitates natural wood loading processes (un-anchored or anchored by burial)? Using a design streamflow threshold for determining the size of key large wood elements, what changes in channel bed and habitat complexity occur after streamflow events above and below the threshold? These are questions we are currently trying to answer with a large wood habitat enhancement project on the East Branch of Soquel Creek within Soquel Demonstration State Forest in Santa Cruz County, California. This project also aims to address a lack of instream habitat complexity that was identified in the National Marine Fisheries Service 2012 Central California Coast Coho Salmon Recovery Plan.

Large wood elements were placed in four project reaches (sites) along East Branch Soquel Creek in 2012 and 2013. These large wood elements consist of a combination of large key pieces (whole or nearly whole redwood trees with rootwads), log vanes with rootwad covers, and loosely racked wood structures. The large key elements were selected based on a size calculated to remain meta-stable through a 5-year return interval streamflow event and consist of single or multi-stem redwood trees with rootwads. The rootwad diameters range from 2.7 m to 3.81 m (9 ft to 12.5 ft) and the stems range from 0.8 m (30 inches) diameter at breast height (DBH; 1.37 m) to 1.3 m (51inches) DBH (largest stems on multi-stemmed structures). In all, 45 stems and 10 rootwads were introduced.

Monitoring observations have been conducted on four separate occasions at Site 1 (installed in 2012) and three times for the remaining three sites (2, 4, and 5) installed in 2013. Site 1 large wood elements experienced an event approximating the 7-year return interval the first winter after installation, which rotated the elements in place. Channel changes occurred the first winter after installation of sites 2, 4, and 5 in response to streamflow events below the design threshold. Thalweg profiles indicate increased complexity in the form of pool formation and localized aggradation and scour through the project reaches. On March 5 and 6, 2016, these structures experienced an approximately 10-year streamflow event which visibly affected the geometry and positioning of the structures along with generally increasing the number, and in some cases the size, of pools. The extent of thalweg changes and the position and orientation of large wood elements will be included as part of the physical monitoring of the four reaches that continues for 5 years after installation.

Keywords: channel morphology, Coho salmon, geomorphology, large wood, salmonids, stream restoration, wildlife habitat

Introduction

At the request of the California Department of Forestry and Fire Protection (CAL FIRE), staff of the California Geological Survey (CGS) completed the design and provided on-site technical support and direction for a Large Woody Debris and Habitat Complexity project. The project design included a large wood restoration literature review, geomorphic assessment, and hydrologic analysis. The installation included 10, large wood (LW) elements in four project reaches (referred to as Sites 1, 2, 4, and 5) along a 1.1 km (0.7 mile) stretch of the East Branch of Soquel Creek, within the Soquel Demonstration State Forest (SDSF) (fig.1). The SDSF is a 1,085 ha (2,681 ac) forest managed by CAL FIRE. Site 1 consists of three elements (1a, 1b, and 1c) within an approximately 91 m (300 ft)

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[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.

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reach and was completed in September of 2012. Sites 2 and 4 also consist of three elements each within an approximately 91 m (300 ft) reach, while Site 5 consists of one element in an approximately 30.5 m (100 ft) reach. Sites 2, 4, and 5 were constructed in August and September of 2013. Site 3 was not constructed due to funding constraints.

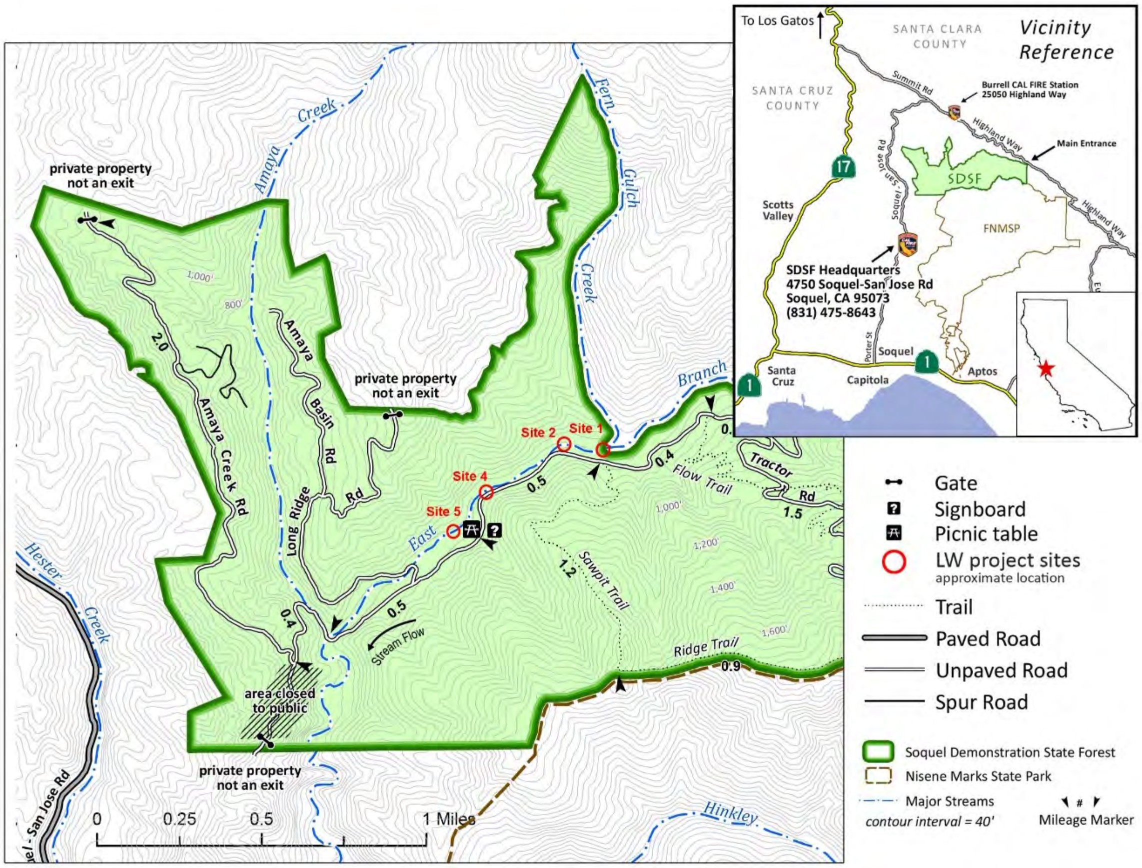

Figure 1—Site location map showing the LW sites within the SDSF boundary.

Soquel Creek was identified by the National Oceanic and Atmospheric (NOAA) Fisheries Service as a “focus” watershed in their recovery plan for the Evolutionarily Significant Unit of Central California Coast Coho salmon, on the basis of low amounts of large wood being a stressor for the recovery of Coho salmon in Soquel Creek (National Marine Fisheries Service 2012).

The SDSF’s Large Woody Debris and Habitat Complexity Project was undertaken in an effort to help address the shortage of LW within Soquel Creek, and to increase overall stream quality from a biological standpoint. Lack of wood can cause simplification of channel characteristics and the wood emplacement is meant to increase channel complexity such as increasing the number and/or depth of pools, storing gravel, and providing high flow refugia areas.

The underlying concept of the project is to emulate natural LW features as closely as possible, including dropping of bank-side trees with rootwad, placement of log vanes and log clusters, and avoiding the use of cables, bolts, and other artificial means of achieving LW stability. The approach to the design process involved the basic steps of 1) geomorphic mapping of potential stream reaches, 2) selection of project reaches, 3) selection of design flow, 4) sizing LW elements. and 5) configuration of LW elements. A summary of the project design and installation details are described below. Additional details are described in "as-built" reports prepared for Site 1[4] and Sites 2, 4, and 5[5].

Methods and Materials

Hydrology and Large Wood Design Considerations

Initial design surveys (including longitudinal profiles, cross-sections, bankfull geometry, and bank heights) indicated that for East Branch Soquel Creek, bankfull channel width varies from 14 m (45 ft) to 17 m (55 ft), bankfull maximum depth varies from 0.6 m (2.2 ft) to 0.8 m (2.6 ft), and bankfull mean depth varies from 0.2 m (0.8 ft) to 0.4 m (1.2 ft). East Branch Soquel Creek was determined to have a bank height ratio (BHR) that is typically on the order of 1.4 and was found generally incised. Bank height ratio is defined as the height of the lowest bank divided by the maximum bankfull depth.

In order to achieve the desired stability for the LW elements, a key piece of information necessary for the project design and sizing of the LW elements was the design streamflow threshold. The design flow was determined in consultation with fisheries biologists from the California Department of Fish and Wildlife, National Marine Fisheries Service, and private consultants, who were asked to provide bounding values for desired LW longevity. A minimum threshold time frame of 5 years was agreed upon for LW features to have a significant biological benefit. That opinion provided a minimum or base-line design flow, i.e., a flow with a recurrence interval of at least 5 years.

The 5-year return interval design flow was calculated by annual-flood frequency analysis using a flow transference method (Waananen and Crippen 1977) from United States Geological Survey (USGS) gage No. 11160000 (USGS 2016) on Soquel Creek located downstream in the town of Soquel, California. A correction factor developed by a local hydrologist for the East Branch of Soquel Creek was used to refine the analysis (B. Kreager, personal communication). A flow of 28 cms (990 cfs) and a stage height of 1.5 m (5 ft) were used in the design.

To form stable LW accumulations Key Logs (typically a complete tree with rootwad attached) were sized to withstand the forces generated at the design flow by conducting a stability analysis that takes into account various forces due to buoyancy, gravity, and flow, among others. The stability analysis indicated that a Key Log of a minimum 24 m (80 ft) length, 1 m (40 inches) DBH, and a 3 m (10 ft) diameter rootwad would remain meta-stable through a 5-year return interval event. Single-stem trees of sufficient mass to be stable through a 5-year streamflow event, were not available. However, several complexes consisting of a single rootwad with two to four stems were available. These clumps were of sufficient height and mass to conform to the LW design parameters.

Large Wood Elements

The LW features installed at SDSF are of two primary types, mobile and anchored. Mobile wood is comprised of features that rely solely on the mass and shape (bole plus rootwad) to provide stability; anchored wood features rely on secondary elements, in this case boulders and burial, to achieve stability. LW elements involving mobile wood were Key Logs dropped into the creek, with rootwads facing both up- and down-stream (Sites 1a, 1b, 1c, and 2c, see fig. 2), and loose-stacked arrays of logs referred to as log clusters (Sites 2a, 4a, and 5). These two features were used in conjunction, the Key Logs having sufficient mass to be stable at the design flows and the log clusters sized to slowly disaggregate and move downstream. This allows the desegregated logs to become entangled with the downstream key logs, thus emulating the natural process of log jam formation. Two types of anchored

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[4] California Geological Survey. 2013. Soquel Creek LWD Project – Site 1 As-Built Report. Unpublished memorandum to Angela Bernheisel, Forest Manager, Soquel Demonstration State Forest, California Department of Forestry and Fire Protection; dated January 18, 2013.

[5] California Geological Survey. 2014. Soquel Creek LWD Project – Site 2, 4, & 5 As-Built Report. Unpublished memorandum to Angela Bernheisel, Forest Manager. Soquel Demonstration State Forest, California Department of Forestry and Fire Protection; dated April 4, 2014.

wood elements were employed. The first is a variant of the dropped whole tree. In places where access was limited, a single large bole with rootwad attached was placed downstream of a log cluster (Site 4b). In order to achieve the requisite stability, 3.7 m to 4.6 m (12 ft to 15 ft) of the bole was buried in the stream bed with large rock (ballast) being used as backfill. The other anchored structure was a simple log vane, a large log that is buried in the bank and partially buried in the channel (Sites 2b and 4c). The logs project upstream from the bank at approximately a 30 degree angle and plunges at approximately 5 degrees. The function of this feature is to reduce pressure and erosive forces on a stream bank by locally reducing stream gradient and thus encourage deposition of bed load leading to the formation of a lateral bar. A secondary benefit is that typically downstream of the vane a pool will form. To enhance the biological value of the pool, a rootwad cover log was added to the vane. Like the vane, the rootwad is buried in the bank. In addition to the primary LW elements, broken or trimmed tops and alders or other smaller trees that came down with the falling of the primary LW elements were incorporated into the structures.

Figure 2—Site 1b Key Log.

All LW elements consist of redwood (*Sequoia sempervirens* (D. Don) Endl.) or Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) trees either from the adjacent banks or from a recent timber harvest within SDSF. Rock for ballast was derived onsite, generally within the reach vicinity. The total amount of wood introduced into Soquel Creek at the four sites includes 45 stems and 10 rootwads. The stems range from 0.25 m (10 inches) to 1.37 m (54 inches) diameter and 7.9 m (26 ft) to 35.4 m (116 ft) long. The rootwads range from 2.6 m (8.5 ft) to 3.8 m (12.5 ft) in diameter. The total calculated volume of wood added is 326.4 cubic meters (m^3) (11,528 cubic feet [ft^3]). The total amount of LW introduced into Soquel Creek for all four project reaches calculates to 11.7 m^3/30.5 m (412 ft^3/100 ft) (LW volume versus channel length) or 0.0076 m^3/m^2 (0.27 ft^3/ft^2) (LW volume versus channel area). This is similar to amounts described in nine studies of disturbed but recovering watersheds of the Pacific Northwest (Benda et al. 2002, Benda et al. 2003, Fausch and Northcote 1992, Faustini and Jones 2003, Keller and MacDonald 1983, Long 1987, McHenry et al. 1998, Swanson et al. 1987, Wooster and Hilton 2004) where median LW values of about 13.3 m^3/30.5 m (486 ft^3/100 ft) or 0.0079 m^3/m^2 (0.28 ft^3/ft^2) of channel are reported.

Monitoring

Annual monitoring primarily consists of thalweg surveys at each site to document changes in channel morphology. The thalweg surveys are completed using a rod and level survey at each site to record distance and relative elevation measurements. Thalweg surveys conducted at Site 1 include a post-installation survey completed in November 14-15, 2012, and monitoring surveys conducted February 22, 2013, December 22-23, 2014, December 1-2, 2015, and June 21-23, 2016. Thalweg surveys represented for Sites 2, 4, and 5 include the post-installation surveys completed October 22-23, 2013, and monitoring surveys noted above conducted in 2014, 2015, and 2016. A baseline thalweg survey was conducted February 4, 2011. Streamflow data from the downstream USGS gage were also used to summarize peak streamflows experienced at the project sites between the annual monitoring events.

Results

LW Response to Hydrologic Events

2012/2013 Winter Rains

Two significant rain events occurred producing peak stream flows in Soquel Creek of roughly 26.9 cms (950 cfs), an approximately 4-year return interval, and 35.7 cms (1260 cfs), an approximately 7-year return interval, on December 2, 2012 and December 23, 2012, respectively. Since Site 1 was installed in September 2012, the LW at this location experienced these 2012/2013 peak flows. These events did not move the Site 1 rootwads from their installation point, however the stems attached to the rootwads rotated from their original position oriented across the creek to an orientation more in line with the flow direction. Several of the smaller wood pieces incorporated into the large wood elements, such as broken tops, did dislodge and either moved downstream out of the reach, or readjusted positions within the reach. Figure 3 shows the change in stem orientation at each of the Site 1 structures.

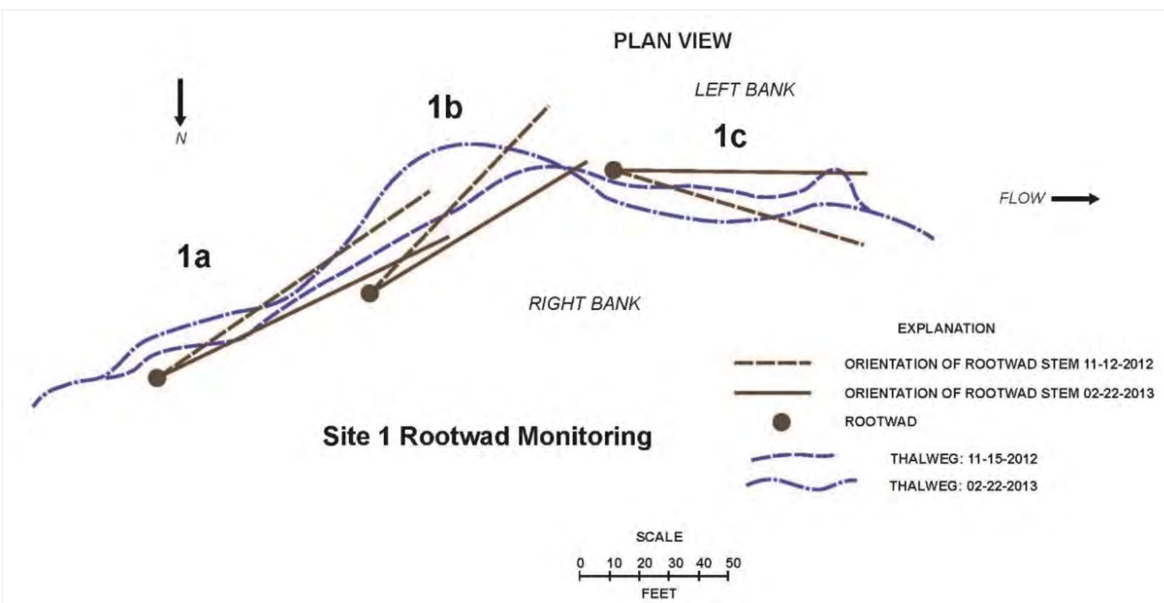

Figure 3—Rotation of Site 1 Key Logs after approximately 7-year return interval streamflow event.

2013/2014 and 2014/2015 Winter Rains

The winter season of 2013/2014 and the early part of the 2014/2015 winter season (though December 20, 2014) did not produce significant storm events. The largest event over this period occurred on

December 12, 2014 producing a peak flow of 6.26 cms (221 cfs), approximately a 1.4-year return interval streamflow, in the vicinity of the project sites. The minor streamflow events were not significant enough to alter the LW elements substantially from their initial placement location.

2014/2015 and 2015/2016 Winter Rains

The winter season of 2014/2015 and the early part of the 2015/2016 winter season (through December 2, 2015) did not produce significant streamflow events. The largest event over this period occurred on February 8, 2015, with a peak flow of 9.8 cms (345 cfs), an approximately 1.7-year return interval streamflow, in the vicinity of the project sites. The minor streamflow events were not significant enough to alter the LW elements substantially from their initial placement location, with the exception of rotating several of the logs in the log cluster sites at Sites 2a, 4a, and 5, and the mobilization of one of the logs from Site 2a. The mobilized log from 2a was entrained downstream in the Site 2c structure.

2015/2016 Winter Rains (Dec 3, 2015 – July 1, 2016)

A significant streamflow event occurred on March 5, 2016, producing a peak flow of 39.8 cms (1405 cfs), an approximately 10-year return interval event. This event had a more substantial impact on the LW project sites. The Key Logs at 1a and 1b both moved downstream approximately 21.8 m (71.5 ft) and 48.8 m (160 ft), respectively, with 1b now farther downstream than the 1c Key Log, though all three elements have remained within the project reach vicinity. At the Site 2a log cluster only one of four logs remains. The Site 2c Key Log moved approximately 113 m (370 ft) downstream of the project reach to create a new log jam where it hung up. At the 4a log cluster, four of nine logs mobilized downstream and have formed a new log jam just downstream of Site 4c. The 4b anchored rootwad backstop mobilized and the rootwad hung up on the 4c vane log. The log cluster at Site 5 had several pieces rotate and has entrained additional wood, including a large stump that appears to have eroded out of the right bank just upstream of the structure.

Channel Bed Response

Site 1

Through the Site 1 reach, baseline 2011 and 2012 post-installation surveys show a generally uniform channel profile with only one incipient pool present after the installation of the three Key Log structures. An incipient pool, for the purpose of the LW monitoring, is defined as a bed roughness element less than 0.3 m (1 ft) in residual depth, meaning the depth calculated from the low point of the roughness element to the top of the next downstream riffle crest or high point, irrespective of water depth. A pool is defined as a bed roughness element having a residual depth of 0.3 m (1 ft) and greater. During the winter following installation, Site 1 experienced an approximately 7-year return interval streamflow event that exceeded the design peak streamflow for the Key Logs. As noted above, the Key Log structures rotated in the streamflow event, but remained in their general installation locations. The channel response included the formation of two pools and three incipient pools for a total of four incipient pools. Based on the survey data localized aggradation occurred just upstream of each of the Key Log sites. During the next 2 years (2014 and 2015) the largest streamflow events were approximately 1.4-year and 1.7-year return intervals. Even with the minimal streamflow events, the channel profile remained dynamic with localized aggradation and scour. In comparison to the other sites, Site 1 was installed prior to the 7-year event in 2012, which may account for the increased aggradation observed at Site 1 relative to the other sites, which were installed the following summer after that event. In March 2016, Site 1 experienced an approximately 10-year return interval streamflow event. The Key Log structures at 1a and 1b both mobilized downstream a short distance, though they remained within the general project reach area. Site 1c remained in its previous position. In addition to the relocation of Key Log structures at Sites 1a and 1b, there are now four pools and seven incipient pools present in the project reach. These channel changes are represented in fig. 4.

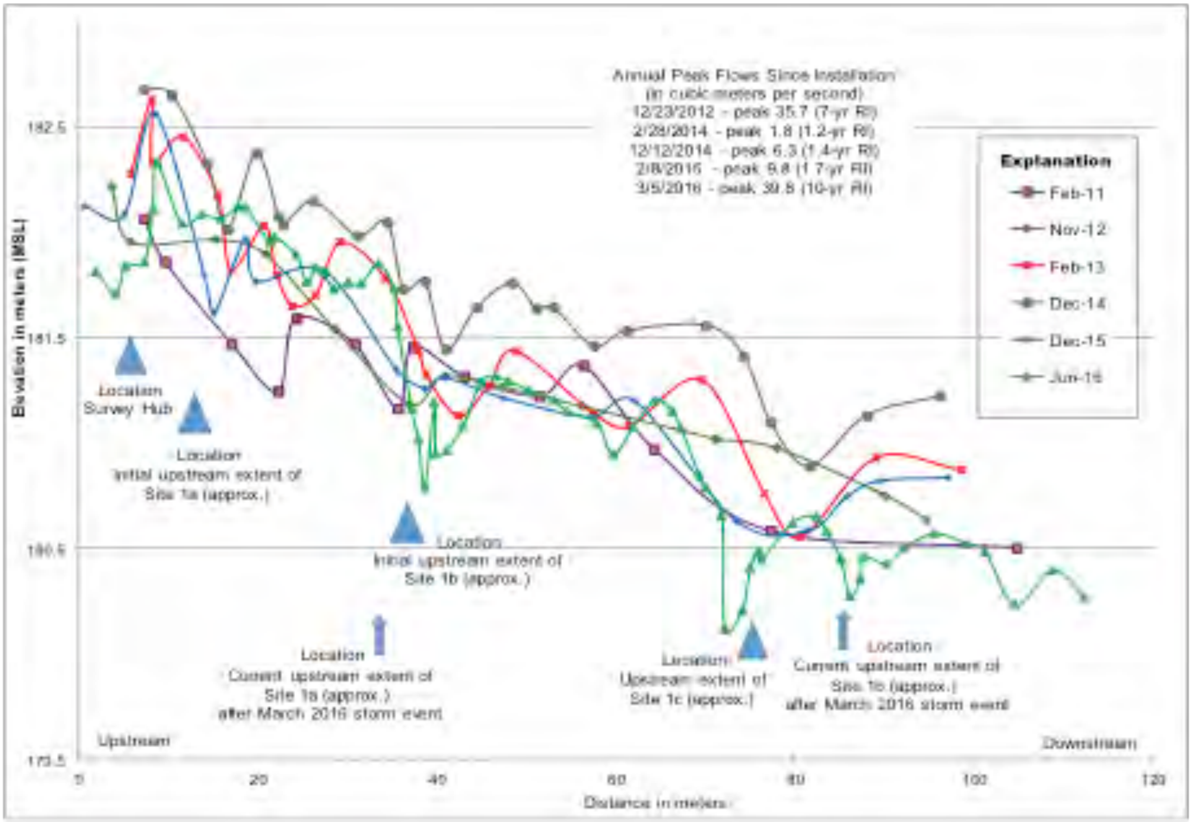

Figure 4—Site 1 thalweg surveys.

Site 2

Site 2 has been one of the more dynamic sites since installation. Although the profiles differ some, both the 2011 baseline survey and the 2013 post-installation survey show a fairly uniform channel profile with up to three incipient pools. In the 2 years following the LW installation (peak flows of 1.4- and 1.7-year return intervals), the channel elevation immediately upstream of the Site 2c Key Log structure increased by approximately 0.76 m (2.5 ft) with aggraded stream gravels. The aggraded material extended upstream from the 2c structure a distance of approximately 61 m (200 ft). At Site 2b a 0.7 m (2.3 ft) deep scour pool was measured at the location of the rootwad cover for the vane log. In that 2-year timeframe, three pools were scoured and the incipient pools varied in number from three to five. In 2016, after the 10-year streamflow event, the entire 2c Key Log structure mobilized downstream out of the project reach. With the 2c structure no longer present to retain bed material, up to approximately 1.2 m (4 ft) of vertical channel scour occurred upstream of the initial 2c location. The scour response was most significant between the 2b and 2c structures, but did continue upstream through the 2a structure. The resulting thalweg profile contained nine incipient pools and no pools. These channel changes are represented in fig. 5.

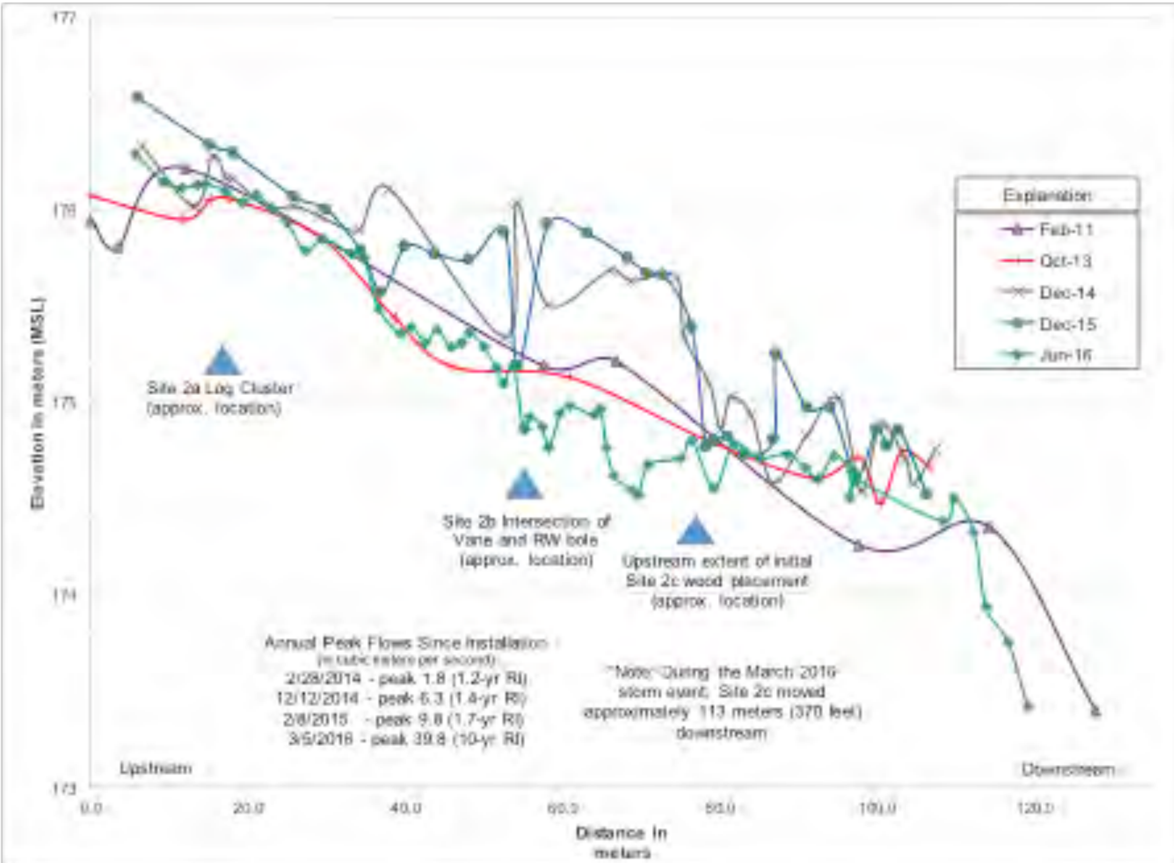

Figure 5—Site 2 thalweg surveys.

Site 4

The 2011 baseline survey through Site 4 shows a relatively uniform channel profile with two incipient pools and one large pool just downstream of the site. The post-installation survey in 2013 reflects a similar channel condition to 2011 with four incipient pools and the large pool downstream. Thalweg changes over the next 2 years (peak flows of 1.4- and 1.7-year return intervals) were generally modest, though incipient pools increase to six in 2014 and remained through 2015. Thalweg complexity also appears to increase during this time with localized scour and aggradation throughout the reach on the order of approximately 0.2 m to 0.3 m (0.5 ft to 1 ft) through most of the reach distance. During the 10-year streamflow event the anchored LW structure at 4b dislodged and traveled downstream where its rootwad hung up on the vane log at the 4c structure. This is where the most significant channel changes occurred. Vertical channel scour of approximately 0.8 m (2.5 ft) occurred at the 4c structure downstream of the vane log and the large downstream pool that had been relatively stable since at least 2011 filled with sediment. Overall, the 2016 survey shows similar channel complexity upstream of the 4c structure and the presence of five incipient pools and three pools within the reach. These channel changes are represented in fig. 6.

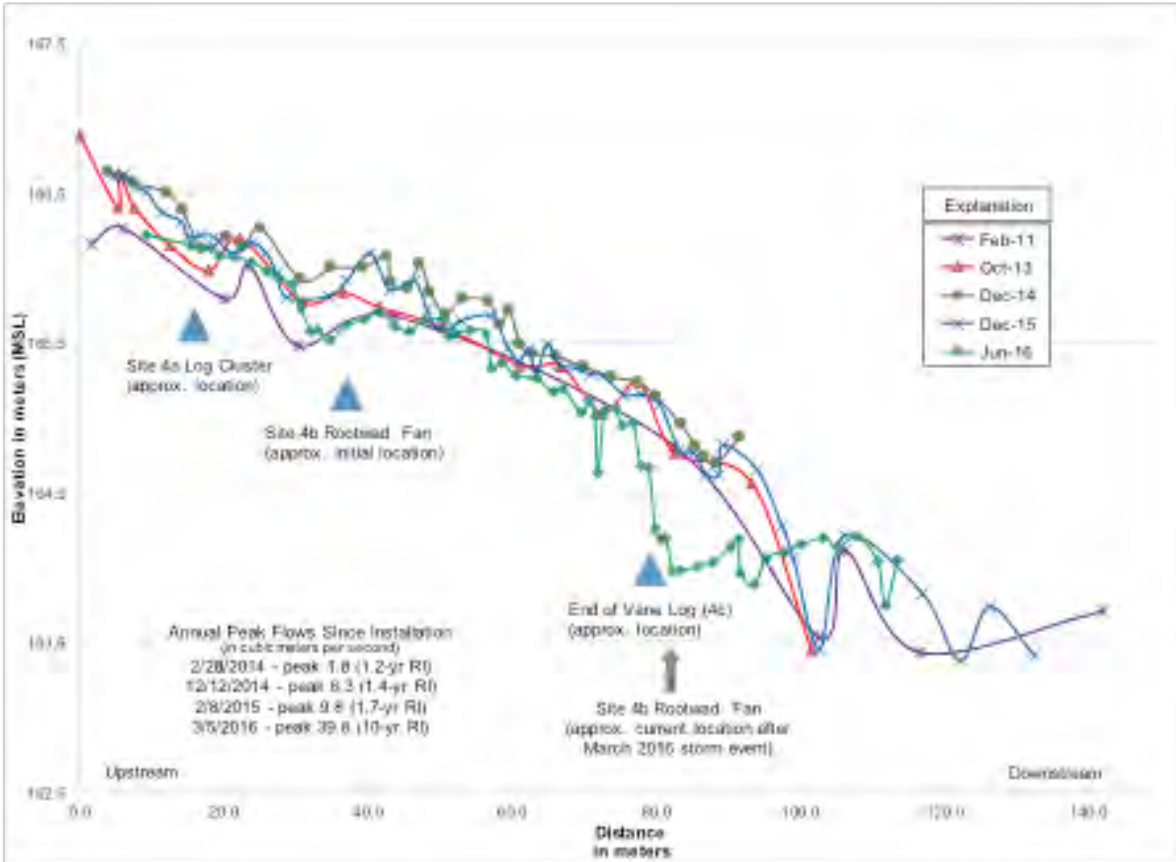

Figure 6—Site 4 thalweg surveys.

Site 5

The baseline 2011 survey shows a uniform channel profile through Site 5, however few data points were collected at that time. The post-installation survey in 2013 shows the presence of two incipient pools. Over the next 2 years (2014 and 2015), the number of incipient pools grew to three and then six, respectively, through the project site and channel elevation appears to have locally increased (via aggradation) approximately 0.2 m to 0.3 m (0.5 ft to 1 ft) over a distance of more than 7.6 m (25 ft) upstream of the log cluster. These channel changes occurred with relatively low annual peak flows corresponding to approximately 1.4-year and 1.7-year return interval events. The 2016 survey reflects significant changes that occurred in response to the 10-year streamflow event and the formation of a log jam at the LW structure. Five incipient pools remain in the reach and a large pool with a residual depth of 0.97 m (3.17 ft) was scoured at the log jam. Additionally, channel aggradation of an average of approximately 0.30 m (1 ft) occurred upstream of the new log jam. The channel changes are represented in fig. 7.

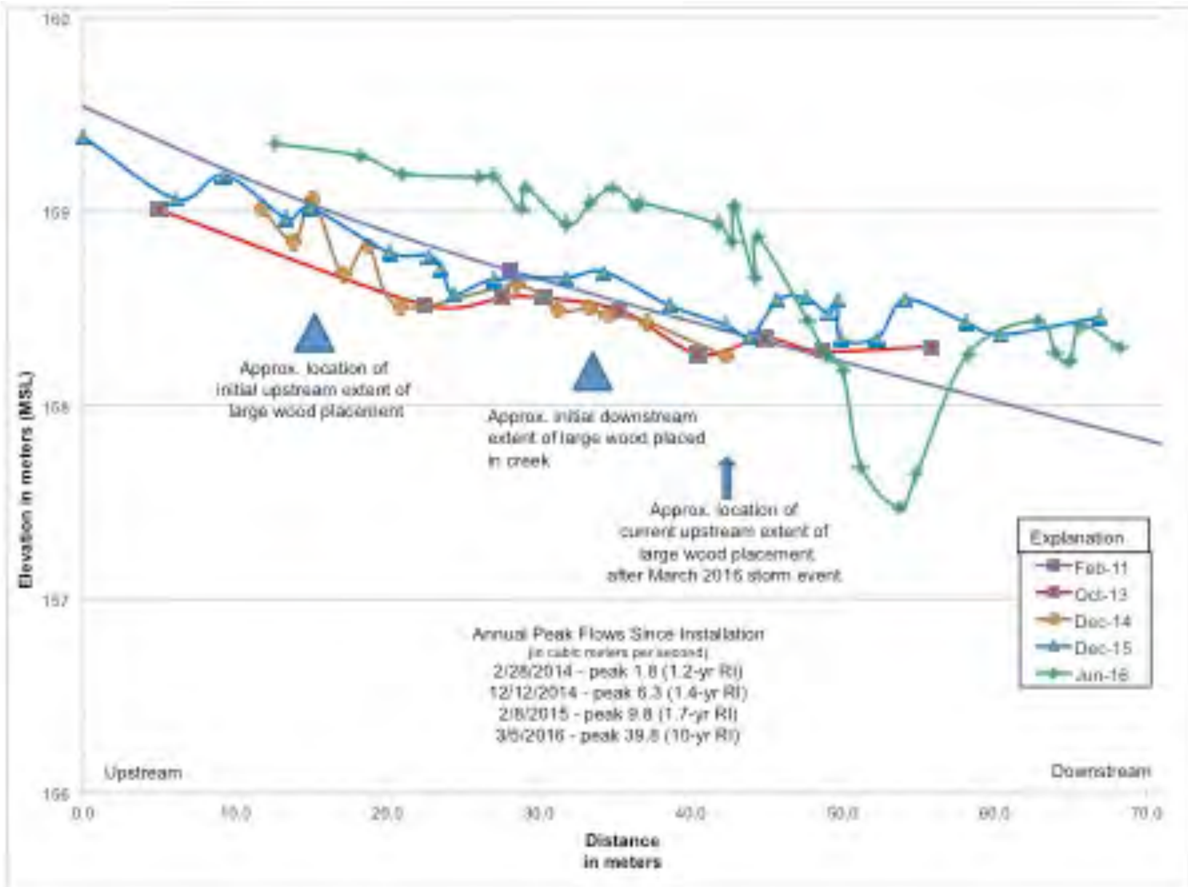

Figure 7—Site 5 thalweg surveys.

Discussion

Instream habitat restoration at Soquel Creek utilized Key Logs in LW structures sized to remain meta-stable through a 5-year streamflow event to provide a habitat benefit and increase geomorphic complexity within each reach for a period of time (5 years) deemed significant by the fisheries biologists. In the 4 years since installation, the Site 1 structures have experienced an approximately 7-year recurrence interval streamflow event and a 10-year recurrence interval streamflow event (fig. 8). Sites 2, 4, and 5 have been in place for 3 years and experienced a 10-year streamflow event. With the exception of one of the Key Log structures, Site 2c, all of the Key Log structures remained within the general project reach vicinity and remain interacting with the stream channel. Though Site 2c mobilized downstream out of the project reach, it has lodged in a new location and will continue to add a positive benefit to the overall health and functionality of the stream. Within our monitoring reaches at Soquel Creek, we demonstrated that the channel profiles respond relatively quickly to constructed LW structures that imitate natural wood loading. All four of the sites showed increased thalweg profile complexity within the first year of installation. These changes included the development of incipient pools (less than 0.3 m [1 ft] deep) and pools (0.3 m [1 ft] and greater depth, summarized in table 1), localized aggradation and scour (particularly aggradation upstream of the wood structures), and the formation of lateral gravel bars (though not represented in the thalweg surveys; see fig. 9). What was particularly interesting is the increase in complexity at Sites 2, 4, and 5 occurred within the first year or two after installation during an extended drought characterized by low rainfall and small peak flows (1.4-year and 1.7-year events).

Figure 8—Site 1 showing 10-year flow features (March 17, 2016).

Table 1—Summary of the number of pools and pool depths at each site

Site 1	Site 2	Site 4	Site 5
November 2012: 1 incipient pool (0.13 m [0.43 ft] deep)	N/A	N/A	N/A
February 2013: 2 pools (0.31 m and 0.37 m [1.01 and 1.23 ft] deep), 4 incipient pools (0.13 m to 0.30 m [0.42 to 0.98 ft] deep)	October 2013: 3 incipient pools (0.11 m to 0.19 m [0.36 to 0.61 ft] deep)	September 2013: 4 incipient pools (0.06 m to 0.23 m [0.20 to 0.76 ft] deep)	September 2013: 2 incipient pools (0.04 m and 0.08 m [0.13 and 0.27 ft] deep)
December 2014: 2 pools (0.31 m and 0.36 m [1.03 and 1.18 ft] deep), 5 incipient pools (0.03 m to 0.24 m [0.11 to 0.79 ft] deep)	December 2014: 3 pools (0.35 m to 0.69 m [1.16 to 2.28 ft] deep), 5 incipient pools (0.16 m to 0.26 m [0.54 to 0.86 ft] deep)	December 2014: 6 incipient pools (0.10 m to 0.18 m [0.32 to 0.58 ft] deep)	December 2014: 3 incipient pools (0.11 m to 0.22 m [0.36 to 0.71 ft] deep)
December 2015: 1 pools (0.35 m [1.15 ft] deep), 4 incipient pools (0.05 m to 0.24 m [0.16 to 0.78 ft] deep)	December 2015: 3 pools (0.31 m to 0.73 m [1.03 to 2.41 ft] deep), 3 incipient pools (0.09 m to 0.24 m [0.28 to 0.78 ft] deep)	December 2015: 6 incipient pools (0.06 m to 0.30 m [0.19 to 0.97 ft] deep)	December 2015: 6 incipient pools (0.06 m to 0.20 m [0.19 to 0.66 ft] deep)
June 2016: 4 pools (0.41 m to 0.63 m [1.33 to 2.06 ft] deep), 7 incipient pools (0.06 m to 0.30 m [0.21 to 0.97 ft] deep)	June 2016: 0 pools, 9 incipient pools (0.06 m to 0.27 m [0.20 to 0.87 ft] deep)	June 2016: 3 pools (0.31 m to 0.42 m [1.02 to 1.39 ft] deep), 5 incipient pools (0.07 m to 0.22 m [0.23 to 0.72 ft] deep)	June 2016: 1 pool (0.91 m [3.17 ft] deep), 5 incipient pools (0.11 m to 0.21 m [0.37 to 0.70 ft] deep)

Figure 9—Site 2b gravel bar (December 23, 2014).

Acknowledgments

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Hydrologic Influences on Stream Temperatures for Little Creek and Scotts Creek, Santa Cruz County, California[1]

Justin M. Louen[2] and Christopher G. Surfleet[2]

Stream temperature impacts have resulted in increased restrictions on land management, such as timber harvest and riparian restoration, creating considerable uncertainty for future planning and management of redwood (*Sequoia sempervirens* (D.Don) Endl.) forestlands. Challenges remain in the assessment of downstream cumulative stream temperature effects given the complexity of stream temperature dynamics. The goal of this research is to identify processes and measurements that can aid the assessment of risk of downstream temperature heating.

Stream temperature, hydrologic, climatic, and channel morphological data were collected on two, approximately 800 m stream reaches on Little Creek and Scotts Creek located in mixed coast redwood and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests of Santa Cruz County, California. Spatially and temporally explicit stream temperature measurements were collected using distributed temperature sensing (fig. 1). A fluorescent dye tracer was used to gather information on summer streamflow including the quantification of residence time and hyporheic exchange. A heat budget approach, adopted from a study by Moore et al. (2005), was used to quantify individual heat flux components and to examine the processes of stream heating and cooling. Downstream temperature change over varying distances through each study site were statistically compared against averaged heat budget components over this respective distance. Average daily maximum stream temperatures were statistically compared against measured physical channel characteristics at each 25 m location throughout the study reaches.

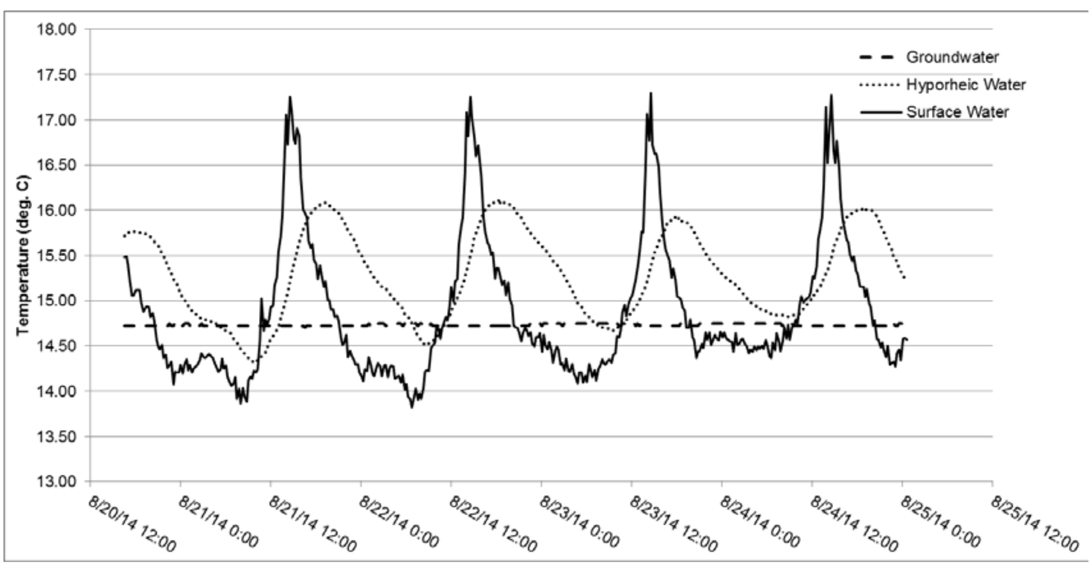

Figure 1—Measured surface, hyporheic, and groundwater temperatures (ºC) at location 390 m on Little Creek from August 20-25, 2014.

[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.
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Potential downstream effects were modeled by implementing hypothetical scenarios in which stream shading was reduced by 50 percent and 100 percent, via canopy reduction from timber harvest or riparian restoration, along the upper 300 m and 200 m of the Little Creek and Scotts Creek study sites respectively. These figures demonstrate the potential effects from future habitat restoration work. An additional analysis was performed by implementing hypothetical scenarios in which groundwater inflows would increase following near-stream vegetation removal, presumably from decreased evapotranspiration, along the upper 300 m and 200 m of the Little Creek and Scotts Creek study sites respectively. Stream temperatures were modeled with streamflow increases of 10 percent, 20 percent, 30 percent, and 50 percent under both the 50 percent and 100 percent shade reduction scenarios. Mean observed groundwater temperature of 14.8 ºC was assumed to be representative of groundwater inflows for modeled temperature mixing with surface water. Further information on the modeling approach taken in this study, including key assumptions, can be referenced in the thesis by Louen (2016).

Average daily maximum stream temperatures for each 5-day measurement period, were found to vary spatially (ranging from 14.95 to 17.91 ºC on Little Creek and 16.52 to 18.67 ºC on Scotts Creek) due to a variety of observed and measured cooling and heating mechanisms occurring throughout both study sites. Comparisons of observed and modeled temperatures between the two sites and the relative influences of individual heat budget components indicated that the magnitude and spatial frequency of subsurface-surface water interactions, along with incoming net radiation, played a substantial role in how heat was transferred through each system.

Relative influences of heat budget variables on downstream temperature change were found to vary between the two study sites with downstream distance (table 1). The measurement and evaluation of a stream's hydrologic characteristics, stream shading, and aspect ratio were statistically significant measurements ($\alpha < .05$) associated with downstream temperature change for Little Creek. Only weak statistical relationships were found for Scotts Creek. Weak relationships may have been attributed to very low streamflow due to drought conditions creating longer water residence times on Scotts Creek. Regression analysis indicated that hyporheic energy fluxes were negatively associated with increases in stream temperatures on Little Creek.

Table 1—Statistically significant heat budget variables on Little Creek and Scotts Creek (variables with p-value > .05 excluded) including net radiation (Nr), streambed conduction (Qc), latent heat exchange (Qe), sensible heat (Qh), and hyporheic flux (Qhyp) per downstream distance (m) evaluated

Study Site	Significant variables and corresponding β term over distance evaluated (m)			
		100	300	800
Little Creek	Nr			0.004
	Qc			
	Qe	-0.011	0.020	0.039
	Qh			
	Qhyp	-0.012	-0.041	-0.088
		50	200	825
Scotts Creek	Nr	0.004	0.029	0.032
	Qc	0.127		0.043
	Qe	0.004	0.005	0.009
	Qh			
	Qhyp	-0.005	0.003	

Observed hyporheic exchange rates and the prevalence of subsurface water mixing within the Little Creek reach appears to be associated with channel morphological characteristics of the stream which include steeper channel gradients and the predominant step-pool and cascade configuration observed in the study site. The step-pool and cascade configuration of Little Creek promotes concentrated areas of downwelling and subsurface water mixing, particularly in riffles downstream of pools as indicated by the observed statistical relationship between higher aspect ratios (width to depth ratios) and lower daily maximum temperatures. The spatial occurrences of these subsurface water interactions in Little Creek are consistent with previous research on streambed topography influences on surface-subsurface water interactions (Burkholder et al. 2008, Harvey and Bencala 1993, Moore et al. 2005). Departures from the steeper, step-pool configuration observed at Scotts Creek- with shallower channel gradients and more frequent occurrences of pools and glides compared to riffles- more than likely influenced the higher residence times of water and lower rates of hyphoreic exchange, particularly in the lower 625 to 825 m reach.

Heat budget modelling results indicated temperature increases in both study sites downstream of the hypothetical riparian canopy removal. Modeled downstream average daily maximum stream temperatures under the 100 percent shade reduction scenarios for both Little Creek and Scotts Creek resulted in increases of 1.6 and 3.1 ºC respectively (table 2). Potential increases in groundwater inflows following hypothetical canopy reduction scenarios reduced the effect of downstream temperature increases with greater reductions in stream temperature cooling with increased groundwater inputs (table 2).

Table 2—Modeled average maximum stream temperature (°C) response to groundwater inflow increases per shade reduction scenario on Little Creek and Scotts Creek

Study site	Shade scenario	Average max. temp. (deg. C)	Groundwater increase 10%	20%	30%	50%
Little Creek	Current condition (92% shade)	16.2				
	50% reduction	17.8	17.5	17.3	17.1	16.8
	100 % reduction	17.8	17.6	17.3	17.1	16.8
Scotts Creek	Current condition (86% shade)	18.0				
	50% reduction	19.8	19.3	19.0	18.6	18.1
	100 % reduction	21.1	20.5	20.0	19.6	19.0

While groundwater inflows were found to be negligible in the measured condition of each study site, the potential for increases in groundwater inflows following near stream vegetation removal can be significant (e.g., Story et al. 2003, Surfleet and Skaugset 2013). Modeled increases in groundwater inflows, as expressed by increases in overall surface water flow, dampened the effect of stream temperature increases from canopy removal with 50 percent increases in streamflow under the 100 percent shade reduction scenario. Modeled stream temperature response to near stream canopy reductions from this study provides pertinent information to land managers and policy-decision makers in the assessment of potential impacts and development of adaptive management strategies for Little Creek and Scotts Creek.

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Post-Landslide Recovery Patterns in a Coast Redwood Forest[1]

Leslie M. Reid,[2] Elizabeth Keppeler,[3] and Sue Hilton[2]

Abstract

Large landslides can exert a lasting influence on hillslope and channel form and can continue to contribute to high in-stream sediment loads long after the event. We used discharge and suspended sediment concentration data from the Caspar Creek Experimental Watersheds to evaluate the temporal distribution of sediment inputs from 11 landslides of 100 to 5500 m^3. Slide-related suspended sediment loads were estimated as deviations from expected loads referenced to nearby control watersheds. For the two largest slides, suspended sediment export during the year of the slide accounted for 5 and 15 percent of the initial slide volumes, while subsequent export accounted for an additional 8 and 2 percent over the period for which export has been tracked (8 and 10 years). Regressions of excess sediment against time and storm size indicate that suspended sediment loads are likely to recover more quickly for small storms than large ones. Measurements of sediment storage along channels affected by the slides generally showed aggradation for 1 to 2 years. For the largest slide, however, downstream accumulation has now continued for at least 8 years. Nearly half of the sediment initially displaced by that slide remains in storage adjacent to channels and so may be subject to re-mobilization during future storms. In addition, in-channel deposits have triggered bank erosion and diverted the channel in places; much of the downstream increase in suspended sediment load during the post-slide years is derived from these secondary sources rather than directly from the slide debris.

Key words: channel condition, cumulative impacts, landslides, sediment yields, watershed recovery

Introduction

Large landslides can be an important source of excess sediment in the redwood region. Sediment from new landslides increases turbidity and may lead to channel blockages and aggradation, but influences can also persist long after a slide occurs. Slide scars can require decades to revegetate and far longer to redevelop forest soils, and slide debris can exert a lasting effect on channel form. In addition, slide deposits may contribute to pervasive increases in stream sediment loads as they succumb to bank erosion, and slow-moving coarse sediment inputs may take decades to reach the mouth of a watershed.

The cumulative impact analyses required for many planned land-use activities need to evaluate potential interactions between the effects of past and planned activities, so the persistent influences of past landslides need to be considered where activities have been associated with increased landsliding. Several studies have described long-term influences of major slide-generating storms on the migration of coarse sediments through the Redwood Creek watershed (Madej and Ozaki 1996, Nolan et al. 1995). In that case, slides were distributed widely across the basin, so the effects of individual slides could not be distinguished. An understanding of long-term effects at the scale of individual slides would also be useful. Records of suspended sediment loads at the Caspar Creek Experimental Watersheds in coastal Mendocino County allow evaluation of the temporal distribution of sediment inputs during and for up to a decade after major landslide-generating storms, and repeated channel surveys allow analysis of changes in storage of landslide-generated sediment after the slides occurred.

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Study Site

Watershed research has been carried out since 1961 in the North Fork (473 ha) and South Fork (424 ha) Caspar Creek Experimental Watersheds (fig. 1) (Cafferata and Reid 2013). The watersheds are underlain by sandstones and shales of the Coastal Belt of the Franciscan Complex. Most soils developed on slide-prone slopes are relatively shallow (50 to 150 cm) loams (North Fork) to very gravelly loams (South Fork). About 15 to 20 percent (North Fork) and 20 to 65 percent (South Fork) of the surficial soil horizons are composed of clasts >2 mm. Clay-rich subsoils are cohesive, compactable, and can impede drainage. Mean hillslope gradient is about 25^{o}, and channels of third order or larger often flow through inner gorges with side-wall gradients of up to 60^{o}. Drainage density in the area is about 4.6 km/km^{2}. Evidence of ancient deep-seated landslides is common (Spittler and McKittrick 1995).

Annual precipitation averaged 1160 mm (standard deviation 339 mm, range 400 to 2200 mm) at the mouth of the South Fork between 1962 and 2015. About 95 percent of the precipitation falls as rain between October and May; snowfall is not important. The watersheds currently support second- and third-growth forests dominated by coast redwood (*Sequoia sempervirens* (D.Don) Endl.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco); old-growth was logged between 1860 and 1904.

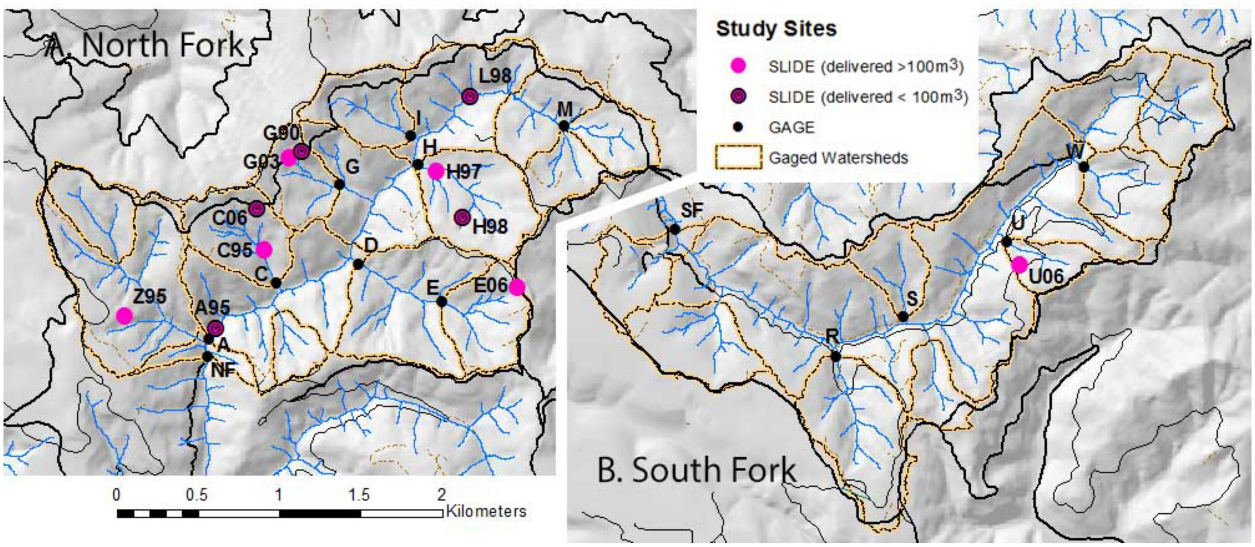

Figure 1—Post-1985 landslides of >100 m^{3} in the A. North and B. South Fork Caspar Creek Watersheds. Single and double letters—initials of the gage names—indicate locations of the gages analyzed.

The Experimental Watersheds have hosted two major experiments, and a third is now beginning. The South Fork was selection logged and tractor-yarded between 1971 and 1973, while the North Fork was left as a control to permit detection of the effects of South Fork logging on sediment and flow. Portions of the North Fork were then clearcut logged and primarily cable-yarded between 1985 and 1992, with three sub-watersheds left as controls. Papers included in Ziemer (1998) describe the experimental designs and results of both experiments. Annual suspended sediment yield at the North Fork before experimental treatment was 68 t km^{-2}yr^{-1} (95 percent CI: ± 39 t km^{-2}yr^{-1}; Reid and Keppeler 2012).

Reid and Keppeler (2012) examined conditions contributing to landsliding at Caspar Creek and found an increased incidence of large landslides after clearcutting, particularly in logged areas adjacent to roads. Results suggested that infrequent slides larger than 2500 m^{3} may contribute more than 20 percent of the suspended sediment load over the long term. Large post-logging slides were found to occur preferentially during the period when small roots have decomposed and foliar interception of rainfall has not yet recovered; the largest slides occurred 10 to 15 years after logging, soon after pre-commercial thinning had again reduced rainfall interception (Reid and Keppeler 2012). Although rotational slides and earthflows also occur, most recent slides—including those >100 m^3—are planar failures.

Methods

Three kinds of data were used for the analysis: stream-gaging records, landslide inventories, and channel surveys. Gaging stations were installed at the mouths of the North and South Fork watersheds in 1962, at 13 new sites in the North Fork watershed in 1984-1985, at two more North Fork sites in 1999 and 2001, and at 10 South Fork sites in 2000 (fig. 1); six of the North Fork gages were decommissioned in 1995. The two downstream gages are located at sharp-crested weirs with inset 120° V-notches, while gaging along the main channels is carried out at rated sections. Tributary gages originally employed Parshall flumes, and these were replaced by Montana flumes in the early 2000s. Henry (1998) describes the monitoring network and protocols in use until 1995. Since then, suspended sediment loads have been calculated from continuous turbidity records calibrated for each storm at each station using suspended sediment samples (Lewis and Eads 2009). Because >90 percent of the suspended sediment is carried during storms, storm sediment loads were used for this analysis. Coarse sediment deposition in the weir ponds is surveyed each year. Lewis (1998) reports that the weirs trap about 40 percent of the suspended load entering the ponds.

The entire gaged channel network is walked at least once a year to map the distribution of new slides capable of contributing sediment to the channels; non-contributing slides are also mapped whenever they are observed. Scar dimensions are measured and the volume of displaced material is estimated, as is the volume of sediment still in storage at the site. In some cases, sediment deposition downstream of major slides is described or mapped. Slides can usually be associated with particular storms on the basis of survey timing, field evidence, and gaging station records.

The 124 cross sections established in the 1980s along the main North and South Fork channels are resurveyed biennially using a rod and level; scour and fill calculations at the cross sections define changes in sediment storage and document channel incision and widening. Following the E06 slide (notation: watershed EAG, hydrologic year hy2006), areas of slide-related deposition were mapped and six cross sections were established and then resurveyed in 2007, 2008, 2015, and 2016 to track deposition and scour along the downstream channel. North Fork channel profiles were surveyed in 1992, and selected reaches were resurveyed using a total station between 2000 and 2006. Along the reach downstream of the E06 slide, measured changes in cross-sectional area were used in conjunction with the deposit maps to determine scour and fill volumes between surveys. In addition, periodic inventories of sediment storage along gaged channels were made annually between 1985 and 1996, and again along selected reaches (including those affected by the E06 slide) in 2006 and 2016. These inventories provide estimates of sediment volumes trapped by debris dams (steps associated with wood or roots) taller than 0.3 m which store >0.14 m^3 of sediment.

Suspended sediment inputs during and after the slide-generating storms were estimated as the deviations from expected sediment loads at a gaging station downstream of each slide. Most analyses used data from the nearest downstream gage and from the North or South Fork control gages (North Fork: HEN, IVE, MUN; South Fork: RIC, SEQ, WIL). Pre-slide relations between loads at treatment and control gages (e.g., fig. 2a) were used to estimate the expected load for each post-slide storm, and the deviation from the expected load was estimated as the mean of the differences between expected and observed loads as calculated using each of the three controls' records (e.g., fig. 2b). The Z95 slide occurred before the XYZ gage was installed, so a relation was developed between storm loads measured at ARF gage, immediately upstream of the XYZ confluence, and at NFC gage, located 120 m downstream. Sediment from Z95 was then estimated as deviations from the pre-slide relation between the ARF and NFC loads. Two slides triggered debris flows that damaged the downstream gage. In the case of E06, data from a station located farther downstream were employed to m downstream. Sediment from Z95 was then estimated as deviations from the pre-slide relation between the ARF and NFC loads. Two slides triggered debris flows that damaged the downstream gage. In the case of E06, data from a station located farther downstream were employed to characterize the slide. At the UQL gage, sampling resumed 24 hours after the U06 slide occurred, and loads were estimated from a hydrograph and turbidity record reconstructed using data from nearby gages. In several cases, slide effects were superimposed on logging-related sediment inputs, complicating the analysis of slide-related inputs.

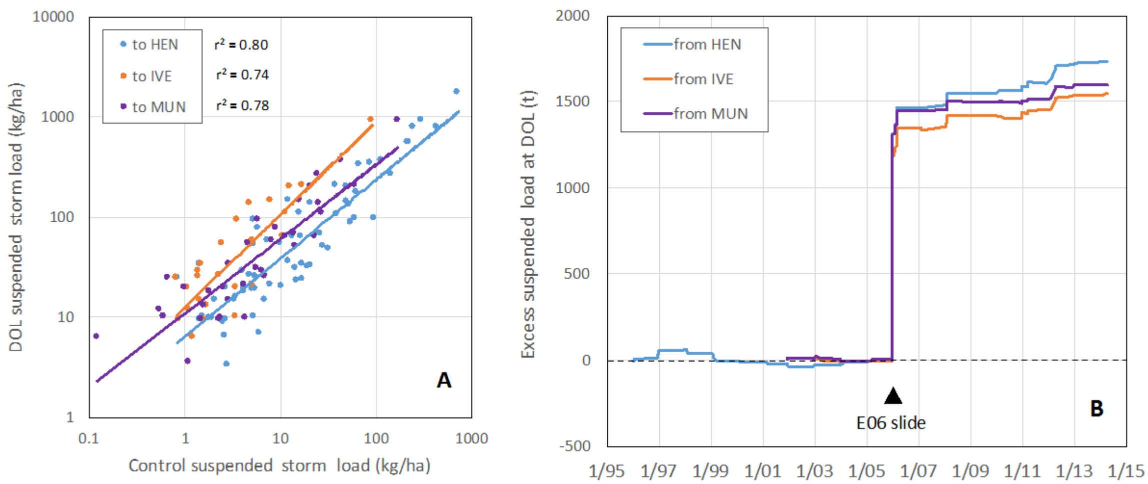

Figure 2—A. Pre-slide relations between storm suspended sediment loads at DOL and at controls HEN, IVE, and MUN; and B. cumulative excess suspended sediment at DOL between 1/1/1995 and 8/1/2014.

The initial phase of suspended sediment export was considered to end at the conclusion of the wet season during which the slide occurred. Revegetation before the onset of the following winter is expected to partially stabilize slide deposits. Delayed sediment export is calculated for the period between the hydrologic year following slide occurrence and the time that the record is either terminated, impacted by the occurrence of another slide upstream of the gage, or reattains its pre-slide characteristics. Storm suspended loads are calculated as kg/ha or tons. For comparison, slide and deposit volumes are converted to estimated mass by assuming an original average bulk density of 1.8 t/m^3 for the combination of soil mantle and weathered bedrock displaced by large landslides, 1.5 t/m^3 for valley-bottom regolith, 1.8 t/m^3 for slide deposits, 2.0 t/m^3 for debris flow deposits, and 2.1 t/m^3 for poorly sorted channel gravels.

Results

Landslide inventories carried out in the North and South Fork watersheds since 1987 and 2000, respectively, have identified 11 landslides or landslide complexes with volumes >100 m^3 (fig. 1, table 1). An average of 47 percent of the sediment originally displaced by 10 of the slides remained on site at the time of the first post-slide inventory, with values ranging from 3 to 97 percent.

Suspended sediment export during the year of the slide accounted for 0 to 15 percent of the initial slide mass (mean: 4 percent), while export following the first year accounted for an additional 0 to 8 percent (mean: 3 percent). For the two largest slides, 5 and 15 percent of the slide mass was removed as suspended load during the first year, and 8 and 2 percent in following years. Most records were terminated by decommissioning of a gage or occurrence of another slide. Only the C06 slide appeared to stabilize within the analysis period, and it was notable for having retained a high proportion of the slide material on the scar; outputs from the E06 and U06 slides continue to be tracked. No signal was detected in two cases (A95 and L98) for which the nearest downstream gage at the time of the slide

ARF load for the year of the slide, so any response is expected to be below the detection limit at this downstream gage.

Channel response to slides delivering >100 m^3 of sediment (n = 6) was evaluated using cross-section data to calculate scour and fill along the mainstem reach downstream of the point of entry for slide sediment (table 1). Four of these slides were associated with positive accumulations the first post-slide year of measurement, which, depending on whether or not the slide occurred during a survey year, was either the summer after the slide or the following summer. Deposits associated with most slides continued to aggrade for only 1 or 2 years, but E06 had a more enduring impact on the mainstem channel. The cross-sectional area at the nearest mainstem cross-section changed by more than 3 m^2 due to deposition after the slide, and this reach showed continued aggradation as recently as 2015. On the South Fork, U06 did not produce identifiable aggradation in the main channel, which for several years had already been responding to sediment inputs from a reactivated 1974 failure and from channel incision associated with road decommissioning. Deposition was evident in UQL tributary itself.

Table 1—Slides of >100 m^3 in the Caspar Creek Experimental Watersheds after 1986, and disposition of slide debris in the year of the slide ("initial") and following years ("delayed")

		Scar (m^3)		Tributary (m^3)		Mainstem (m^3)		Suspended load (t)		Period
Slide	Date	Total	On-site	Initial	Delayed	Initial[a]	Delayed	Initial	Delayed	(yr)[b]
G90	5/90	280	280	--	--			0	19	5
Z95	1/95	3600	1600	--	--			340	490	10
A95	3/95	370	280	NA	NA	32	-6	nd	nd	2
C95	3/95	130	120	--	--			4	--	0
H97	12/96	120	4	--	--	28	-15	11	1	1
L98	<2/98	180	90	NA	NA	-9	6	nd	nd	2
H98	3/98	100	70	--	--			2	6	1
G03	12/02	2000	760	--	--	14	-69	48	--	3
E06	12/05	5500	--[c]	1853	-18	115	47	1410	210	>8
C06	12/05	210	200	--	--			11	2	>8
U06	3/06	250	20	--	15	-35	-10	16[d]	6	>8

Tributary and mainstem values represent deposition or erosion; "nd" indicates that a suspended sediment response was not detected; and dashes indicate that measurements were not made at the site or, for suspended sediment measurements, ongoing changes in logging-related inputs prevented interpretation of results. Slides are designated by sub-watershed initial and hydrologic year (e.g., G90 occurred in GIB during hy1990).

[a] For mainstem deposition, "initial" may include the first 1 or 2 years of aggradation.

[b] Period: duration over which post-slide analysis is possible; analysis may be terminated due to decommissioning of a gage or occurrence of a new slide in the watershed upstream of the gage

[c] Estimated to be <10 percent of the slide volume.

[d] A minimum value for suspended sediment as the period of hyperconcentrated flow was not sampled.

The three large slides for which nearby downstream gaging records are available (Z95, E06, U06) were evaluated in more detail. The 5500 m^3 E06 slide complex (approximately 9500 t of displaced material, 3000 t from the debris flow track) occurred upstream of four gages, one of which (EAG) was located 150 m downstream of the toe of a debris flow triggered by the slide; a portion of the flow travelled on and damaged the gaging flume. Consequently, neither the initial sediment pulse nor the next storm were recorded at EAG (drainage area: 27 ha). Downstream 560 m, the DOL gage (77 ha) recorded a suspended sediment load of 1260 t for the pulse (fig. 3a), equivalent to about 13 percent of the sediment displaced by the slide. The ARF gage (384 ha), another 1100 m downstream of the EAG gage, recorded 1390 t for the initial pulse, with the difference likely to be attributable to a combination of estimate uncertainty, breakdown of clasts, and the greater capacity of North Fork flow to carry sediment in suspension. The EAG gage was repaired within 17 days, and subsequent measurements showed that the primary source of slide-related sediment to downstream reaches had shifted from above the EAG gage to the reach between the EAG and DOL gages (fig. 3b).

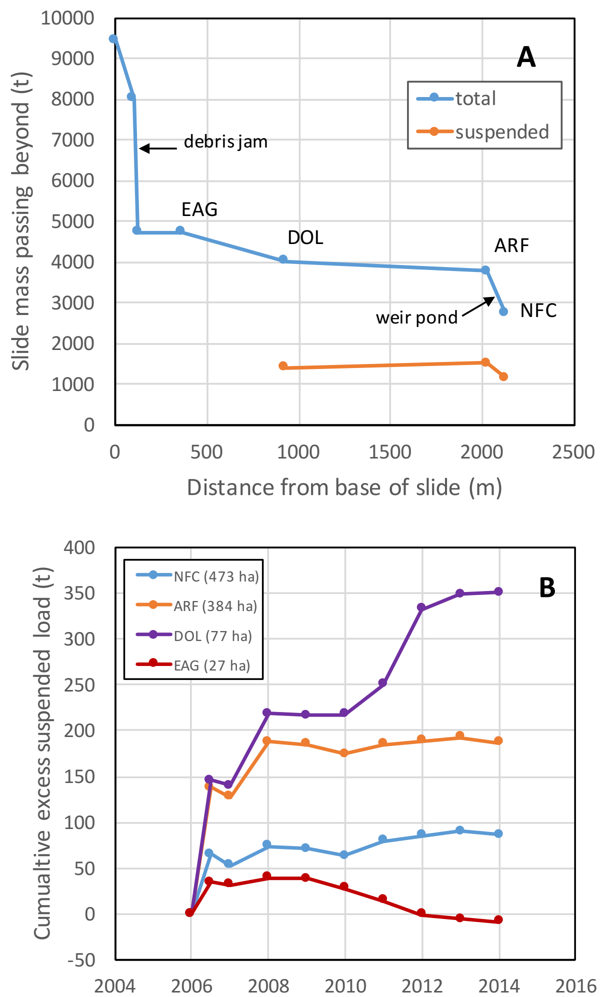

Figure 3—A. Initial distribution of sediment from E06 (through hy2006), and B. E06 excess suspended sediment loads during subsequent storm events at four downstream gages for hy2006 through hy2014.

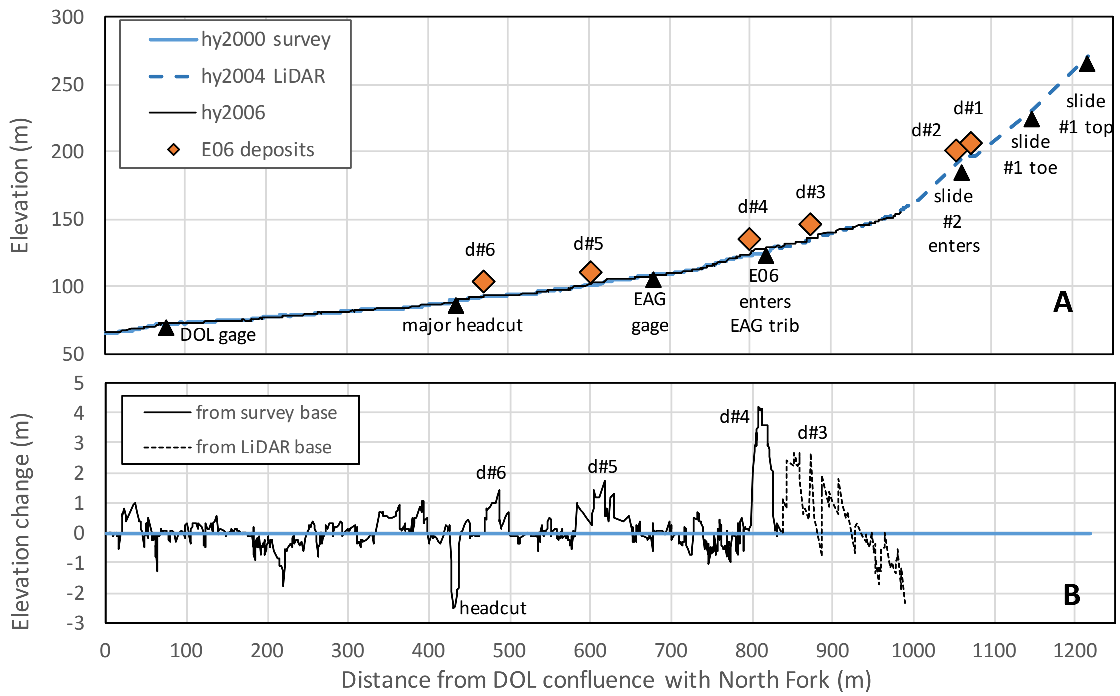

Figure 4—A. hy2000 or hy2004 (pre-slide) and hy2006 (post-slide) channel profiles in DE tributary, and B. elevation change between surveys (survey precision: ±1 cm).

Aggradation from E06 is evident along a 400-m reach where thalweg profiles were surveyed in 2000 and 2006 (fig. 4a). Following the slide, mean thalweg elevation increased 0.33 m along the reach, with a maximum increase of 4 m (fig. 4b). Most of the aggradation occurred in three 30- to 70-m segments with a total estimated deposit volume of 1090 m^3 (d#4, d#5, d#6). These depositional features were not contained within the pre-existing banks, and an entirely new channel was created at the d#5 site. Additional deposits along the unsurveyed slide track above this reach were estimated on the basis of field measurements of deposit dimensions to contain 1660 m^3 of slide material for a total of 2750 m^3—about 68 percent of the volume of the slide complex.

Channel deposits from E06 showed substantial reworking in the months following the slide, and cross-sections established in February 2006 document later changes (table 1). Material accumulated during the first two post-slide winters at d#3, but scour prevailed by year 9. Just downstream, a 2-m debris step storing most of the new d#4 deposit had begun to incise by 2008 and retreated about 5 m by 2016. At d#5, the channel was stable after hy2006, having established a 3-m-wide U-shaped channel with a maximum depth of 0.5 m. This channel occupies about one-fourth of the deposit area, and the pre-slide channel remains buried. Channels through the upstream deposits are similarly confined to only 6 to 30 percent of the deposit width. Channel depths at these cross-sections range from about 0.5 to 1.75 m.

Step inventories describe in-channel sediment storage at a finer scale. Along the 400-m surveyed reach below the slide track confluence, 33 organic steps were present in 1995 and in 2006, after E06 occurred; 3 had disappeared by 2016. However, overall in-channel storage behind debris steps increased from 52 m^3 to 740 m^3 between 1995 and 2006, mostly due to wood associated deposits d#5 and d#4, behind which most of the over-bank deposition occurred. The in-channel step deposition component at d#5 (110 m^3 of the total d#5 deposit of 225 m^3) was deemed stable in 2016 because the post-slide channel was no longer in contact with the sediment. Of the original 572 m^3 of step deposition at d#4, an estimated 10 m^3 had been excavated by 2016 and about 16 m^3 remains within the active channel, while the remainder has revegetated and appears stable. One-third of the other steps along the EAG-DOL channel were relatively stable, showing increased volumes of 20 to 34 percent (mean: 24) between the pre-slide (1995) and post-slide (2006) inventories and returning to approximate pre-slide totals by 2016. In 2006, 21 steps showed signs of accretion, and only one showed erosion. A decade later, the numbers of eroded and aggraded steps were nearly equal. Two of the 11 steps initially buried by slide debris were re-exposed by 2016, while two others had washed away. Gravel and cobbles dominated the deposits at all but a few steps. Although most of the sediment storage within the active channel is associated with organic steps, the volume of slide-derived sediment stored in these steps (160 m^3) is small compared to the amount stored outside of the active channel above the major debris jam (d#3, d#4: 1630 m^3) and on the valley bottom (d#5, d#6: 400 m^3).

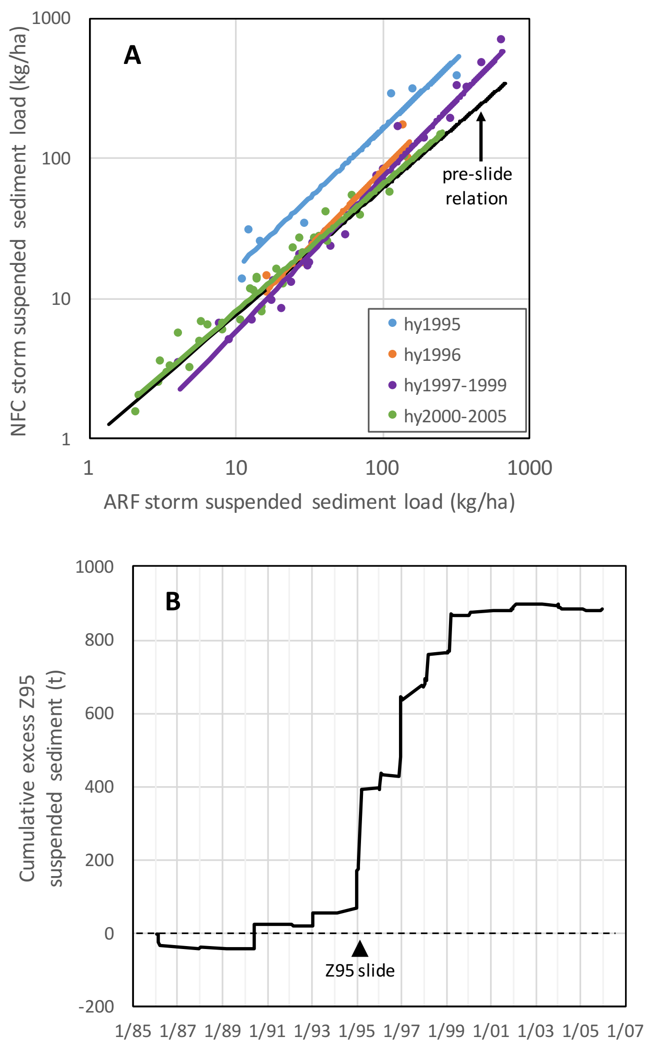

Figure 5—A. Relations between storm suspended sediment loads at ARF and NFC before and after the Z95 slide; and B. excess storm loads from XYZ tributary (draining the Z95 slide) through time.

Sediment data for the Z95 slide are more limited. Because only the tributary affected by Z95 (XYZ tributary, 77 ha) enters the North Fork channel between gaging stations ARF and NFC, change in output through time from Z95 is evident as a shift in the relationship between storm suspended sediment loads at stations ARF and NFC (fig. 5a). The cumulative sediment deviation through time shows most of the slide effect to have dissipated by 5 years after the slide (fig. 5b). Analysis ends at the time of the E06 slide, which is expected to have changed the relation between loads at ARF and NFC due to the unusually large influx of slide-derived sediment during that event. Deposition was not surveyed for Z95.

The 250 m^3 U06 slide appears to have occurred in two phases, with a preliminary slide releasing about 4 t of suspended sediment in late December, followed 74 days later by second failure which triggered a debris flow that destroyed the UQL gage (drainage area: 13 ha). Although neighboring gage records and a reinstalled sampler allowed estimation of the storm load at the site, the reported value represents an underestimate because hyper-concentrated flows associated with the event were not sampled. A minimum of 16 t was exported in suspension after the slide, along with a large volume of debris flow sediment; and an additional 6 t was exported over the following 8 years (fig. 6).

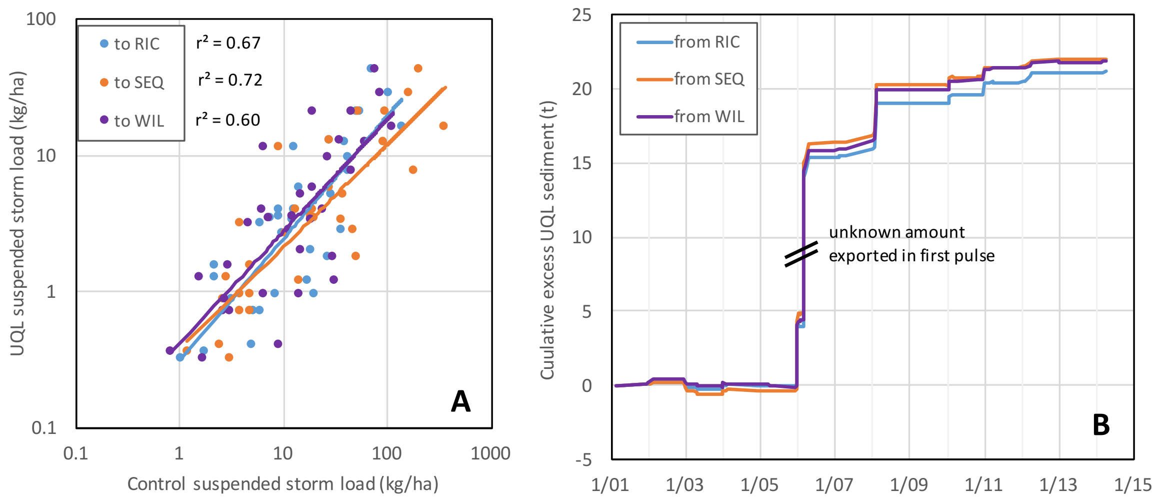

Figure 6—A. Pre-slide relations between storm suspended sediment loads at UQL and at controls RIC, SEQ, and WIL; and B. cumulative excess storm suspended load at UQL between 1/1/2001 and 1/1/2015.

Discussion

Although large landslides mobilize considerable volumes of sediment over a short period, much of that sediment is quickly redeposited on or immediately below the landslide scar. At Caspar Creek, about half the displaced sediment remained on-site for the 10 major slides for which on-site measurements were made. During the slide, the rest of the sediment may be deposited lower on the slope, in the channel below, or along downstream channels. Landslide debris in contact with flow becomes subject to further erosion and contributes to suspended sediment loads downstream. Because most Caspar Creek landslides are associated with major rainstorms, the slide scar and proximal deposits often also contribute fine sediment to downstream loads. Some slides incorporate enough water to move down-channel as debris flows. During and immediately after such flows, downstream suspended sediment concentrations can be extremely high. Concentrations at the DOL gage reached 94 g/l during the E06 event and remained above 20 g/l for more than 4 hours.

Fluvial erosion of landslide and debris flow deposits can become a persistent source of suspended sediment. However, only a portion of the deposited sediments are fine enough to be transported in suspension. At Caspar Creek, about 50 to 80 percent of the mass of landslide-prone soils is of sand size and smaller, and large slides also incorporate weathered bedrock with a much lower proportion of suspendible sediment. Weathering and abrasion of larger clasts will make additional sediment susceptible to suspension through time. However, revegetation of the deposits increases the stability of even fine-grained deposits, and winnowing of fine sediments from around large clasts leaves an armor layer that progressively reduces surficial erosion rates on the slide scar and deposits.

For the two largest slides at Caspar Creek (table 1), 5 and 15 percent of the debris contributed to the suspended load during the slide-generating storm and during later storms of the same wet season.

Subsequent erosion of the slide scar, deposits, and disturbed downstream channels accounted for increased suspended sediment loads for several years after the initial slides, with the effect dwindling with time after the event. Overall, suspended sediment contributions after the initial year accounted for an additional 8 percent of the total slide mass from Z95 and 2 percent from E06.

Opportunities for sediment storage are limited in the high-gradient, incised channels typical of 1st-order streams at Caspar Creek. Organic debris dams common in headwater channels provide a finite storage capacity that may be orders of magnitude less than the volume of sediment mobilized by a major slide. Two debris jams emplaced by the E06 debris flow provided key obstacles that accounted for about one-third of the deposition along the first 500 m of channel below the slide (d#4 and d#5, fig. 4); another is where the flow had to negotiate an abrupt turn to continue down-valley (d#3). These jams impounded sediments across the valley bottom, spanning widths more than four times that of the active channel. In contrast, U06 travelled straight down an abandoned skid trail and continued its path directly down the channel; any obstacles present were insufficient to trap the debris. Where deposits overwhelm the transport capacity in an existing channel, flow may be diverted to carve a new channel through valley-bottom deposits, as occurred at d#5 during the E06 event. In larger, low-gradient channels such as the North Fork, storage is largely controlled by woody debris loading and floodplain or terrace connectivity.

The amount of excess sediment transported in years following a slide depends in part on the progress of stabilizing influences, but it also depends strongly on the size and timing of the subsequent storms; during a drought year, not much sediment moves. Calculated excess suspended sediment loads for 2006 to 2014 from the UQL (*U*, t/ha) were evaluated in a multiple regression against the average of storm sediment loads at nearby control watersheds RIC, SEQ, and WIL (*RSW*, t/ha) and time after the U06 slide (*t*, yr). The resulting relation shows significant ($p < 0.001$) dependence on both influences:

$$U = 10.5 + 0.81\ RSW - 4.3\ t \qquad n = 41 \qquad r^2 = 0. \qquad (1)$$

This relation suggests that recovery for small storms occurs more quickly than for larger storms, with the 0.5-yr return-interval events (averaging 34 kg/ha at the control watersheds) showing recovery in 8 years, while recovery for the 2-yr event (146 kg/ha at the controls) is expected to require several decades. A 1.62-yr event (as evaluated from flow records at NFC) did occur in December 2015, 10 years after the U06 event. Equation 1 predicts an excess load of 64 kg/ha for the event at the UQL gage, and the preliminary data for the storm indicate that the observed excess load was 62 kg/ha. Predicted excess load for the same storm would have been 103 kg/ha had it occurred during the year following the slide.

Six of the 8 years following the U06 slide had below average rainfall, and the largest storm during this 8-yr analysis period (hy2006 to hy2014) had a return interval of about 1.7 yr at the NFC gage. The net post-slide suspended sediment export for the period is thus expected to be substantially lower than it would have been for a more typical 8-yr sequence of storms. We thus applied Equation 1 to an average distribution of storms of 10-yr return interval or less for an 8-yr period. Results suggest that the excess suspended load observed during the first 8 years after U06 is about a third of that expected for the hypothetical average storm distribution. However, weather conditions also affect rates of stabilization of the slide scar and deposits. Drought years may hinder revegetation, for example, while wet years may trigger new failures on exposed scarps.

Although Z95 and U06 slides and their watershed conditions differ in character, excess loads associated with Z95 show a similar pattern:

$$Z = 6.3 + 0.42\ A - 2.8\ t \qquad n = 80 \qquad r^2 = 0.66 \qquad (2)$$

where *Z* is the excess storm sediment from XYZ tributary (kg/ha), *A* is the storm suspended sediment load at ARF (kg/ha), *t* is years after the slide occurred, and $p < 0.05$ for each variable.

The U06 slide is notable in that most of the slide debris left the watershed as a debris flow, leaving few deposits on the slide scar or in the channel above the UQL gage. The Z95 slide was similar in that it also did not leave a persistent channel-blocking deposit in the main XYZ channel. At E06, in contrast, much of the debris flow that mobilized the slide debris came to rest upstream of the EAG gage at d#3 and d#4 (fig. 4), blocking the EAG channel with a deposit as much as 4 m thick, and a tongue of the flow moved beyond the gage to deposit 390 m^3 of sediment (d#5 and d#6) in the channel between the EAG and DOL gages. Probably in part because of this difference, the post-slide sediment patterns at the EAG and DOL gages are quite different from those produced by the Z95 and U06 slides. After E06, storm sediment loads increased by nearly an order of magnitude over expected values at the EAG gage, but dropped the following year and have remained significantly lower than pre-slide levels since 2010 (fig. 7A, $p < 0.001$). The large debris jam may have impeded sediment transport from about 52 percent of the watershed, and erosion of revegetating slide deposits in and downstream of the jam evidently do not make up for the reduction in sediment from upstream. The decline in sediment production at EAG also reflects a general reduction in sediment inputs from slopes not affected by the slide as they recover from 1990-1991 logging and from 2001 pre-commercial thinning. A similar decreasing trend in non-slide-related inputs would also have been present during recovery from Z95. Logging in that area occurred in 1985-1986, with pre-commercial thinning carried out in 1993.

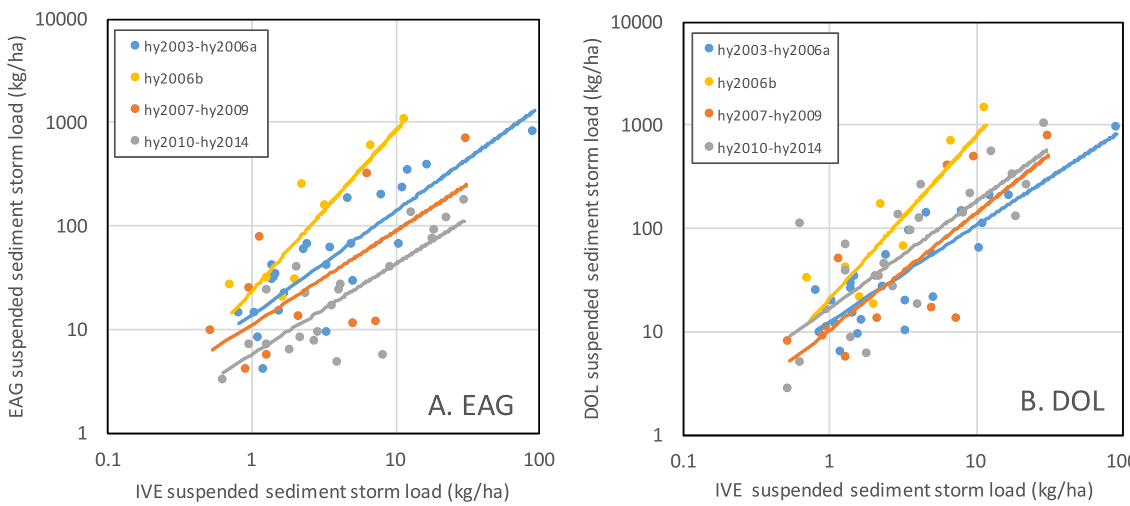

Figure 7—Relations between storm suspended sediment loads at A. EAG gage (drainage area: 27 ha) and B. DOL gage (77 ha) and those at control watershed IVE (21 ha) for periods before (hy2003-hy2006a) and after (hy2006b-hy20014) the 0.13 ha E06 slide.

In contrast, sediment loads at DOL (fig. 7B) were high during the remainder of the hy2006 wet season, then significantly decreased the following season ($p < 0.05$) but showed no further systematic decline after that. Although some of the excess sediment load at DOL is likely to be from remobilization of landslide sediments, a major portion represents 1) adjustment of the DOL channel to accommodate the new deposits and 2) accelerated headcut retreat through valley-fill deposits that date from old-growth logging and earlier. The DOL channel has widened about 20 percent since 2006, and a major headcut (fig. 4) activated soon after logging accelerated its retreat after the slide and is now located 50 m upstream of its initial 1993 location; this headcut has now excavated about 300 m^3 of sediment.

The spatial and temporal patterns of sediment production from major Caspar Creek landslides have several implications for cumulative impact analysis. First, there can be a lengthy period over which new activities may superimpose impacts on those persisting from earlier activities. Deposits from large slides can trigger secondary erosion in downstream channels, and these secondary in-channel sources may require longer recovery periods than erosion of the slide scar and primary slide deposits; this is the case along the DOL-EAG channel, where much of the post-E06 sediment production is from old deposits destabilized by flow and deposition associated with the slide. Some of those old deposits appear to themselves have resulted from debris flows and landslides, and it is likely that E06 deposits will play a similar role in the future. In addition, differences in sediment load recovery rates for large and small storms may lead to an erroneous assumption—after several years without major storms—that sediment production has recovered following a slide, when, in reality, future large storms may continue to produce excess loads even after those from smaller storms have returned to pre-slide levels.

Second, long-term influences from major slides may make it difficult to assess “background” sediment input rates where initial land-use impacts occurred long ago. Field evidence suggests that some of the valley fill deposits along the North and South Forks and their tributaries date from the period of old-growth logging, and valley-fill deposits are the source of a major portion of today’s sediment yield at Caspar Creek (Reid et al. 2010), which increased markedly after recent logging (Lewis et al. 2001). To the extent that erosion processes associated with old-growth logging contributed to valley-fill deposits, such processes would be influencing the expression of today’s land-use impacts, and so would be contributing to an on-going cumulative impact.

Third, downstream changes in the expression of effects associated with large landslides means that different resources will be affected in different ways at different locations along the channel system. Currently, the first-order channel draining E06 continues to sustain perennial flow during the dry season because fill in the swale and channel were scoured out by the E06 debris flow; this represents a major change in dry-season access to surface water in the headwater catchment. In contrast, aggradation at downstream sites may have reduced the annual duration of surface flow and increased fine sediments in pools. Channel response along the mainstem of North Fork is of particular concern because the stream provides habitat for threatened and endangered coho salmon (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*), and aggradation reduces both the amount and quality of summer habitat.

The current study has focused on aspects of deposition and suspended sediment transport after large slides. However, a large quantity of landslide-derived sediment is transported as bedload, and bedload can itself be transformed to suspended load as clasts weather or as tributaries contribute bedload to streams with higher transport capacities. Future work is planned at Caspar Creek to evaluate patterns of coarse sediment transport after major landslide generating storms.

Conclusions

Landslides are an important source of suspended sediment at Caspar Creek, and a single large slide may influence channel form and sediment production and transport for many years. However, the observed short-term and long-term effects differ between slides, reflecting the particular setting and characteristics of each slide. For example, sediment loads decreased to significantly below pre-slide levels at the EAG gage by 4 years after the E06 event, probably due both to the emplacement of a debris flow deposit that may partially restrict sediment transport from half of the EAG watershed and to the on-going recovery trend from logging and pre-commercial thinning. Downstream, loads again increased, but along this reach the increase is due primarily to slide-related destabilization of the channel rather than to direct erosion of deposited slide debris. In order to evaluate potential cumulative watershed effects, it would be useful for resource planners to consider both landslide history and the vulnerability of legacy sediment storage features to re-mobilization.

Acknowledgments

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Development of Preventative Streamside Landslide Buffers on Managed Timberlands[1]

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Abstract

Shallow streamside landslides are a principle source of sediment on managed timberlands in northern California. Using an adaptive management process, LiDAR, and a detailed field-based landslide inventory, Green Diamond Resource Company (GDRCo) has redefined the interim preventative landslide tree-retention buffers it applies to steep streamside slopes along fish bearing (Class I) and non-fish bearing (Class II) watercourses. The application of these buffers are dependent on slope gradients and when applied, enhance and in some cases, expand upon the customary riparian buffers associated with these watercourse types in our California Timber Harvesting Plans (THP). They are designed to significantly reduce the amount of management related sediment delivery associated with landsliding when compared to historical management practices.

Initially, the steep slope prescriptions were derived from a pilot field inventory of streamside landslides during the developmental stages of an Aquatic Habitat Conservation Plan (AHCP). The Steep Slope Delineation study was a long term research project associated with the AHCP monitoring program with an objective of redefining the initial prescriptions based on a comprehensive field-based landslide inventory. The first phase of the steep slope project was completed in 2011, and in 2015 we completed the final phase.

The final results of the Steep Slope Delineation project covered roughly 145,690 ha (360,000 ac) of privately owned timberlands in California. The work included a review of aerial photographs, detailed field survey of slopes adjacent to 357, 0.8 km (half-mile) long, watercourse segments, and analysis of the resulting data. These data, characterizing more than 2,000 landslides, are used to develop new maximum buffer widths and new slope gradient buffer triggers which are exclusive to the four geographic areas within the ownership. While the majority of the buffer widths decreased, nearly one third increased in width. As for the slope triggers, slightly more than half of the slope gradients decreased, nearly half had no change, and a few increased. The revised steep slope prescriptions were submitted to federal agencies in December of 2014 and were successfully incorporated into the AHCP in January of 2015.

Introduction

The Steep Streamside Slope (SSS) Delineation project is an analysis of streamside landslides on privately owned timberlands that are bound by an Aquatic Habitat Conservation Plan (AHCP). The results of this analysis determine the new SSS default protection measures for the ownership in northern California. These buffers are applied to specific areas that are known to have a high potential for streamside landsliding and enhance the standard Riparian Management Zones (RMZ) in those areas (A generalized example of a SSS is shown in fig. 1). This project is an expansion of a previous landslide study that produced the AHCP Steep Streamside Slope initial "default" protection measures during development of the AHCP.

The primary goal of the SSS prescription is to reduce the amount of sediment delivered to watercourses as a result of streamside landslides generated by forest management related operations. The objective of the SSS prescriptions, which will be assessed at a later date as part of the SSS Assessment project, is to achieve a 70 percent reduction of delivered streamside landslide volumes in comparison to historical management related streamside landslides. This paper presents the findings of the final phase of the SSS Delineation project which involved a review of streamside slopes in each

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of the Hydrographic Planning Areas (HPA) except the Coastal Klamath, which was the focus of the first phase of this project completed in March of 2011. A summary of the Coastal Klamath findings is also included in this report.

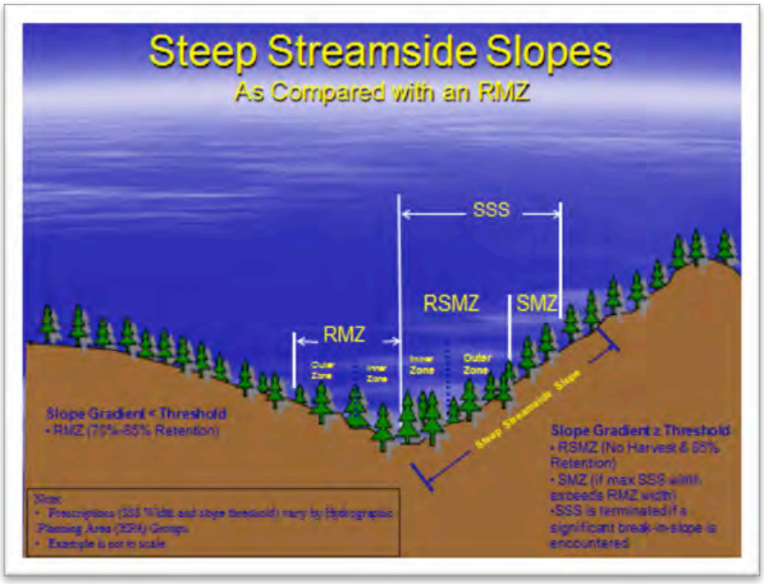

Figure 3—Generalized schematic diagram of a SSS buffer. The SSS Zone shown at right is compared with an RMZ to the left. When the SSS -gradient threshold is triggered the RMZ is enhanced by the application of SSS measures through an increase in overstory canopy retention in that area.

The initial default SSS prescriptions were established during the developmental stage of the AHCP. These default prescriptions were based on an initial study that evaluated streamside landslides. The purpose of the study was to develop an expanded protection zone adjacent to watercourses that would reduce the amount of streamside landslides related to timber harvesting. Furthermore, the initial study was small in scope and designed to produce conservative results. The study intentionally targeted areas that exhibited high concentrations of landslides due to the limited scope and compressed time frame to conduct the study. The SSS Delineation project expands upon the initial landslide evaluation effort with: a larger sample size, more thorough review of landslides, and a random sampling process applied to each of the HPA's outlined in the AHCP. This project will more accurately define the SSS protection measures and achieve the AHCP objectives directed at streamside landslides.

The 70 percent reduction in sediment delivery compared to historical sediment delivery was a goal developed between Green Diamond Resource Company (GDRCo), U.S. Fish and Wildlife Service, and National Marine Fisheries Service during the development of the AHCP. The actual performance of the SSS buffers will be evaluated over the next 15 years during the SSS Assessment Project and reviewed by an independent scientific review panel.

Our revised SSS slope gradient trigger was determined by reviewing streamside landslide data collected from our work and selecting the slope gradient that corresponds with landslides that account for 80 percent of the cumulative volume of sediment delivered to a watercourse. The maximum SSS buffer distance is determined by evaluating the distance from the main scarp to a watercourse that correspond with landslides that represent 60 percent of the cumulative volume of sediment delivered. This process was established and used during development of the AHCP. Both of the aforementioned cumulative volume assessment values were chosen with the assumption that the majority of the landslide data to be collected would have occurred under historical logging practices that are no longer used (i.e., reduced or no riparian protections, intensive ground-based operations, oversized harvest units, poor road building practices, etc.). Both the sediment reduction goal and cumulative volume based buffer criteria are thought to yield conservative values based on these assumptions.

The project focuses on shallow streamside landslides that were active to historically active, were not caused by roads or skid trails, and have observably delivered sediment to a watercourse based on field observations. The AHCP road management and harvest related prescriptions are designed to address road and skid trail related landslides; accordingly, they are excluded from this study.

Additionally, the SSS prescriptions are not designed to address deep-seated landslides as they are addressed separately and on a case by case basis at the THP level. Shallow landslides associated with an active or historically active deep-seated landslide were also excluded from this project because the primary causal mechanism of failure of these features is due to movement of the deep-seated landslide which results in weakening of earth materials and over steepened slopes. These types of slides are addressed as part of their corresponding deep-seated landslides at the THP level.

Project Area

The project area is located on the north coast of California in a tectonically active area just north of the Mendocino Triple Junction (MTJ) where the North American, Gorda, and Pacific plates collide. Green Diamond Resource Company ownership spans from the California/Oregon border on the northern end to the town of Rio Dell on the southern end and as far inland as the headwaters of Redwood Creek. The ownership is broken up into hydrographic planning areas (HPA) that are associated with local watershed boundaries. There are nine HPAs within the project area, each of which is shown in fig. 2. Those HPAs with similar physical characteristics are lumped together into HPA Groups in order to apply regional variations in management prescriptions.

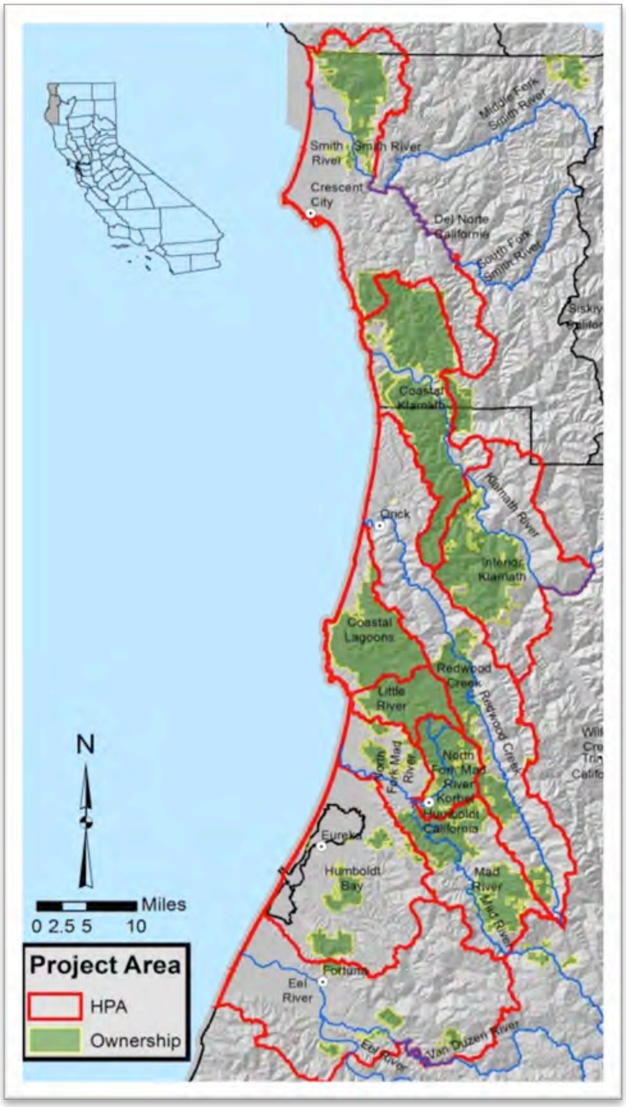

Figure 4—Project area; figure includes individual HPA, county and ownership boundaries.

Seismogenic fault systems in the area are part of the MTJ and include the north end of the San Andreas Fault zone to the southwest, the Mendocino fracture zone to the southwest, and the southern end of the Cascadia subduction zone to the west, just off the coastline. There are also numerous on-land upper plate thrust faults throughout the region that are thought to be considered as potential sources for seismic shaking (Cao et al. 2003, Kelsey 2001, Petersen et al. 1996), they include, but are not limited to: the Little Salmon fault, Mad River fault zone, Bald Mountain-Big Lagoon faults, Grogan, Surpur Creek fault, Saint George Fault and the Smith River Faults. The structural orientation of these faults is typically northwest-trending as a result of the compressional forces exerted on the region due to the converging North American, Pacific, and Gorda Plates.

Earth materials vary throughout the property due to the highly active tectonic regime in the region. At the southern extent of the ownership the bedrock is dominated by Miocene to late Pleistocene deposits of the Wildcat formation (Ogle 1953). The Wildcat formation is thought to be a coarsening upward regressional sequence of the ancestral Eel River basin. To the north the remainder of the property is dominated by deposits of the coastal and central belt of the Franciscan formation, which range in age from Pliocene to early Jurassic (McLaughlin et al. 2000). Bedrock within the Franciscan includes sedimentary, igneous, and metamorphic rock types; the most common earth materials encountered (generally speaking from north to south) are sandstone and metasandstone, mélange, schist, and the broken formation of the Franciscan. These units are typically characterized by broken to sheared moderately indurated sandstone and metasandstone (largely Korbel and Klamath HPAs), highly sheared siltstones and mudstones in an argillaceous matrix (largely found in Korbel and Klamath HPA Groups), quartz-mica schist (primarily found in the Redwood Creek HPA) and moderate to well indurated fractured greywake (primarily found in the Klamath and Smith River HPAs). Throughout the ownership bedrock may be capped by Pleistocene to Holocene alluvial sediments or marine terrace deposits (Irwin 1997). Surficial deposits are also found throughout the ownership in the form of alluvial deposits in the low lying areas along active streams and at the mouths of valleys. In addition, colluvium collects in the low lying zones such as swales and low lying slopes throughout the hillside. Due to rapid uplift, faulting, and subsequent down cutting through these young and poorly consolidated earth materials the general morphology of the region is typically characterized by immature topography. Steep valleys and landslide prone terrain are common throughout this region.

Methods

This project generally follows the same framework we established in our previous analysis of the Coastal Klamath HPA (Woodward et al. 2011). The results of that analysis are summarized in table 1. As part of that work we established protocols for aerial photo review, field methods, and calculations; each of which are briefly discussed below. In addition we developed distinct areas based on morphology which included using a topographic ruggedness model. We applied this same model to the remaining areas of the ownership and the results of that analysis are discussed below under "GIS Analysis".

Table 1—Comparison of revised Coastal Klamath SSS prescriptions and initial default prescriptions

Coastal Klamath SSS maximum slope distances m (ft) and minimum slope gradient thresholds			
SSSMU	**Class I**	**Class II-2**	**Class II-1**
1	72 (240) @ 65%	34 (110) @ 70%	41 (135) @ 75%
2	130 (425) @ 75%	59 (195) @ 85%	
Initial Default Buffers	145 (475) @ 70%	61 (200) @ 70%	30 (100) @ 70%

Historical Context

We reviewed historical aerial photographs for landslides and past land management practices for all of our field sites. Aerial photographs dated back to as early as 1942 and we typically included one set from each decade thereafter, as available, in our review. The majority of the aerial photos in our collection are at a scale of 1:12,000. Only landslides visible at the scale of the photos were mapped, which included slides typically 148.6 square meters (1,600 square feet) and larger. These landslides were transferred into our GIS landslide layer with associated tabular data that included; photo year and label, land use and approximate stand age at the time of failure, road and/or skid trail association, landslide type, slope curvature, geomorphic association, watercourse association, feature certainty and delivery. Landslide types are based on definitions modified from Cruden and Varnes (1996).

In reviewing the historical aerial photos and conversations with staff foresters and local historical logging experts, we developed a brief summary of the logging history of the ownership. Beginning in the late 1800s and early 1900s up until the mid-1930s, the central and southern portions of the ownership saw railroad and steam donkey logging. By the late 1940s into the late 1960s much of these areas were thinned or clearcut using ground-based tractor yarding methods, which utilized networks of skid trails. During this same time period, cable-yarding harvest methods were used on steeper areas. Some areas were thinned or clearcut on a smaller scale in the 1970s and 1990s. Much of the northern half of the ownership did not see harvesting start until much later. Tractor logging of old-growth timber in Redwood Creek, Klamath and Smith River areas started in the early 1950s to late 1960s. The tractors would construct networks of skid trails often times using side-cast fills on steep slopes, which tend to trigger road-related landslides. Interior Klamath saw the highest concentration of road and skid trail related slides of all HPAs. Some steeper areas were logged using cable-yarding harvest methods starting in the late 1950s and continuing until the mid-1970s. Recent timber harvesting across the ownership, from the late 1990s up to the present day, have utilized cable- and tractor-yarding with shovel yarding beginning to replace tractors around the year 2002.

While we understand the importance of comparing historic landsliding to climactic events, this was beyond the scope of this particular project. We are currently evaluating climactic impacts and the relationship with landsliding as part of a more encompassing mass wasting assessment of the ownership which is currently under review.

Project Design

The project design and methods involve the development of sample reaches, field measurements, calculations, data entry, and the analysis of the resulting data. Most were derived from our previous work in the Coastal Klamath HPA.

Our sample area involves hillsides adjacent to watercourses that are classified on the ownership as Class I and Class II watercourses. Class I watercourses are fish bearing streams and Class II watercourses are perennial flowing streams that support other aquatic life. The Class II streams are further subdivided into 1st and 2nd order (II-1 and II-2) stream types. Using the same approach applied during our work in the Coastal Klamath HPA, we sampled random hillside areas adjacent to Class I and Class II watercourses by breaking up the mapped Class I, Class II-2 and Class II-1 watercourses into half-mile survey reaches throughout the property.

The geographically distributed systematic random sampling method used both random selection and spatial distribution of the 0.8 km (half-mile) segments within the study area. This method involved delineating whole streams, from the confluence to the upstream end of a Class II, breaking these streams into approximate 0.8 km (half-mile) sample reaches. In addition, we stratified the Class I watercourses to ensure an even distribution of sample reaches from the lower, middle, and upper portions of these streams. Both the current sample and the previous Coastal Klamath reaches are shown in fig. 3. The current sample set, which excludes the Coastal Klamath, covers approximately 74 percent of the ownership (111,690 ha, 276,000 ac).

Our target sample rate was five percent (by distance) for Class II-2 and Class II-1 watercourses and 10 percent for Class I watercourses. The final sample percentages vary slightly from our original sample draw due to a variety of factors. Field review of watercourses during operations typically results in fluctuations in the location of a watercourse transition which affected some of the selected reaches. Another is that we ran into logistical issues in the field that prevented access to certain areas. For the current sample we surveyed a total of 293 km (182 mi) of streams, 93 km (58 mi) (11 percent) of Class I, 123 km (76.5 mi) (6 percent) of Class II-2, and 76 km (47 mi) (5 percent) of Class II-1 watercourses. An additional 77 km (48 mi) were surveyed in the Coastal Klamath HPA. For the entire SSS Delineation project, there were a total of 357 sample reaches, of which 264 were part of the current study and 93 were part of the Coastal Klamath work.

Field Work, Measurements, and Calculations

All landslides were reviewed in the field. Field work involved surveying the hillsides adjacent to the sample reaches for shallow streamside landslides. Our study focused on those landslides that were: a) active to historically active, b) not associated with active or historically active deep-seated landslides, c) non road- and non skid trail-related, and d) had observably delivered sediment to a watercourse. Only landslides greater than 3 m by 6 m (10 ft by 20 ft) were included in the survey. Data collected for each landslide included a field-developed cross section using a tape measure and clinometer, causal factors, slope characteristics, dimensions of the source area and slide debris, a field estimate of the delivery volume, distance from the crown of the slide to the edge of the watercourse, and the average slope gradient of the hillside effected. Cross sections show the main scarp, projected failure surface, the estimated original surface, and the extent of slide debris relative to the associated watercourse. We utilized the cross sections to determine the length of the rupture area, length of debris, estimated thickness of the failure, and estimate the thickness of the remaining slide debris.

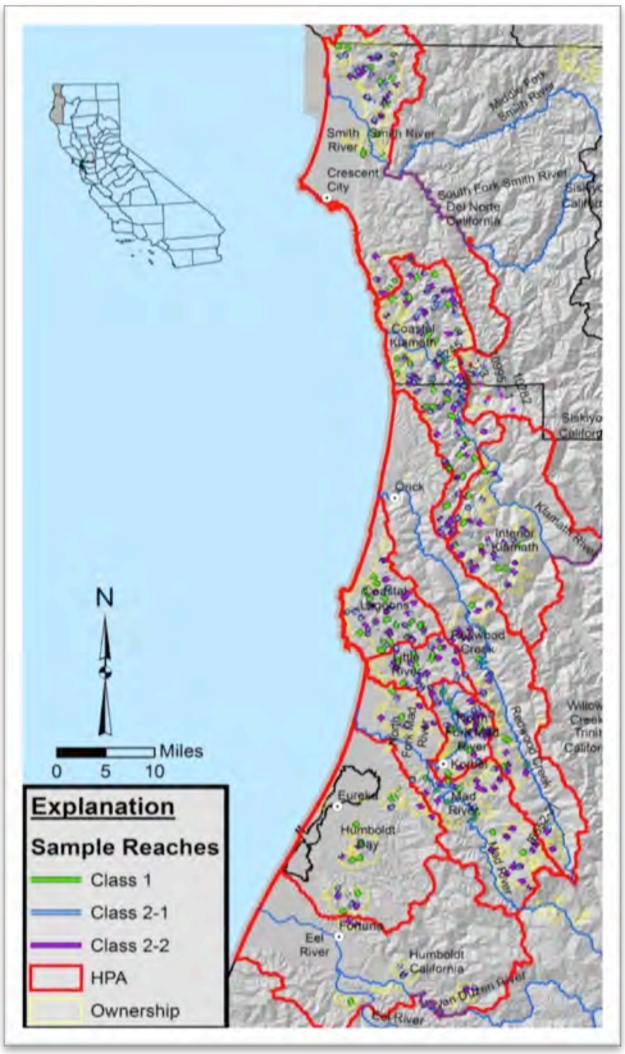

Figure 5—Project area and survey locations.

The average slope gradient of the hillside associated with the failure was obtained primarily by field estimates using a clinometer and the projected original surface gradient from the field-developed cross sections. In some cases, typically larger landslides or areas obstructed by thick shrubbery or excessively steep slopes, we utilized slope gradients derived from LiDAR. In each case we evaluated the average slope of the hillside associated with the failure defined as the area from the crown of the slide to the base of the hillside.

As mentioned above, we did not include landslides that were thought to have been caused by roads or skid trails in our analysis as road related landslides are addressed at the THP level and in our road management program. The purpose of the SSS prescription is to reduce the potential for streamside landslides typically associated with harvesting. Therefore our efforts focused on open slope streamside landslides not associated with roads. Determining whether or not a slide has been caused by a road or skid trail is a difficult task; especially if the failure is not a recent one. Often times, professional judgment is required in attributing a causal mechanism such as roads to the failure of a landslide. As a result we attempted to attribute road- or skid trail caused only to failures that appeared to have a reasonable or obvious negative association with a road or skid trail.

Volume estimates were derived from a calculation based on the length, width, and depth of both the rupture area and the remaining slide debris observed in the field. The calculation (Eq. 1) treats the slide rupture area and debris as a half of an ellipse and was obtained from published work by Cruden and Varnes (1996).

Eq. (1): Volume of delivered material = $(1/6\ \Pi\ Lr*Wr*Dr) - (1/6\ \Pi\ Ld*Wd*Dd)$

In this equation Lr, Wr, Dr, and Ld, Wd, Dd are defined as the length (L), width (W), and depth (D) of the rupture "r" and debris "d" of the landslide. In more complicated situations we found smaller slides nested within larger slides. In these instances, the slope distance and slope gradients of a smaller "nested slide" was not counted separately for the SSS analysis since it had failed as part of the larger slide. We did however calculate the volume of debris that had been delivered by a nested slide and added it to the volume of delivered material of the larger slide.

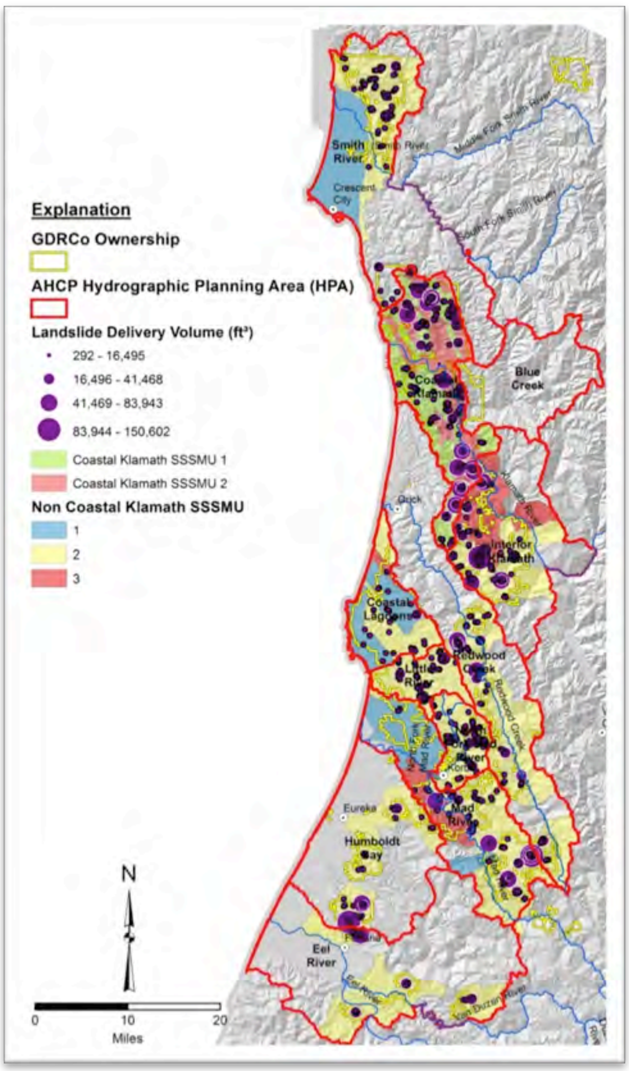

Figure 6—Landslide distribution by volume (includes previous work in Coastal Klamath HPA).

GIS Analysis

During our work in the Coastal Klamath we developed discrete areas based on morphologic complexity which we termed "Steep Streamside Slope Morphologic Units" (SSSMU). Looking at the shaded relief model from a 1-meter LiDAR DEM we determined that the morphologic complexity of this region could be separated into three discrete units (although two were combined due to lack of landslide data in those areas). Each of which showed a varying degree of landsliding. By doing so we could develop multiple SSS buffers within the HPA that would be more accurately applied to specific areas based on their morphology and landslide patterns. Using the same topographic ruggedness model from our work in the Coastal Klamath (Riley et al. 1999), we applied it to the rest of the ownership (fig. 4).

Although there were three discernable groupings within the Terrain Ruggedness Index (TRI) data, the groupings were not as strong as they had been in the Coastal Klamath. The majority of the ownership fell into the mid-range SSSMU group 2; roughly twenty five percent of the sample area fell within the SSSMU 1 or 3 groups. We evaluated these areas to see if they warranted specific SSS prescriptions but we did not find enough variation in the landslide data between these areas and the rest of the property to justify separate buffer criteria. This is illustrated in fig. 4 where we see a fairly even distribution of the number of landslides as well as an even distribution of the number of landslides by volume throughout the Non Coastal Klamath SSSMUs. If there were reason to identify prescriptions based on the SSSMUs, we would expect to see a larger portion of landslides, especially larger landslides, clustered within a specific SSSMU. This was evident and worked well for the Coastal Klamath HPA, but as we applied this methodology to the rest of the ownership we found that the topography and landslide patterns outside of the Coastal Klamath are not quite as variable. Based on this evaluation we did not apply SSSMUs to the remaining HPAs. Instead we analyzed the landslide data in relation to the HPA and HPA groups. A comparison of landslides to bedrock units was also made, however no observable correlations were found that would contribute to redefining the SSS zones.

Results

A total of 1,676 landslides were analyzed for our evaluation of the SSS prescriptions in the final phase of the SSS Delineation project. The SSSMUs were not applied in the final phase of the analysis. We did however analyze the landslide data in relation to the HPAs and found three relatively distinct groups that stood out. Our final HPA groupings include revised SSS prescriptions for the Coastal Klamath HPA (completed in 2011), Smith River HPA, Interior Klamath HPA, and together the Korbel and Humboldt Bay HPA groups (minus the Interior Klamath, which was originally part of the Korbel HPA Group). As a result the original HPA groups outlined in the AHCP have been revised and the new groupings are shown in table 2. The revised SSS prescriptions are specific to each of these groups.

Table 1—Revised HPA groups

	Revised HPA groups
HPA group	**HPAs**
Smith River	Smith River
Coastal Klamath	Coastal Klamath
Interior Klamath	Interior Klamath
Korbel	Coastal Lagoons, Little River, Redwood Creek, North Fork Mad River, Mad River, Humboldt Bay, Eel River

Slope Distances

The initial slope distance thresholds were determined by evaluating the maximum distance from the watercourse to the main scarp of all landslides reviewed with a total cumulative sediment delivery volume of 60 percent. The same cumulative volume value of 60 percent was used to determine the revised SSS slope distances. The cumulative volume of delivered sediment versus landslide distances from crown to watercourse is shown in fig. 5(a-c). The revised SSS slope distances have been calculated and a summary of the results are shown in table 3.

Table 2—Revised default SSS slope distances

	Revised SSS slope distances m (ft)		
HPA Group	**Class I**	**Class II-2**	**Class II-1**
Smith River	30 (100)	23 (75)	24 (80)
Interior Klamath	59 (195)	30 (100)	27 (90)
Korbel	41 (135)	34 (110)	32 (105)

Slope Gradient Thresholds

Slope gradient thresholds were based on the minimum slope gradient associated with all landslides within a cumulative volume of delivered sediment of 80 percent. In the initial study the slope thresholds were lumped together for all watercourse types. For the most part there was little variance of slope thresholds between watercourse types observed at that time. We found this to be true again as we assessed the remainder of the ownership. As a result we grouped the revised slope gradients for all watercourse classes by prescription area. The distribution of cumulative volume of sediment delivered versus landslide slope gradients are shown in fig. 5(d) and a summary of the slope gradient thresholds is shown in table 4.

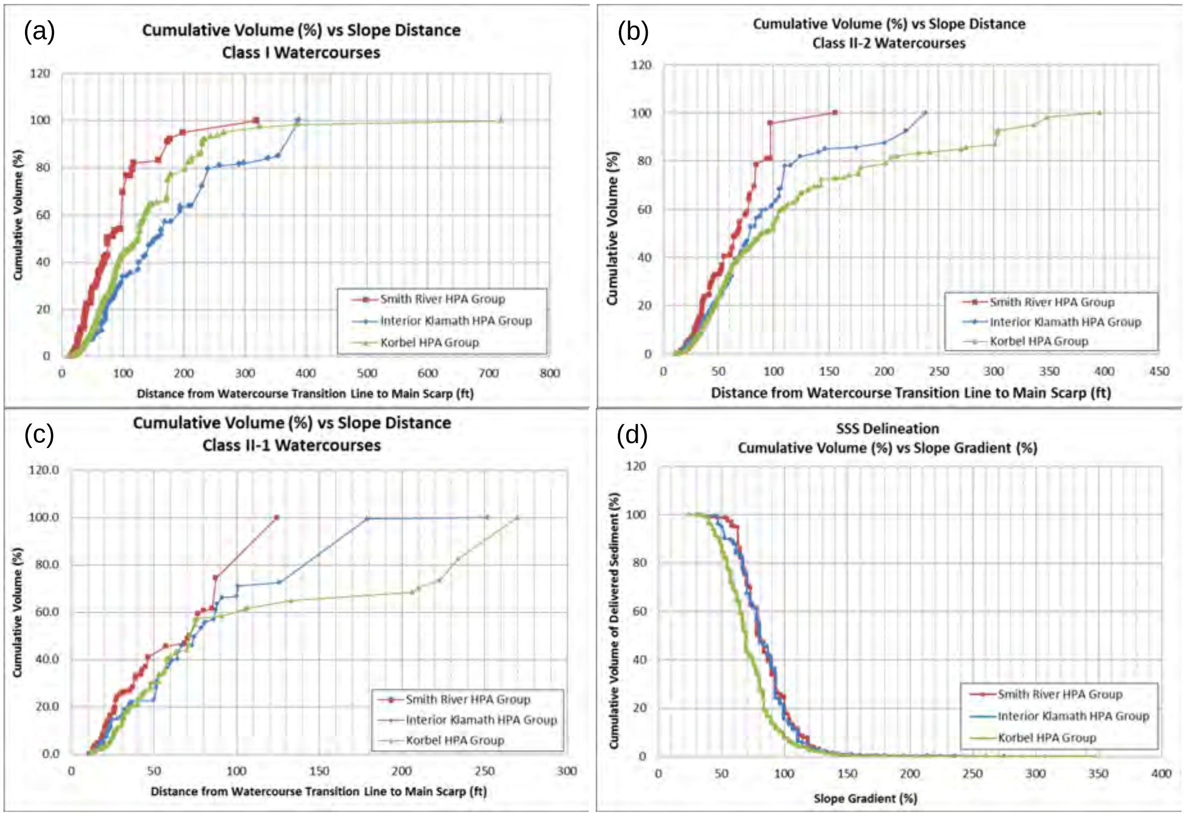

Figure 7—Cumulative volume vs slope distance; (a) Class I watercourses, (b) Class II-2 watercourses, (c) Class II-1 watercourses, (d) Cumulative volume vs slope gradient.

Table 3—New default SSS slope gradient thresholds for Smith River, Interior Klamath, and Korbel groups

Revised SSS slope gradient triggers (all watercourse classes)	
HPA group	**Slope gradient**
Smith River	65%
Interior Klamath	65%
Korbel	55%

Landslide Data

The distribution of landslides by HPA and watercourse type is shown in table 5. Taking into account the sampling rates, 10 percent of the lineal distance on the Class I's and 5 percent on each of the Class II's, the majority of the landslides observed occurred along the Class II-2 watercourses. As was the case in our work for the Coastal Klamath, fewer landslides were found along the Class I and II-1 watercourses throughout the remainder of the ownership. This is a logical observation as the Class II-2 streams are erosion and transport reaches that are characterized by higher flows than the Class II-1 streams. They are generally characterized as having much steeper stream gradients than the Class I streams which are often depositional reaches. As a result, these streams and adjacent hillside areas are subject to more erosion and down cutting, so we would expect there to be a higher occurrence of landsliding compared to the other watercourse types.

Table 4—Distribution of the number of landslides by watercourse class and HPA grouping

Watercourse class (% sampled)	Number of landslides by watercourse classification Smith River HPA	Interior Klamath HPA	Korbel HPA group
Class I (10%)	100	123	380
Class II-2 (5%)	106	165	585
Class II-1 (5%)	54	63	100

When assessing the effectiveness of the preventative “SSS” prescriptions, GDRCo, U.S. Fish and Wildlife Service and National Marine Fisheries Service agreed on a value that would exceed the value identified under historical conditions and settled on 70 percent cumulative volume as an achievable goal. In general we saw that the number of landslides decreased over time. Based on our review of aerial photographs from 1941 to 2009 for the SSS Delineation study, 86 percent of the landslides observed on aerial photographs occurred between the 1941 and 1988 aerial photo sets resulting in 6.65 slides per year, 14 percent of the slides were observed in photographs after the 1988 photo set resulting in 2.57 slides per year, and only 2 percent occurred after the 1997 set resulting in 0.75 slides per year. Our preliminary estimates on landslide erosion rates show similar decreases over time. Current erosion rates are only a fraction of what they were in the 1960s and 1970s when they were at the peak historically (0.06 m^3/ha/yr (28 U.S. Tons/mi^2/yr) since the implementation of the AHCP in 2007 compared to 1.5 m^3/ha/yr (684 Tons/mi^2/yr) in the 1960’s and 1.79 m^3/ha/yr (820 Tons/mi^2/yr) in the 1970’s).

Discussion

This study did not include the data collected in the initial study. However each of the areas of the initial study had an opportunity to be resampled as part of the SSS Delineation based on the random selection process. In fact a few areas were resurveyed. Data from the initial study were significantly limited. The sample areas from the study were not random. Volume estimates for the initial study were based solely on ocular estimates. Additionally, slide locations were mapped onto transparent overlays of aerial photographs. Slide locations from these data would be difficult to determine and transfer to our GIS as the overlays used for mapping did not capture any sort of reference markers such as the photograph fiducials. As a result we did not include these data in this project.

Table 6 outlines a comparison of the initial maximum SSS buffer distance and slope gradient triggers with the revised SSS buffer criteria. The majority of the maximum buffer widths decreased compared with the preliminary prescriptions while nearly one third increased in width. As for the slope triggers, just over half of the slope gradients decreased, nearly half had no change, and a few, in the Coastal Klamath, resulted in increased slope gradient triggers. Modeling potential SSS areas across the ownership, we estimate the new prescriptions will reduce the amount of SSS applied to streamside slopes by roughly 20 percent compared with the initial default prescriptions. A decrease was anticipated as the initial default prescriptions were created from a dataset that was intended to produce conservative values in the interim, until a full evaluation of steep streamside slopes could be completed under the SSS Delineation project.

The resulting slope distances and slope inclinations presented in table 6 highlight a need for flexibility of the prescriptions across much of the landscape. However, geomorphic characteristics were fairly homogenous in the southern portion of the ownership. As discussed earlier we explored the use of the SSSMU’s across these areas but found little to no variation in the TRI model data or the landslide data. Additionally we attempted to further subdivide the Korbel HPA Group even more but found no significant subdivision within the HPAs that would justify separate SSS prescriptions. Although the bedrock geology varies throughout the Korbel HPA Group, average slope inclinations are fairly consistent. In fact, we evaluated the average slope inclination for each of the HPAs within the Korbel HPA Group. For this evaluation we looked at all slopes over 20 percent in order to eliminate low lying slopes such as streams and prairies where landslides are rarely found. The lowest slope gradient we observed a landslide on was 24 percent. Reviewing these slope gradients we found that the average slope inclination varies by no more than four, between any given HPA. Additionally the standard deviation of the average slope between these HPAs is also similar and does not vary by more than 4 percent. This shows that slope inclinations are fairly consistent in these areas. Hence, we may expect landslide run out lengths also should be similar across these areas. Given that we are assessing shallow landslides that largely involve fine grain materials such as colluvium and regolith; we can expect that the physical characteristics such as length and failure inclination would be fairly consistent and thus result in similar SSS prescriptions.

Table 5—Comparison of initial default SSS prescriptions to the revised SSS prescriptions

SSS maximum slope distances [m (ft)] and minimum slope gradient thresholds				
Results from previous work: coastal Klamath HPA group				
SSSMU	Class I	Class 2-2		Class 2-1
1	72 (240) @ 65%	34 (110) @ 70%		41 (135) @ 75%
2	130 (425) @ 75%	59 (195) @ 85%		
Initial default buffers	145 (475) @ 70%	61 (200) @ 70%		30 (100) @ 70%
Results from current work				
Smith River HPA group				
Watercourse c	Initial distance (ft)	Revised distance (ft)	Initial slope (%)	Revised slope (%)
C-I	46 (150)	30 (100)	65	65
CII-2	30 (100)	23 (75)		
CII-1	23 (75)	24 (80)		
Interior Klamath HPA group				
Watercourse class	Initial distance (ft)	Revised distance (ft)	Initial slope (%)	Revised slope (%)
C-I	61 (200)	59 (195)	65	65
CII-2	61 (200)	30 (100)		
CII-1	23 (75)	27 (90)		
Korbel HPA group[a]				
Watercourse class	Initial distance (ft)	Revised distance (ft)	Initial slope (%)	Revised slope (%)
C-I	61 (200)	41 (135)	-	-
CII-2	61 (200)	34 (110)	-	-
CII-1	23 (75)	32 (105)	-	-
Korbel	-	-	65	55
Humboldt Bay	-	-	60	

[a] Korbel HPA group includes; Mad River, North Fork Mad River, Little River, Coastal Lagoons, Redwood Creek, Humboldt Bay and Eel River HPAs.

Conclusions

The revised steep slope prescriptions were submitted to federal agencies in December 2014 and were successfully incorporated into the AHCP in January 2015.

Our analysis of landslides in the Smith River HPA, Interior Klamath HPA, and the Korbel HPA Groups has resulted in changes to both the slope distance and slope gradient criteria associated with the initial default SSS prescriptions across these areas. These new criteria offer specific protections to each Hydrologic Planning Area. The new default SSS buffers present a reduced encumbrance across the GDRCo ownership in comparison with the initial default prescriptions. A result that was not unexpected, as the initial study was intended to provide a rapid assessment with interim results that were intentionally biased towards areas known to be steep and with high concentrations of recent landsliding. Our sampling methods significantly reduced bias and spatially distributed the samples across the HPAs such that we were able to produce a robust data set that more accurately characterizes the geomorphic conditions of the region as they pertain to streamside landsliding.

The goal of the SSS buffer prescriptions is to achieve a 70 percent reduction in management-related sediment delivery from landslides compared to delivery volumes from landslides in appropriate historical clearcut reference areas. The significant reduction in landslide occurrence observed in aerial photographs over the last 18 years suggests that this goal set forth in the AHCP is achievable. It is our judgment that these new default buffer prescriptions will help meet this goal. If we consider that the SSS landslide data set consists almost entirely of historical landslides that occurred under historical logging practices that no longer exist (i.e., historical tractor logging, steam donkey logging, reduced or no riparian management zones), we expect to see a natural decrease in landslide related sediment over time and our preliminary review of historical erosion rates supports this. Additionally, since the implementation of the AHCP in 2007, much has been done to improve management practices such as implementing our AHCP Riparian Management Zones and road management measures, as well as adopting less impactful logging methods such as shovel yarding. Therefore we expect that the revised prescriptions will achieve the SSS goal identified in the AHCP due to our site specific preventative landslide prescriptions in conjunction with the much improved forest practices currently applied to this property. It should be emphasized that the effectiveness of these new prescriptions will be tested through the SSS Assessment study, which will be reviewed by an independent scientific review panel and modified as necessary through the adaptive management process of the AHCP.

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Development and Implications of a Sediment Budget for the Upper Elk River Watershed, Humboldt County[1]

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Abstract

A number of watersheds on the North Coast of California have been designated as sediment impaired under the Clean Water Act, including the 112 km^2 upper Elk River watershed that flows into Humboldt Bay just south of Eureka. The objectives of this paper are to: 1) briefly explain the geomorphic context and anthropogenic uses of the Elk River watershed; 2) develop a process-based sediment budget for the upper watershed, including an explicit assessment of the uncertainties in each component; and 3) use the results to help guide future management and restoration. Natural (background) sediment inputs are believed to be relatively high due to high uplift rates, weak Miocene-Pliocene bedrock materials, steep slopes, high rainfall, and resulting high landslide frequency. The primary land use in the upper watershed is industrial timberlands, and intensive logging in the 1980s and 1990s greatly increased sediment production rates and downstream aggradation. Road improvements and major changes in forest practices have caused anthropogenic sediment inputs to drop by roughly an order of magnitude since the 1990s. Suspended sediment yields plotted against annual maximum peak flows indicate a decline since 2013, suggesting that the legacy pulse of sediment is now moving into the lower portions of the watershed and that improved management practices are having a beneficial effect. Recovery and restoration in the lower watershed is far more challenging as very low channel gradients cause sediment deposition, in addition to development of the floodplain for agricultural and residential use, forcing the river into a single-thread channel, and positive feedback loop between reduced flow velocities, aggradation, and dense vegetative growth in portions of the mainstem channels.

Introduction

The 152 km^2 Elk River watershed flows into Humboldt Bay just southwest of Eureka, California. It can conceptually be divided into a steep, forested upper watershed that comprises the majority of the basin (112 km^2) and a more developed lower watershed with a wide, low-gradient alluvial valley bottom and floodplain (fig. 1). Designated beneficial uses of particular concern include municipal and agricultural water supply, endangered coldwater fisheries habitat, and water contact recreation (TT 2015). Downstream flooding and high turbidity are critical concerns for the residents, and are believed by many of the residents to have been greatly aggravated by upstream forest management activities. These problems led the North Coast Regional Water Quality Control Board (NCRWQCB) to designate the entire Elk River watershed as impaired for sediment in 1998 under Section 303(d) of the Clean Water Act, and to identify an affected reach that spans the upper and lower watersheds (fig. 1).

There is considerable uncertainty and debate over the relative magnitude of past and current anthropogenic sediment inputs relative to natural erosion rates, and hence the extent to which past and present forest management activities are increasing downstream flooding and impairing the

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designated beneficial uses. It also is not clear to what extent flooding in the affected reach is exacerbated by human-induced changes to the main channels and valley floor. An assessment of natural sediment sources, anthropogenic sediment sources, and other modifications to the Elk River watershed has direct implications for the potential success of different watershed restoration options. Since an extensive hydrodynamic modeling study is underway to evaluate water and sediment conveyance in the affected reach and lower watershed (TT 2015), our focus is on the production and delivery of sediment from the upper watershed to the affected reach. The specific objectives of our ongoing study are to: 1) develop a sediment budget for the upper Elk River watershed, including inputs, outputs, and storage; 2) identify the greatest sources of uncertainty and how these might be resolved; and 3) discuss the implications of the sediment budget for downstream water quality, nuisance flooding, and future watershed recovery. This paper is an initial summary of our work.

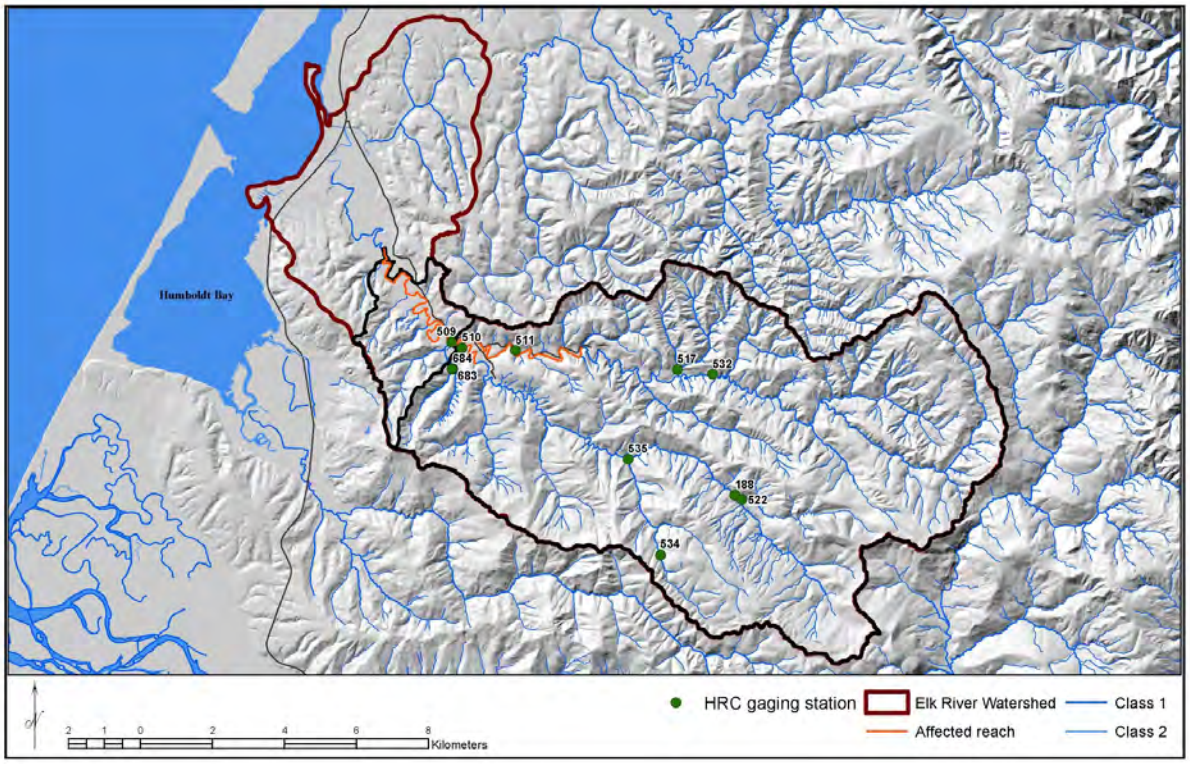

Figure 1—Map of the Elk River watershed with the affected reach in orange. The labelled green circles show the gaging stations that have been or are being operated by Humboldt Redwood Company. The upper watershed as defined in this paper is the area above the lowest gaging station (509), while the upper watershed for regulatory purposes is 8 km^2 larger as it includes all of the area draining to the affected reach.

Sediment Budget

A sediment budget is an accounting of sediment sources within a watershed, sediment outputs from the watershed, and changes in storage over a specified period of time (equation 1):

Inputs = Outputs + Change in storage.

For this paper the units for each component are mass (Megagrams or metric tons) per square kilometer per year, and multiple techniques are often needed to constrain and estimate each component. Each technique has an inherent spatial and temporal scale and uncertainty, so care must be taken to define the data and the extent to which these can be extrapolated in both time and space (Dietrich and Dunne 1978, Reid and Dunne 1996, Smith et al. 2011). We are fortunate because Humboldt Redwood Company has operated a minimum of 10 gaging stations in the Elk River watershed that measure both discharge and suspended sediment loads as part of their Habitat Conservation Plan. The gaging station that is furthest downstream (509) is on the mainstem just below the confluence of the North and South Forks, and this sets the downstream boundary for our sediment budget (fig. 1). Suspended sediment loads have been measured at station 509 since water year (WY) 2003, and the data from this station provides the output term in our sediment budget. The watershed area above station 509 is 112 km^2.

Sediment Inputs

For scientific and restoration purposes, sediment inputs in the upper Elk River watershed can be divided into: 1) natural (or background) sources; 2) inputs resulting from past timber management (1860s through the 1990s), which are designated as “legacy” sources in this paper, and 3) sediment inputs from forest management following major shifts in forest and road management practices beginning in 2000. Each of these sources is briefly discussed in the following sections along with their respective uncertainties.

Natural Sediment Sources

The magnitude of natural sediment sources is largely a function of the geologic and geomorphic conditions in the upper watershed. The majority of the upper watershed is underlain by the undifferentiated sedimentary Wildcat Group of Miocene-Pliocene age (fig. 2). This sequence of marine siltstones and fine-grained sandstones unconformably overlies Franciscan Complex bedrock, specifically the Jurassic-aged Central belt and the Late Cretaceous to Early Tertiary Yager terrane (Stillwater Sciences 2007). Undifferentiated Wildcat Group sediments are poorly indurated and considerably softer and more erodible than the low grade metamorphic sediment associated with the Central belt and Yager terrane. In the westernmost portion of the upper watershed the Wildcat Group is capped by the Hookton Formation. This is a non-marine, mid- to late-Pleistocene sedimentary formation that in the Elk River is dominated by silts intermixed with sands and clays. The Hookton Formation has a high propensity for natural failure and is highly erodible. Valley floors adjoining the main channels are mantled with a variably thick package of Quaternary-aged alluvial deposits (fig. 2).

Figure 2—Geologic map of the Elk River Watershed (TT 2015).

Estimates of natural or background erosion rates are extremely variable, but tectonic uplift, weak bedrock, and high precipitation rates all suggest that natural erosion rates should be relatively high. The proximity of the Elk River watershed to the Mendocino Triple Junction and Little Salmon Fault system produces uplift rates in the upper watershed that are on the order of 0.5 mm yr^{-1} (Balco et al. 2013, Stallman and Kelsey 2006). If erosion rates equal uplift rates, this uplift rate would convert to an erosion rate of roughly 1200 Mg km^{-2} yr^{-1} assuming a bedrock bulk density of 2.5 Mg m^{-3} (Bennet et al. 2015) and a 5 percent dissolution loss, which is comparable to the value in the nearby Eel River watershed (Milliman and Farnsworth 2011). This high erosion rate is consistent with the high mean sediment yields of 1100-3700 Mg km^{-2} yr^{-1} calculated for five North Coast watersheds with substantially unaltered flows (Andrews and Antweiler 2012). Alternatively, Stallman and Kelsey (2006) estimated a denudation rate of 0.10 mm yr^{-1} over the last 330 to 590 thousand years for the immediately adjacent Ryan Creek watershed, which converts to only about 250 Mg km^{-2} yr^{-1}.

Denudation rates over several thousand years also can be estimated using the cosmogenic isotope beryllium-10 (^{10}Be). Balco et al. (2011) presented a series of ^{10}Be data from a variety of watersheds along the North Coast, and denudation rates peak at more than 1 mm yr^{-1} between 40 and 41 degrees north latitude. This peak can be attributed to the northward propagation of the Mendocino Triple Junction. The centroid of the Elk River watershed is at 40.7^{o} N, which is at the leading edge of this peak. We have collected alluvial sand samples from the Elk River watershed for ^{10}Be analysis, but these data are not yet available. Mapping from the California Geologic Survey shows that most of the upper Elk River watershed is subject to mass movements (Marshall and Mendes 2005). While additional work is needed to better quantify the natural erosion rate, these data and a report from the California Geological Survey (Bedrossian and Custis 2002) strongly indicate that the long-term natural erosion rate is at least 3 to 14 times the values of 60 Mg km^{-2} yr^{-1} (about 0.024 mm yr^{-1}) estimated in the sediment TMDL (NCRWQCB 2016) and 94 Mg km^{-2} yr^{-1} estimated by HRC in their watershed analysis (HRC 2014) (table 1).

Legacy Sediment Sources

Estimates of legacy and current sediment sources are facilitated by the fact that nearly 84 percent of the upper watershed is industrial timberlands and another 15 percent is protected public lands. Only about 1 percent is in other private or public ownership, and land use for these parcels includes residences, agricultural use such as orchards or pasture, timber harvest, and unmanaged. Sediment source estimates have been made for the industrial timberlands from 1955 to 2011 (NCRWQCB 2013, TT 2015), but there are no comparable data for the other private lands.

Legacy sediments originate from materials deposited from forest management activities prior to 2000 that were stored within the watershed (e.g., in floodplains, terraces, or colluvial hollows), or sediment coming from sources that originated from management activities prior to 2000 (e.g., landslides that have not yet stabilized). This definition of legacy sources stems from the strong consensus that sediment production rates from forest management activities and roads greatly declined beginning around 1999 due to a marked shift in practices, particularly reduced logging on unstable slopes, stormproofing and decommissioning roads, restrictions on wet weather timber hauling, increased protection of riparian zones, and stronger controls on tractor logging operations (HRC 2014). In 2008 Humboldt Redwood Company, which owns 76 percent of the upper watershed, shifted from even-aged silviculture (clearcuts) to uneven-aged or selection harvesting (HRC 2014). About the same time the other major timber company began using shovel logging in areas suitable for ground-based timber harvest, and this generally minimizes ground disturbance and maximizes residual ground cover.

Table 1—Estimated sediment inputs to streams in Mg km^{-2} yr^{-1} from the sediment TMDL for three periods from 1988 to 2011 (TT, 2015), the values for 2001-2011 from the watershed analysis conducted by Humboldt Redwoods Company (HRC), and the estimated trends since 2011 relative to the HRC 2001-2011 values (upward arrow indicates an increase, an equal sign indicates little or no change, and a downward arrow indicates a decrease) (The values in parentheses in the column labelled HRC (2014) are the percentages that HRC apportions to legacy sources)

	Sediment TMDL (TT, 2015)			**HRC (2014)**	**Est. trend**
Natural sources	1988-97	1998-2003	2004-2011	2001-2011	2012-2016
Bank erosion	4	4	4	64	=
Streamside landslides	13	13	13		
Bank erosion: deep-seated	49	53	37		
Shallow landslides	15	15	15	2	=
Deep-seated slides	1	1	1	1	=
Creep	0	0	0	28	=
Sub-total (Mg/km2 yr)	**82**	**86**	**71**	**94**	=
Land use/management					
Low order channel incision	10	11	7	0	=
Bank erosion/streamside landslides	107	118	78	25 (89%)	=
Road-related landslides	150	6	12	20 (91%)	↓
Shallow landslides	98	42	2	0	↑
Untreated anthropogenic sites	32	27	19	0	↓
Post-treatment	0	4	12	0	↓
Skid trails	6	11	7	6 (100%)	↓
Road surface erosion	67	27	11	28 (63%)	↓
Surface erosion: harvest units	2	3	2	10 (77%)	↓
Sub-total (Mg/km2 yr)	**474**	**249**	**151**	**89 (81%)**	↓
Management related (%)	85	74	68	48	↓

It is extremely difficult to quantify or separate many of these legacy sources from natural sources, particularly: headwater incision of old sediment deposits; erosion from old skid trails, roads, and abandoned crossings; and subsurface erosion from soil pipes under or through the anthropogenic sediment deposited in headwater areas. A report from the adjacent Freshwater Creek watershed suggested that the first cycle of logging did increase drainage densities and channel scour, and estimated that this generated approximately 15,000 Mg km^{-2}, primarily from first- and second-order streams (PWA 1999). The HRC 2014 Watershed Analysis estimated that legacy sources are equivalent to 80 percent of background sources or about 80 Mg km^{-2} yr^{-1}.

Sediment From Current Management Activities

Process-based studies indicate that improved management practices have reduced sediment inputs from the industrial timberlands by at least a factor of four for 2004 to 2011 relative to 1988 to 1997 (NCRWQCB 2016, TT 2015) (table 1). The actual reduction in management-related sediment sources is almost certainly larger for several reasons. First, streamside landslides and bank erosion accounted for 52 percent of the anthropogenic sediment sources in the sediment TMDL (TT 2015), but this value was based in large part on the assumed threefold-difference in drainage densities between managed areas and unmanaged areas; field investigations have documented that drainage densities in managed areas are about 6 km km^{-2} rather than the 10 km km^{-2} that was used for calculations in the TMDL. An extensive survey of more than 40 km of different-order streams in 2012 found that erosion voids left by streamside landslides and bank erosion were due to unstable geology and natural flow deflection (i.e., deep seated features being eroded at their toes and large woody debris, respectively) (SHN 2013). The conclusion was that “causal mechanisms due to recent management were virtually non-existent” (SHN 2013, p. 6). These and other data suggest that the vast majority of the bank erosion and streamside landslides that were attributed to management should in fact be considered part of the natural sediment load (table 1).

Second, landslide occurrence has greatly decreased (table 1). Landslide-related sediment loads for 2003 to 2011 were heavily influenced by the numerous landslides in water year (WY) 2003 resulting from the record 24-hour rainfall of 172 mm at Eureka. Annual helicopter surveys and aerial photograph analyses show a continuing decline as the mean sediment delivery from landslides (table 1). From WY 2010 to WY 2016 mean sediment delivery from landslides further decreased to less than 1 Mg km^{-2} yr^{-1}. Notably, this period did include two drought years, and nearly all of the recent landslides are road-related rather than from harvested areas (table 1). Third, road sediment production and delivery has been greatly reduced by continuing efforts to upgrade road crossings, rock and disconnect roads, and road decommissioning (table 1), with HRC spending an estimated $7 million since 1999 to decommission more than 20 percent of their roads and stormproof nearly 80 percent of the remaining 340 km. Fourth, surface erosion and sediment delivery from harvest units has been largely eliminated by the changes in harvest techniques and increased protection of riparian zones; these have greatly reduced the amount of bare ground, soil compaction, and surface disturbance.

The net result is that sediment inputs from forest management have further declined since 2011 and are probably only about 20 to 40 Mg km^{-2} yr^{-1} (HRC 2014), plus perhaps another 20 Mg km^{-2} yr^{-1} of short-term inputs as a byproduct of treating those legacy sites that are accessible and feasible to rehabilitate (TT 2015) (table 1). The latter inputs should be considered as a short-term cost in exchange for eliminating both chronic legacy sources and legacy sources that might fail and cause much larger sediment inputs.

Sediment Outputs and Sediment Storage

The overall quality of the discharge, turbidity, and suspended sediment data are very good. In each year about 100 to 450 suspended sediment samples are collected at each gaging station from stage-triggered automated pump samplers. Annual suspended sediment yields are calculated by multiplying discharge times storm-specific turbidity-suspended sediment relationships (Lewis and Eads 2009). From WY 2003 to 2016 the mean annual suspended sediment yield at station was 260 Mg km^{-2} yr^{-1} with a high interannual variability (c.v. = 1) and a range from 960 Mg km^{-2} yr^{-1} in the WY with the highest peak flow on record (2003) to just 20 Mg km^{-2} yr^{-1} in the WY with the second lowest peak flow on record (2014). Most of the annual sediment load is transported during the largest one or two storms, and 83 percent of the annual variability in suspended sediment yields can be explained by the instantaneous annual maximum peak flow (fig. 2). There are no bedload data, but bedload in five North Coast rivers was estimated to range from 1 percent to 10 percent of the calculated suspended loads (Andrews and Antweiler 2012). Data from Caspar Creek, which is the closest analog to Elk River with reliable bedload data and has similar geology to the upper portions of the Elk River watershed, indicate that bedload equals about 50 percent of the suspended load (Cafferata and Reid 2013). If we assume that bedload is half of the suspended load, the total annual sediment yield at station 509 would be around 400 Mg km^{-2} yr^{-1}, or a watershed-scale denudation rate of 0.14 mm yr^{-1}.

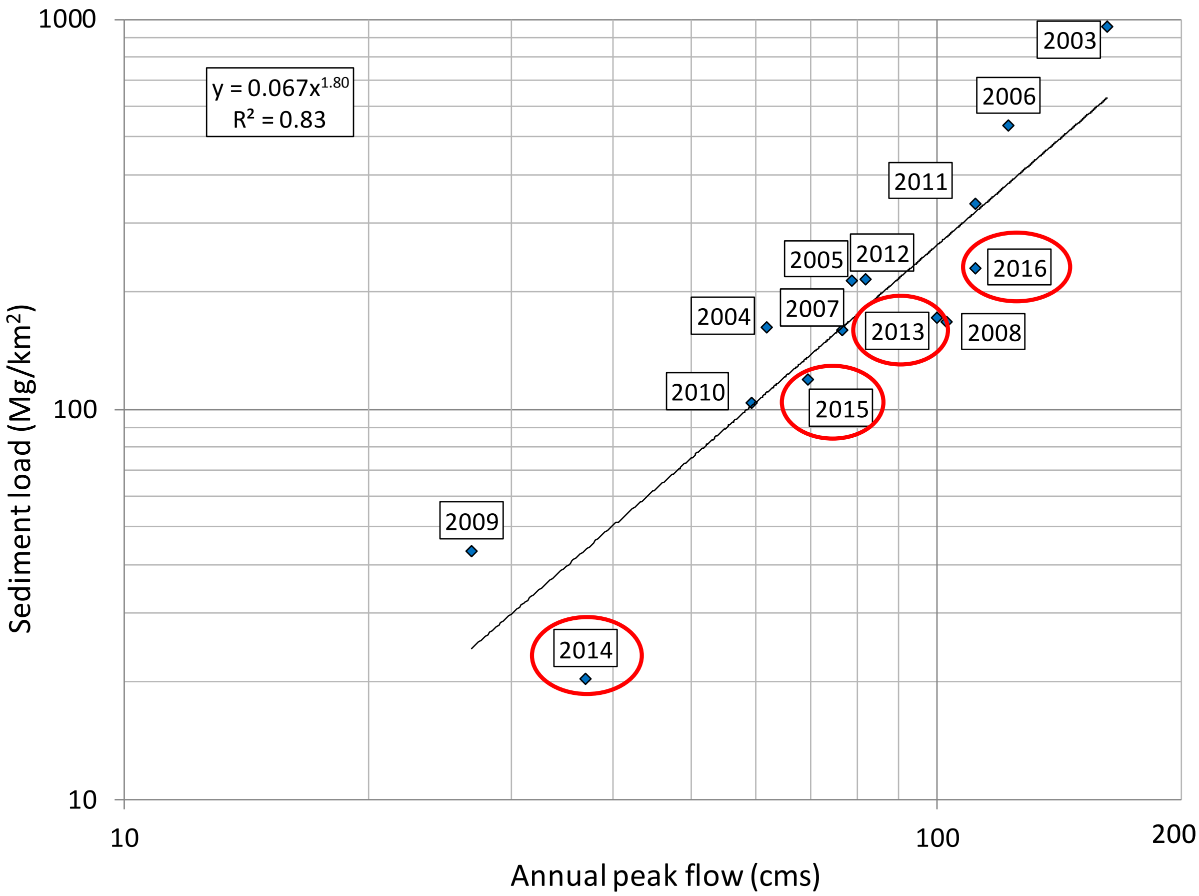

Figure 2—Plot of the annual suspended sediment loads at station 509 versus the annual instantaneous maximum peak flow. Note that each year since 2013 falls below the regression line developed with all the data, indicating a reduction in the annual sediment load for a given peak flow.

The difference between sediment inputs and outputs is sediment storage (equation 1). Storage from natural, legacy, and current sediment sources is very difficult to quantify and hence another major source of uncertainty in the sediment budget. Simply rearranging equation 1 to calculate storage as the difference between sediment inputs and outputs also is problematic given the uncertainty in sediment inputs.

Ferrier et al. (2005) noted relatively low mean sediment residence times and hence low storage capacities in Redwood Creek and the North Fork of Caspar Creek. Conversely, the 2012 survey of streambank erosion and streamside landslides in the upper Elk River watershed noted that "stream valleys tend to have broad cross sections with wide valley bottoms" and that this is in contrast to the steeper, more incised valleys in the upper Eel, Bear, and Mattole watersheds (SHN 2013, p. 5). The lower portion of the main channels in the Elk River watershed are clearly depositional zones as indicated by the sharp change in gradient and associated widening of the valley bottom. The valley bottoms along the mainstem and extending up the North and South Forks are in the 100-year flood zone (fig. 4). Technical documents supporting the sediment TMDL estimated that 640,000 yd^3 or nearly 500,000 m^3 of sediment had accumulated in the affected reach since the late 1980s or early 1990s (NCRWQCB 2013, TT 2015), but this value is highly uncertain given the large spatial extrapolation from very limited data and probably substantially overestimated given more recent analyses of the cross-sectional changes over time and measured sediment yields at station 509.

The magnitude of storage is of critical concern for determining the potential for watershed recovery. While the high interannual variability of suspended sediment yields makes it difficult to identify statistically significant trends in the 14-year record, the suspended sediment yields for each of the last 4 years fall below the regression line shown in fig. 3. Qualitative observations and some of the monitoring data on bed material particle size and residual pool depth (MacDonald 2014) indicate a coarsening of the streambed in the upper portions of the affected reach. If these trends are confirmed, this would suggest that the pulse of legacy sediment is finally starting to be removed from the upper watershed (Trimble 1999). Alternatively, if these trends are not confirmed, the bulk of current sediment yields are due to some combination of natural and legacy sediment sources given the generally recognized reduction in current anthropogenic sources (TT 2015) (table 1).

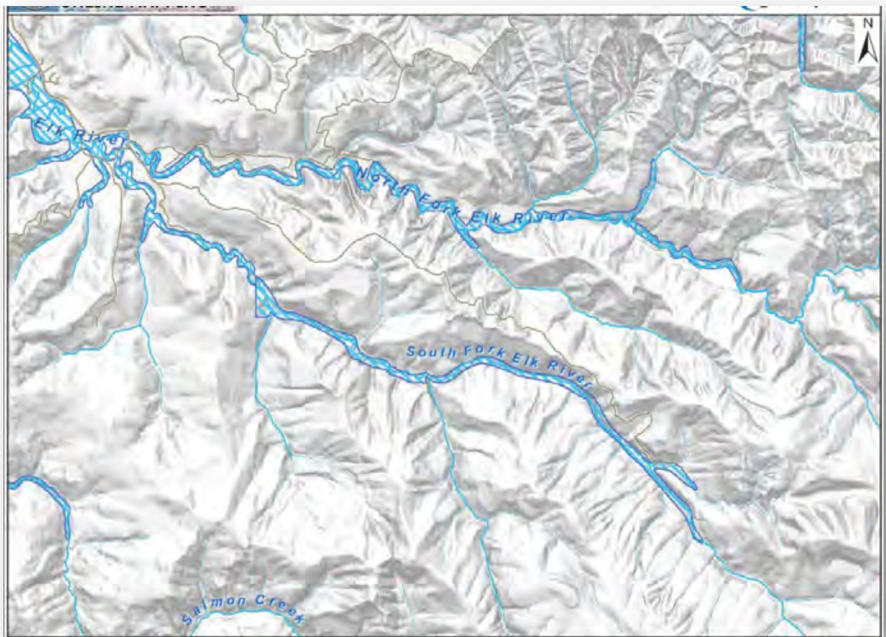

Figure 3—Map of the 100-year flood zone in the North and South Forks of the Elk River as mapped by the Federal Emergency Management Agency.

Implications for Restoration and Management

The three issues of greatest concern to resource management agencies and the local residents are: 1) the high turbidity and suspended sediment concentrations that limit the use of the Elk River for domestic and agricultural water supply during the winter months; 2) nuisance flooding of road access and residences; and 3) the effects of high turbidity and fine sediment deposition on salmonid spawning, feeding, and rearing. Data indicate that each of these issues has been exacerbated to some degree by human activities in the watershed, and the following will address each of these in turn.

First, the high natural erosion rates and fine-grained, weak rocks in the upper Elk River watershed lead to inherently high turbidity levels during the winter wet season compared to many other forested landscapes, even in the absence of any human disturbance. For example, turbidity levels in the undisturbed Little South Fork of the Elk River exceed the unfiltered drinking water standard of 5 NTU at least 10 times per year, and the mean duration of these exceedences is 31 days per year. Turbidity data from other undisturbed watersheds envelope the values from the Little South Fork (Klein et al. 2012). Extensive gravel filtration was required to maintain water quality when the Elk River was used as the major source of water for Eureka (Springer 1995). These data indicate that the Elk River will never be, and almost certainly never has been, able to meet the turbidity criteria for unfiltered municipal water supply. Industrial forest management, including the extensive road network, has increased turbidity levels in the rest of the Upper Elk River watershed, but the sediment budget indicates that this anthropogenic increase in sediment inputs is primarily due to legacy and stored sediment rather than current forest management (table 1). It is not clear to what extent the turbidity and associated suspended sediment concentrations limit agricultural pumping since agricultural users primarily pump river water during the growing season rather than during the winter (B. Alexandre, 2016, personal communication).

The most frequent nuisance flooding is associated with a 60-m section of road that dips below the bankfull channel on the North Fork ("Elk River flood curve"). Floodwaters begin encroaching on the roadway at about 20 $m^3 s^{-1}$, which is only about one-third of the estimated bankfull flow and mean annual flood. Hence this section of road is inundated approximately three to four times a year, which is a major concern because this road is the only access for some residents. While there are no data on aggradation over time at this location, this frequent flooding appears to be at least as much a function of poor road layout as any human-induced channel aggradation.

Flooding of residences and roads in the remainder of the lower basin is a function of both the inherent characteristics of the watershed along with human-induced alterations to the stream channel and valley floor. There is a very sharp decrease in channel gradient from the upper watershed to the affected reach, and the geologic map indicates that the valley bottoms are filled with Quaternary alluvial deposits. Nearly all of the valley bottom along the affected reach falls within the 100-year flood zone as designated by the Federal Emergency Management Agency (fig. 3). According to the classic text by Dunne and Leopold (1978, pages 599–608) "The channel is formed and maintained by the flow it carries but is never large enough to carry without overflow even discharges of rather frequent occurrence….The floodplain is indeed part of the river under storm conditions." Hence much of the valley bottom along the affected reach is by definition an active floodplain and naturally subject to flooding.

Historical accounts and aerial photos indicate severe human-induced changes to the main channel of the Elk River and adjoining floodplain (HRC 2014, MacDonald 2016). These include a loss of overflow channels and wetlands; confining the river to a single-thread channel; forcing the river through a tight series of unnatural right-angle bends; and dense vegetative growth within the channel. These changes probably have led to a positive feedback loop in which the reduced velocities and the subsequent aggradation facilitate vegetative growth in the channel, which further reduces channel capacity. Our preliminary estimate is that the stage associated with the historic bankfull flow of a little over 60 $m^3 s^{-1}$ has increased by around 0.8 m between 1967, when the gaging station operated by the U.S. Geological Survey ceased operating, and 1999 when HRC began cross-section measurements at nearly the same location. Annual cross-section data from 1999 through 2016 show a small increase in bankfull cross-sectional area and 0.4 m of thalweg incision since 1999. Given the biogeomorphic processes of deposition and colonization by vegetation and all the downstream changes, it is not clear whether further reductions in sediment inputs from legacy and current forest management will induce sufficient channel incision to reduce nuisance flooding.

Efforts to dredge the channel and/or remove some of the vegetation within or adjacent to the channel will be harmful to the threatened fish populations, and are ultimately short-term solutions given the low gradients and high natural sediment yields. Prior to European settlement the lower portion of the watershed was almost certainly a wetland with the Elk River flowing through a complex network of channels (HRC 2014) that would periodically avulse in response to aggradation. Restoration of the Elk River will necessitate a tough choice between maximizing conveyance, with the associated adverse effects on fish, or allowing a mechanism for the Elk River to reclaim part of the valley bottom while still protecting life and property and allowing some economic use.

Conclusions

The upper Elk River watershed has been designated as impaired for sediment due to the high turbidities, suspended sediment loads, and observed or inferred channel aggradation that has exacerbated nuisance flooding. The extent to which these problems can be eliminated depends in large part on how much of the high sediment loads are due to natural versus anthropogenic sediment sources. Natural erosion rates are believed to be relatively high given the uplift rate of around 0.5 mm yr^{-1}, and that the weak geology and high annual precipitation induces debris flows, landslides, and deep-seated earthflows. Nearly 84 percent of the upper watershed is designated as industrial timberlands and intensive logging—particularly the tractor-based logging in the 1980s and 1990s—greatly increased erosion rates and induced downstream aggradation.

Sharp changes in logging practices and a wide range of road improvements mean that sediment inputs from roads and current forest management have dropped by more than an order of magnitude. Both legacy sediment inputs and sediment storage are extremely difficult to quantify, but the recent drop in sediment yields as normalized by peak flows and the minimal aggradation over the last 15 years indicate that the legacy sediment pulse has at least partially passed through the 509 gaging station and is now moving downstream into the lower watershed. The implication is that sediment inputs from current forest management practices have declined to the point that water quality conditions are improving. Achieving the designated beneficial uses in the lower portions of the Elk River will be much more difficult given the extensive agricultural and residential use and the associated large alterations to the main channel and floodplain. Dredging and riparian vegetation removal may produce a localized, short-term increase in conveyance, but this will almost certainly have a negative impact on water quality and the endangered salmonids.

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Using Caspar Creek Flow Records to Test Peak Flow Estimation Methods Applicable to Crossing Design[1]

Peter H. Cafferata[2] and Leslie M. Reid[3]

Abstract

Long-term flow records from sub-watersheds in the Caspar Creek Experimental Watersheds were used to test the accuracy of four methods commonly used to estimate peak flows in small forested watersheds: the Rational Method, the updated USGS Magnitude and Frequency Method, flow transference methods, and the NRCS curve number method. Comparison of measured and calculated results for 10-year return-interval flows demonstrates that, under the conditions tested, the direct flow transference method provides the most reliable results if suitable data are available; results for 100-year flows show similar patterns. None of the methods consistently underestimated the values derived from the gaging record. This indicates that these methods are unlikely to result in an under-design of drainage structures with respect to flow capacity. However, design of stable stream crossings in steep forested areas also requires consideration for passage of sediment, woody debris, and fish, so estimation of required flow capacity represents only a first step in the design process.

Keywords: culvert sizing, flow estimation methods, forest hydrology, watercourse crossings

Introduction

The California Forest Practice Rules require that new or replaced watercourse crossings associated with commercial timber operations on non-federal forestlands in California be designed to accommodate the 100-year flood and its associated sediment and debris. Registered Professional Foresters (RPFs) must estimate the 100-year flood discharge using flow measurement records and empirical relationships; then they must determine if that estimate is reasonable based on actual channel cross-section measurements (CAL FIRE 2016). A variety of methods have been developed over the past 150 years to estimate peak flows in urban watersheds to aid in design of drainage structures (Tolland et al. 1998). However, estimating large peak flows in small, ungaged forested watersheds is difficult because these sites often have steeper slopes and higher infiltration capacities than the sites for which the estimation methods were originally developed.

Refinement of existing methods is a high priority, since appropriate design of stream crossings for roads in forested watersheds is critical for reducing sediment inputs to streams and for decreasing road maintenance and repair costs (Furniss et al. 1998, Weaver et al. 2015). Past monitoring work in California forestlands has shown that crossings are high-risk sites for sediment delivery to streams (Ice et al. 2004, Staab 2004).

The most direct way to test the validity of existing peak-flow estimation methods is to compare predicted and measured flows at stream gaging stations. Few small forested watersheds have gaging records long enough for such testing (Forest Service Stream-Simulation Working Group 2008), but long-term records are available from the Caspar Creek Experimental Watersheds. We use those data to test the accuracy of four methods commonly used to estimate peak flows in forested watersheds in

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the redwood region. This study expands on work reported by Cafferata et al. (2004) and Cafferata and Reid (2013), and updated by Cafferata et al.[4]

Study Site

The North Fork Caspar Creek Experimental Watershed (fig. 1) is located in the northern part of the California Coast Ranges southeast of Fort Bragg. Watershed research has been conducted in the North and South Forks of Caspar Creek since 1961 under a partnership between the U.S. Department of Agriculture Forest Service Pacific Southwest Research Station and the California Department of Forestry and Fire Protection.

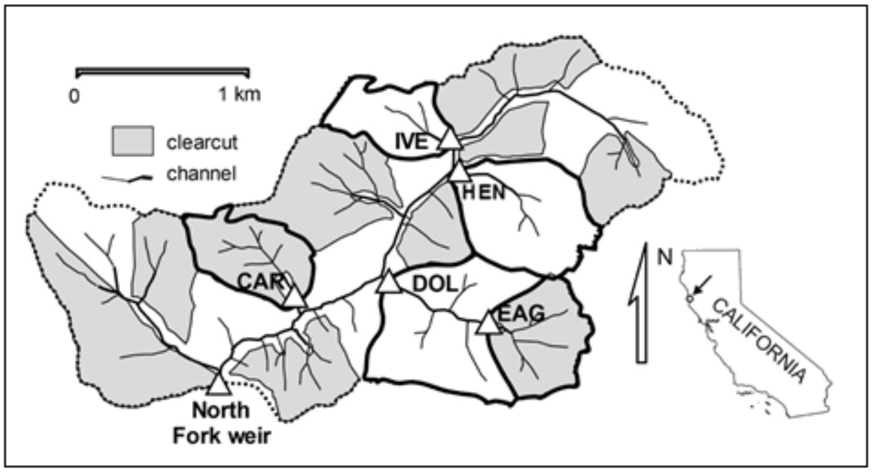

Figure 1—The North Fork Caspar Creek Watershed. Triangles indicate the locations of the gaging stations used for the analysis.

Caspar Creek drains 2,170 ha (5,362 ac), of which 1,958 ha (4,838 ac) are located in Jackson Demonstration State Forest. The 473 ha (1,169 ac) North Fork Caspar Creek watershed is underlain by marine sandstone and shale of late Cretaceous to early Cenozoic age and is incised into Pleistocene marine terraces. Elevations range from 82 to 317 m (270 to 1,040 ft). Soils are 0.5 to 2 m (1.6 to 7 ft) deep and are generally well-drained, with textures ranging from loams and sandy loams to very gravelly loams; most are in hydrologic groups B and C (Rittiman and Thorson 2006). Channel heads are generally present in catchments larger than 1.9 ha (4.7 ac). Approximately 95 percent of the average annual precipitation of 1,190 mm (47 inches) falls between October and April, and many tributaries are intermittent. Nearly half of the incoming precipitation runs off as stream flow, and snow is not hydrologically significant.

Coast redwood (*Sequoia sempervirens* (D. Don) Endl.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) are the dominant conifer species present in the Caspar Creek watershed; old-growth trees were logged from the mid-1860s to 1904. Two major watershed experiments have been carried out at Caspar Creek to study the hydrologic effects of second-growth harvesting of coast redwood and Douglas-fir, and a third is currently being implemented. The entire South Fork watershed was selectively logged from 1971 to 1973, and monitoring demonstrated the resulting influences on runoff volumes and peak streamflows. The North Fork experiment was designed to quantify the cumulative effects of clearcutting on suspended sediment, storm runoff volume, and peak flows; logging for the

[4] Cafferata, P.; Lindsay, D.; Spittler, T.; Wopat, M.; Bundros, G.; Flanagan, S.; Coe, D.; Short, W. Designing watercourse crossings for passage of 100-year flood flows, wood, and sediment (2016). Revised California Forestry Report No. 1. Manuscript in preparation. Sacramento, CA: California Department of Forestry and Fire Protection. 115 p.

second experiment took place from 1989 to 1992, and short-term results were reported by Ziemer (1998) and Lewis et al. (2001).

Methods

Flow Measurements

Large concrete weirs were constructed in 1962 to monitor streamflow and sediment at the North and South Forks of Caspar Creek; flow measurements span the period from 1962 to the present and will continue into the foreseeable future. In 1984, 13 gaging stations were installed in the North Fork watershed, eight of these in small headwater basins (< 40 ha, < 100 ac). Flow monitoring began in water year 1985 (1 Aug 1984 to 31 July 1985). Henry (1998) describes the sub-watersheds, monitoring methods used, and management practices. Flow was measured with wooden Parshall flumes (replaced with fiberglass Montana flumes in 2004), stilling wells, and pressure transducers. The five sub-watersheds having the longest flow records (control sub-watersheds HEN and IVE; clearcut sub-watersheds CAR and EAG, and partially clearcut sub-watershed DOL; fig. 1, table 1) were selected for testing the accuracy of four commonly used flow estimation methods. To provide field-based peak flow values for comparison to estimates, flow frequency analyses were conducted for these five sub-watersheds using the log-Pearson Type III distribution option in PeakFQ, a program available online from the U.S. Geological Survey.

Table 1—The Caspar Creek test sub-watersheds, and empirically derived and modeled estimates of the 10-year return interval (RI) flow in the sub-watersheds

	HEN	IVE	CAR	EAG	DOL
Area [ha (ac)]	39 (96)	21 (52)	26 (64)	27 (67)	77 (190)
Percent clearcut	0	0	96	99	36
Flow record duration (yr)	31	31	31	30	31
Years logged	--	--	1991	1990-91	1990-91
			(all values in cfs)		
10-yr flow	16	7.2	12	12.9	31.7
95% confidence limits	12.9-21.5	5.7-10.0	9.5-16.4	10.6-16.8	25.4-42.6
Flow model predictions, 10-yr flow					
Rational Method (C = 0.2)					
CA culvert practice equation	52	28	34	36	102
Airport drainage equation	29	16	20	21	47
BCMOE equation	24	15	17	18	40
Updated USGS	41	24	28	30	74
Flow transference method					
Standard method	25	15	18	18	46
Direct flow transference	19	10	13	13	37
NRCS WinTR-55	47	27	39	35	95
Tests of modifications					
Rational Method (storm lag)	8	4	6	6	17
Rational Method (burst lag)	10	4	7	7	19

Estimation of Peak Streamflows

We tested four methods that are often used to estimate peak flows associated with crossing design for small forested watersheds in California. These include (1) the Rational Method (using different runoff coefficients and methods for calculating times of concentration), (2) updated USGS Magnitude and Frequency Method equations for the North Coast region (Gotvald et al. 2012), (3) flow transference method (Waananen and Crippen 1977) and a variant of the method (Skaugset and Pyles 1991), and (4) the NRCS WinTR-55 small watershed hydrology program (NRCS 2009).

Rational Method

The Rational Method has been used by engineers for more than 150 years to predict peak runoff rates (Dunne and Leopold 1978). It was developed before long-term flow records were available, and this method remains widely used for estimating design floods in small ungaged watersheds because it requires few data and is easy to use. The Rational Method is often applied in urban watersheds, where most storm flow travels as overland flow on impermeable surfaces, and in small undeveloped watersheds. The method assumes that runoff is generated due to limited infiltration, an assumption that does not hold in many forested watersheds (Skaugset and Pyles 1991). Past studies have found that this method tends to overestimate design floods for non-urban basins (Tolland et al. 1998).

The Rational Method equation for the 100-year flood flow is $Q_{100} = CIA$, where Q_{100} is the predicted peak runoff from a 100-year storm[5] (cfs), C is the runoff coefficient, which may vary by storm size, I is the rainfall intensity for the 100-year storm (in/hr), and A is the drainage area (acres). Flood peak flows of other return intervals (e.g., 5, 10, 25, or 50-year peaks) can also be estimated by using an appropriate rainfall intensity and runoff coefficient.

To determine the rainfall intensity, one must (1) estimate the time of concentration (T_c), the time it takes water falling at the top of the watershed to reach the crossing location, and (2) use rainfall depth-duration-frequency data to identify the 100-yr rainfall for a storm duration equivalent to the T_c. A value for T_c can be estimated using one of more than 30 equations, including the California culvert practice equation (California Division of Highways 1944; modified Kirpich equation), Airport Drainage method (FAA 1970), and the BCMOE (1991) nomograph and equation (Gregori 2003). T_c calculations commonly introduce significant errors in peak flow estimation (Tolland et al. 1998). Selecting the appropriate runoff coefficient (C) is also difficult for small forested watersheds unless the value is locally calibrated, and C can vary with storm size as the relative importance of various flow sources changes (Dunne and Leopold 1978, ODOT 2014). The Rational Method should not be used for watersheds larger than 80 ha (200 ac) (Dunne and Leopold 1978), and is most reliable for those smaller than 40 ha (100 ac). Some authors recommend that it not be used in forested watersheds due to problems in estimating C and T_c (Skaugset and Pyles 1991).

USGS Magnitude and Frequency Method

The updated USGS Magnitude and Frequency Method, which replaces the method described by Waananen and Crippen (1977), is based on a set of empirical equations derived from precipitation and runoff data. Data from 630 stream gaging stations located throughout California were used to derive equations to predict peak flows for 2, 5, 10, 25, 50, 100, 200, and 500-year flow recurrence intervals for six regions of California (Gotvald et al. 2012). The equations were generated from watersheds with drainage areas ranging from approximately 10 ha (25 ac) to over 1,000,000 ha (2,500,000 ac). The 10-yr and 100-yr regression equations for the North Coast region are:

$$Q_{10} = 14.8\, A^{0.880}\, P^{0.696}$$
$$Q_{100} = 48.5\, A^{0.866}\, P^{0.556}$$

where Q_{10} and Q_{100} are the predicted 10-year and 100-year flood flows (cfs), A is the drainage area above the crossing (mi^2), and P is the mean annual precipitation (in). This method is easy to use, mean annual rainfall data are readily available, average standard errors of prediction for each flow recurrence interval in each region are provided, and flow estimates are based on discharge data from numerous, widely distributed locations, including large watersheds subject to rain-on-snow flow events. The primary disadvantage of this method is that it generalizes vast regions of the state, resulting in overestimation in some areas and underestimation in others (Cafferata et al. 2004).

[5] Runoff data are presented using English units for consistency with common usage by RPFs when designing watercourse crossings (i.e., cfs rather than cms).

Flow Transference Methods

If a gaging station is located on a stream that is hydrologically similar to that at the proposed crossing site, it is possible to adjust the estimate for a peak flow of a given recurrence interval at the gaged site to provide an estimate for the ungaged site simply on the basis of the drainage areas. For 100-year flow estimation (Forest Service Stream-Simulation Working Group 2008, Waananen and Crippen 1977),

$$Q_{100u} = Q_{100g}\ (A_u/A_g)^b$$

where Q_{100u} and Q_{100g} are the 100-year flows (cfs) at the ungaged and gaged sites, respectively; A_u and A_g are the drainage areas at those sites (mi^2), and b is the exponent for drainage area from the appropriate USGS Magnitude and Frequency equation (e.g., 0.866 for the 100-year flow in the North Coast Region). Flows of other return intervals and regions are calculated using their corresponding *b*-values, which are tabulated by Waananen and Crippen (1977). The gaging station records should span at least 20 years, and the peak flow at the gaged station must be estimated for the desired return interval (e.g., 10, 25, 50, 100-year). This method is most reliable where the drainage area of the ungaged site is between 50 and 150 percent that of the gaged site (Sumioka et al. 1998). When adequate records are available from a nearby gaging station, this method is expected to provide more reliable results than either the more general USGS Magnitude and Frequency Method or the Rational Method (Cafferata et al. 2004).

An alternative flow transference method can be used if the gaged and ungaged watersheds are relatively small (e.g., < 1000 ha or <~ 2,500 ac), are in close proximity, are hydrologically similar, and are within approximately one order of magnitude in size. The Direct Flow Transference Method (Skaugset and Pyles 1991) simply adjusts the value at the gaged station by the ratio of watershed areas:

$$Q_{100u} = Q_{100g}\ (A_u/A_g)$$

NRCS WinTR-55 Small Watershed Hydrology Program

Several computer programs are available that use the unit hydrograph approach to estimate flood flows for more complicated situations. One of the most widely used is the NRCS WinTR-55 program, which was developed for estimating runoff from small agricultural catchments and watersheds with other kinds of land uses. NRCS (2009) provides detailed information on the TR-55 program, which was constructed using the SCS curve number (CN) methodology. Curve numbers are defined as an "empirical rating of the hydrologic performance of a large number of soils and vegetative covers." They range from 0 to 100, and a spatially weighted average CN provides an index of storm runoff generation capacity. The maximum area for this method is 6,500 ha (~16,000 ac), and up to 10 sub-watersheds may be considered.

The NRCS WinTR-55 program is not commonly used for estimating design flows at forest road crossings, but it is often used to assess the potential hydrologic effects of timberland conversion projects (e.g., vineyard conversions) and has been accepted for routine use by some regulatory agencies. It is particularly useful when streamflow is regulated by upstream detention ponds or reservoirs. The main disadvantage for designing forest stream crossings is that curve numbers are not well defined for forested areas, and this often results in problematic estimates at such sites (Fedora 1987, Skaugset and Pyles 1991). Despite these problems, unit hydrograph analysis using SCS curve numbers is generally thought to provide reasonable estimates for predicting a relative change in peak flows due to land-use modification.

Applying Peak Flow Estimation Methods to the Test Watersheds

Each peak flow estimation method requires measurement or estimation of the values of various parameters. Several approaches are available for estimating the *C*-value for use in the Rational Method. A value of 0.3 or higher has often been recommended for use in woodland areas on loam or clay soils (Cafferata et al. 2004, Dunne and Leopold 1978), and prior applications of the method have often used a value of at least 0.3 at similar sites. However, Cafferata et al. (2004) and Cafferata and Reid (2013) demonstrated that the 0.3 value produces overestimates of peak flows in the Caspar Creek watershed for both for 10-yr and 100-yr calculations. Therefore, we adopted the recommendations of ODOT (2014) for woodlands and forests and used a value of 0.2 for a 10-yr storm and 0.25 for a 100-yr storm; we then compare these results with those obtained using a value of 0.3 for both 10-yr and 100-yr flows. T_c was calculated for each of the five sub-watersheds using the California culvert practice, Airport Drainage, and BCMOE equations. Required information, including elevation difference, average channel gradient, and flow distance, was determined using a digital topographic map. Rainfall depth-duration-frequency data were obtained from the NOAA website, “Atlas 14 Point Precipitation Frequency Estimates for California.”

The USGS Magnitude and Frequency Method requires an estimate of the mean annual precipitation, for which we used the value of 1190 mm (47 in) as indicated by the 1961-1997 record (Henry 1998). Application of flow transference methods makes use of flow frequency information from a long-term stream gage. We used the 53-year record from the North Fork Caspar Creek gage, located downstream of the five test sub-watersheds (fig. 1). For the NRCS WinTR-55 method, weighted curve numbers for sub-watersheds were determined by estimating the percent of the basin drainage area in each hydrologic group (e.g., B, C) using NRCS soil series data. The NOAA atlas website was used to obtain rainfall depth-duration frequency data for a 24-hour duration for a variety of return periods.

We used 10-year flows for the primary analysis because the largest flows in some of these sub-watersheds had return intervals of less than 25 years, and the length of the measured record is not sufficient to allow accurate estimates of the 100-year flows. However, we expect that the relative reliability of methods for predicting 10-year peak flows is likely to also characterize their reliability for estimating 100-year events (Cafferata et al. 2004), and we tested this assumption by carrying out calculations also for the less-well-defined 100-year flows.

Results

The flood frequency distribution calculated from HEN gaging records using the USGS PeakFQ program shows only minor variation about the best fit model and is typical of those constructed for the other four sub-watersheds (fig. 2). We then compared the values predicted by each of the flow estimation methods to the 10-year flow calculated from gaging station records for each of the five test sub-watersheds (table 1).

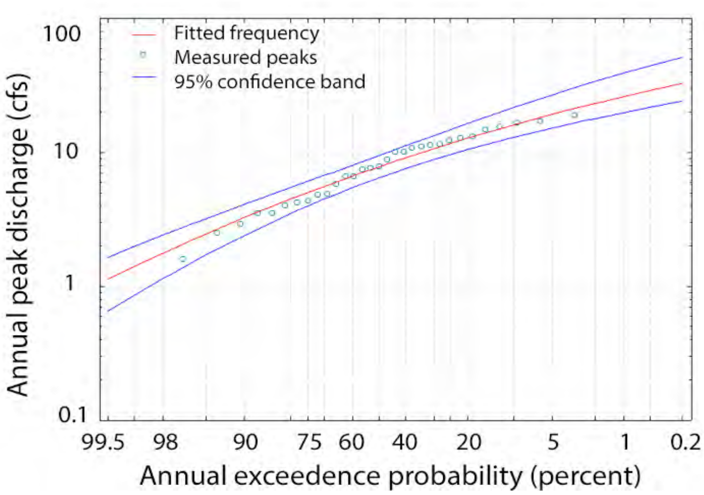

Figure 2—Flood frequency curve for sub-watershed HEN.

Each of the four flow estimation methods tested over-predict the 10-year event for the five test sub-watersheds (table 1, fig. 3A). Departures from the flow frequency analysis results were highest (mean: 220 percent) for the Rational Method using the California culvert practice equation to determine the T_c, while the direct flow transference method provided the lowest overestimate (mean: 17 percent).

The 31-year duration of flow records is too short for a highly reliable estimate of 100-year flows, but calculation of those flows remains useful in order to determine whether the patterns of accuracy established for the 10-year flows are likely to hold also for the larger flows. Results indicate that the patterns of deviation are indeed similar (fig. 3B). In this case, too, using the California culvert practice equation with the Rational Method—this time with C = 0.25, as recommended by ODOT (2014) for 100-yr flows—provided the least reliable estimates, while the direct flow transference method provided the lowest overestimate (9 percent).

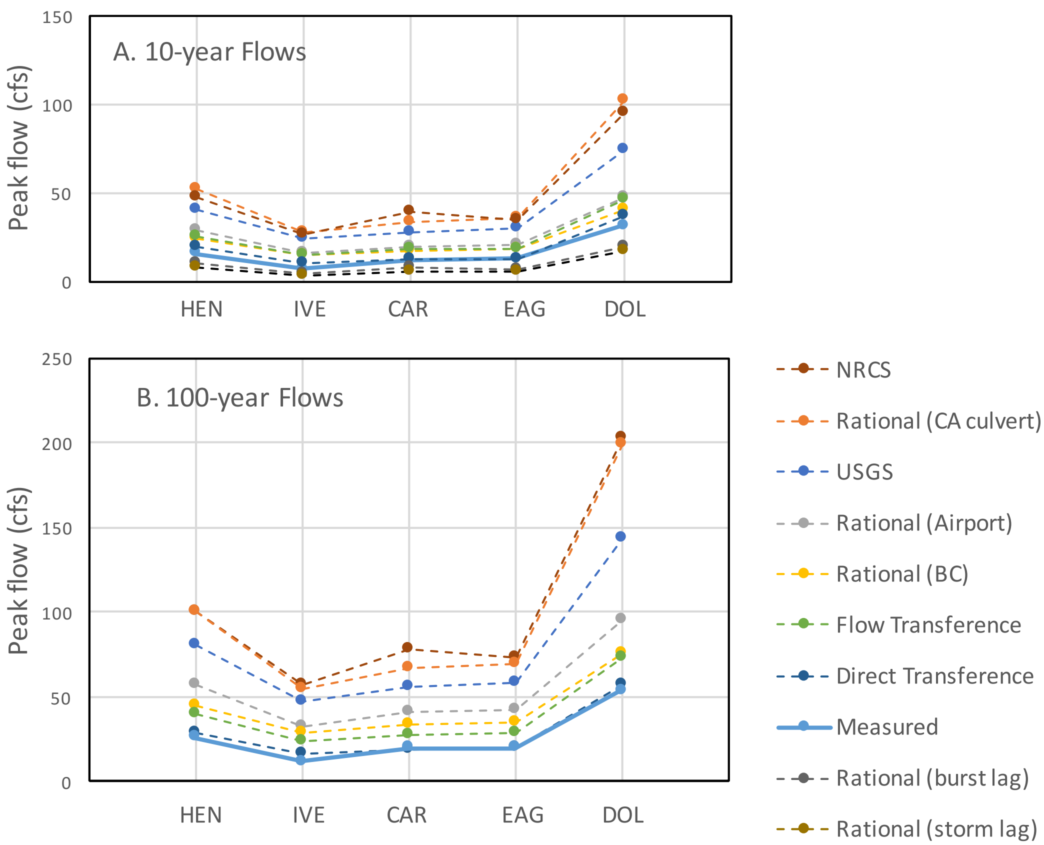

Figure 3—Predictions of the (A) 10-year and (B) 100-year discharge for the five gaging stations.

Discussion

Comparison with Results From Elsewhere

Results of this study are generally similar to those reported for other west coast studies that compared peak flow prediction methods or T_c equation results to measured storm peaks. Fedora (1987) found that the SCS curve number methodology over-predicted peak discharge by a factor of two in the Alsea watershed located in the Oregon Coast Range. Gregori (2003), using data from the H.J. Andrews Experimental Forest in the Oregon Cascades, and Loukas and Quick (1996), working with data from the Carnation Creek watershed in British Columbia, found that standard T_c equations considerably underestimated watershed response time, which would result in overestimation of peak flows. Cafferata et al. (n.d.) reported that the Rational Method using the California culvert practice equation for T_c overestimated the 100-year flow by 130 percent for a headwater tributary in the Teakettle Experimental Forest in the Sierra Nevada. The Rational Method using the Airport Drainage and BCMOE equations for T_c, the USGS Magnitude and Frequency Method, and the flow transference method produced estimates that were within 20 percent of the estimated 100-year discharge at that site.

Comparing the Models

In the present case, the Rational Method, using the California culvert practice equation for T_c and a *C*-value of 0.2 for 10-yr flows, produced results that were the most divergent from the values derived from the flow frequency analysis. This outcome in part reflects the difficulty of defining appropriate values for *C* and T_c in a region where flow generation processes are not those for which the method was developed. T_c, in particular, has a clear physical meaning for this application only in watersheds dominated by overland flow. In contrast, subsurface flow dominates at Caspar Creek, with soil matrix flow draining into a network of soil macropores. The first channelized flow is thus through soil pipes of unknown extent. Consequently, neither the typical flow path nor the portion of the watershed that directly contributes flow to a particular storm's runoff can be reliably defined. It is thus useful to explore alternative approaches to defining Tc and C to evaluate whether modifications to the approach might be effective in such settings.

To determine whether a more empirically-based index of hydrologic response time might improve the performance of the Rational Method under these conditions, we calculated the storm centroid lag to peak (fig. 4) from hydrographs and hyetographs for a 10-yr event in each sub-watershed and used these values in place of calculated T_c values to identify the relevant rainfall intensity, again using C = 0.2. The resulting estimated flows consistently underestimated the 10-yr peak flows (fig. 3A, table 1).

A second set of calculations, this time using the centroid lag to peak from just the within-storm rain period that generated the peak (the “burst lag” in fig. 4), also consistently underestimated observed values (fig. 3A, table 1). These modifications would lead to valid estimates only if the value for *C* is about twice that expected. In the case of sub-watershed HEN, the response times for the storm centroid and burst centroid lags to peak were 366 and 269 minutes, respectively.

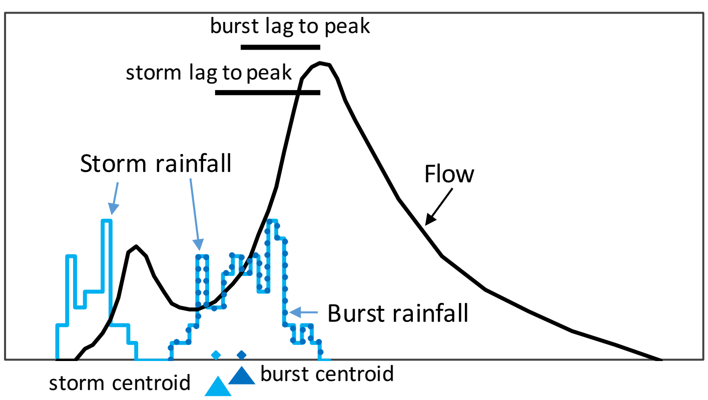

Figure 4—Hypothetical hydrograph showing definition of terms for lag calculation.

Definition of an appropriate *C*-value is also problematic. Forestland *C*-values recommended by various sources span the range of 0.1 to 0.6, and because *C* is a simple coefficient, the resulting estimated peak flows would differ by up to a factor of 6. In the case of coastal forestlands, past recommendations have suggested using values no lower than 0.3. For the present application, use of C = 0.3 appreciably increased the overestimates. Cafferata and Reid (2013) used the Caspar Creek data to attempt to identify an appropriate *C*-value for this area. For that application, T_c was calculated using the Airport Drainage equation (T_c = 34 min for sub-watershed HEN), and an appropriate *C* value was back-calculated from the observed 10-yr peak. In effect, inaccuracies in both *C* and T_c were collapsed into a single variable under the assumption that these inaccuracies would be relatively uniform for conditions across the area of interest. The resulting value (C = 0.13) was tested by comparing predicted and observed 10-yr flows in a variety of nearby watersheds. Results showed reasonable agreement for watersheds smaller than 80 ha (200 ac). This result suggests that the Rational Method might become a useful approach in an area if sufficient data are available for calibration, but the need for local data to a large extent counters the attraction of the original method.

The USGS and flow transference methods are similar to one another in that both are based on calibrations—the first at a regional scale and the second more locally. The USGS method was less accurate than either of the flow transference methods tested, but it has the advantage of not requiring local data for calibration, and it can be applied to larger watersheds. At Caspar Creek, the flow transference methods provided the most reliable estimates, but application at this site is not typical because the data used to calibrate the model were from a stream gage in the same watershed as the sub-watersheds studied. Cafferata and Reid (2013) tested the transference methods for a more typical case, using data from the Noyo River USGS gage 6 km (4 mi) from the watershed, and found that the methods performed similarly to the USGS Magnitude and Frequency Method and the Rational Method using the Airport Drainage equation to calculate T_c. Finally, the NRCS WinTR-55 approach did not perform well at Caspar Creek; as is the case with the Rational Method, it functions better in watersheds where overland flow is an important source of runoff.

Differences Between Sub-watersheds

Examination of the differences in model performance between individual sub-watersheds showed that the models performed consistently less well for the IVE watershed (fig. 5), suggesting that the hydrologic response at IVE differs from those at the other sites tested. This difference is also supported by field observations: IVE hydrographs show larger lags to peak and more protracted peaks than other sites, and the catchment also supports perennial flow, a rarity for watersheds of this size (21 ha, 52 ac) in this area.

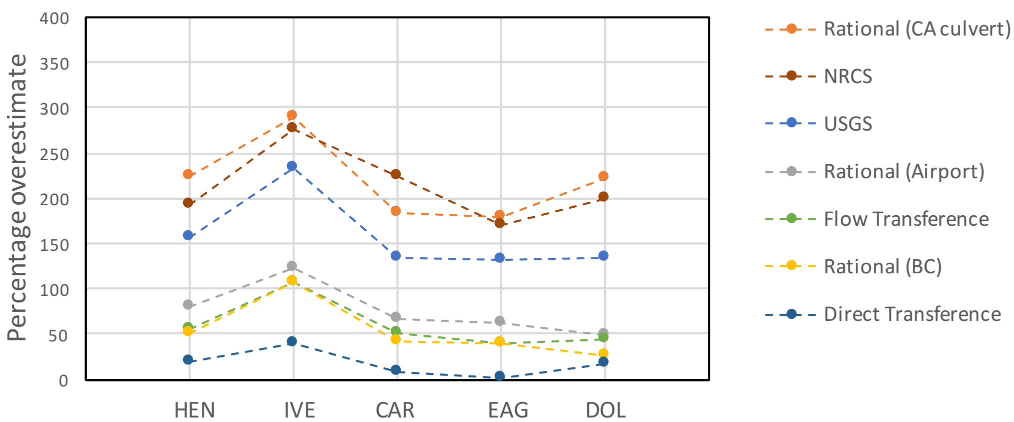

Figure 5—Percent over-estimation for 10-yr flows in the test sub-watersheds.

Little information exists on whether peak flow changes associated with forest management are large enough to affect crossing design. Previous studies demonstrated increased peak flows after clearcutting at Caspar Creek (e.g., Lewis et al. 2001, Ziemer 1998). The current study employed data for sub-watersheds CAR, EAG, and DOL that spanned the period before logging, immediately following logging, and during the hydrologic recovery of the watersheds. Periods of uniform conditions could not be isolated for analysis due to the need for a lengthy record in order to adequately define 10-yr return-interval flows, and because a trend toward recovery begins soon after logging is completed. Furthermore, gaging records used to calibrate the USGS and flow transference methods also reflect partially logged watersheds. It is possible that the generally lower overestimates obtained by these methods for the clearcut and partially clearcut sub-watersheds (fig. 5) may simply reflect an increase in peak flow in these basins relative to the control sub-watersheds HEN and IVE, though data are insufficient to test this possibility. In any case, crossing designs should take into account potential changes in peak flows that might take place due to land-use activities upstream of the crossing.

Implications for Crossing Design

Each of the flow prediction methods tested has different data requirements. The flow transference methods, while producing the best results at Caspar Creek, are often limited by the availability of gaging data from nearby hydrologically similar watersheds. In contrast, the USGS, NRCS, and Rational Methods all require data that are now readily available from digital topographic maps and internet sites. Choice of the most effective model to use for a particular application thus depends in part on the kinds of information available. For any application, it is advantageous to apply several of the methods in order to evaluate the likely uncertainty associated with any one method's results. Office-generated results always need to be evaluated in the light of field observations of factors such as bankfull channel capacity, active channel width, and crossing performance at nearby sites after large flood events.

In the coast redwood region, large flows alone generally are not the primary cause of watercourse crossing failures (Flanagan 2004). For storms with return intervals of < 12 yr, some combination of woody debris and sediment deposition accounted for 86 percent of the crossing failures inventoried across a range of site conditions in northwestern California, while hydraulic exceedance and debris torrents produced 12 and 2 percent, respectively (S. Flanagan, BLM, Arcata, unpublished data; n = 57). Similarly, Furniss et al. (1998) found that only 9 percent of failed crossings in the Pacific Northwest and northern California resulted from hydraulic exceedance. The California Forest Practice Rules thus require that crossings be designed to allow adequate passage not just of water, but also of wood and sediment.

Unfortunately, analytical methods analogous to those for peak flow prediction are not available to aid in sizing culverts for wood and sediment passage. Furniss et al. (1998) outline several approaches that can help to reduce failure risk: (1) ensuring that the pipe diameter (D) is large enough that headwater depth (HW) remains well below the top of the pipe ($HW/D \leq 0.67$ preferred), (2) installing culverts of similar width as the active channel, (3) installing culverts at the same gradient as the natural channel, (4) aligning culverts so that they are parallel to the natural channel, and (5) eliminating wide areas near pipe inlets. Additionally, the risk of culvert failure can be reduced by installing a single large pipe rather than multiple pipe barrels, placing flared metal end sections at culvert inlets, using mitered pipe inlets, and installing trash racks where winter maintenance is possible. Flanagan (2004) noted that if a culvert is sized for wood passage (i.e., the pipe width is approximately equal to the active channel width), hydraulic capacity is generally adequate for the 100-year flow.

In many situations, the best approach for reducing the risk of crossing failure is to not install a culvert. Use of rock fords, rock-armored crossings, bridges, and open-bottom arch installations has become much more common in the past 15 years in the redwood region. Site-specific conditions that may lead to preference for these types of crossings include winter maintenance issues, landslide-prone terrain, the presence of large amounts of mobile wood, and fish passage requirements. These types of crossings must also be sized for 100-year flows but are less sensitive to both flow prediction errors and wood- or sediment-induced failure than are culverts. The most failure-resistant design, however, is not to use permanent structures, but to instead install temporary crossings that are removed prior to winter.

Conclusions

Numerous approaches are available to RPFs in California to estimate 100-year flood flows for crossing design. The four commonly used methods we tested at Caspar Creek produced widely varying results. The Rational Method, often used for small watersheds, was shown to be capable of producing reasonable flow estimates if appropriate C and T_c factors are used. We do not recommend using the California culvert practice equation to calculate T_c; both the Airport Drainage and BCMOE methods produced more realistic values. The flow transference methods that used data from a nearby stream gage provided the most accurate estimates. The NRCS WinTR-55 method did not produce accurate estimates for the 10-year peak flows. The USGS Magnitude and Frequency Method equations produced results better than those from the NRCS method, but considerably poorer than those of the flow transference methods. Future flow data from Caspar Creek will allow more rigorous testing of estimates for larger flows (> 10 yr RI).

Acknowledgments

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Shrinking Streamflows in the Redwood Region[1]

Randy. D. Klein,[2] Tasha McKee,[3] and Katrina Nystrom[3]

Abstract

The ongoing, severe drought in the redwood ecosystem has many ramifications, including loss of summer rearing habitat for juvenile salmonids. Many 'perennial' streams now cease to flow during parts of the summer and fall, either drying up completely or disconnecting pools as riffles go dry, subjecting fish to increased predation, high water temperatures, and desiccation. Numerous factors have contributed to this hydrologic impairment, including natural hydrologic cycles, legacy land use effects, human consumptive uses, and climate change. The search for solutions is now in full swing, and will be most effective if based on site-specific monitoring to identify controllable causes and a non-confrontational approach to changing water usage in time and space.

Monitoring of low flows in the region has been ramping up in recent years, led by the program pioneered by Sanctuary Forest, a non-profit group located in the Upper Mattole River. Their data collection program has run from 2004 through the present and consists of measuring low summer/fall stream discharge at a network of key locations in the watershed. The primary goals of hydrologic monitoring are to identify locations of extreme flow impairment and to guide efforts for water conservation.

A crucial element of Sanctuary Forest's program has been to heighten the awareness of landowners of the low flow problem and involve those willing in a forbearance program that offers increased water storage capacity in exchange for cessation of water withdrawals during drought conditions. The date of water withdrawal cessation varies each year, but this date is important for landowners involved with the program: it determines when their water sources switch from the creeks to their storage tanks. Correlations with online-accessible, real time streamflow and precipitation data have provided convenient means to determine, and even forecast, the date when pumps must be turned off.

Recent data show that Sanctuary Forest's forbearance program is having a positive effect on low flows. Measurable increases in low flows in the Upper Mattole River have been observed since 2009, elevating extreme low flows and reducing the number of days when flows fall below minimums needed for juvenile salmonid migration. With growing participation in the program, benefits to summer low flows will continue to accrue and improve conditions for fish in the Upper Mattole River.

Because of the success of Sanctuary Forest's monitoring and forbearance program, it is now being replicated in nearby watersheds. If the recent trend of worsening droughts proves to be the new 'normal', maintaining and augmenting forbearance in the Upper Mattole River, and indeed the entire redwood region, will become increasingly important for juvenile salmonids.

Keywords: drought, forbearance, low flow, streamflow, water withdrawal

Introduction

Lack of adequate late summer and early fall streamflow was recognized by the State of California as one of the most important limitations on salmonid habitat in the Mattole River basin (NCWAP 2001). In recent years, juvenile salmonids have become stranded in pools due to excessively low flows, causing mortality and necessitating fish rescue operations. With the exception of 2005, 2010, and 2011, late summer and early fall discharges were quite low for most of the past 15 years, with the summer of 2008 being the driest and 2014 the second driest in the 67-year record of flows on the

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Mattole River near Petrolia. A variety of factors influence low flows, such as, climate (rainfall, air temperature, fog, relative humidity, wind speed), vegetation species and age distribution, ground disturbance, streambed sediment depth, water use for domestic and agricultural purposes. Of these, only vegetation, ground disturbance, human water use, and possibly riparian aquifer storage are subject to human influences and therefore might be modified to improve low flows.

Sanctuary Forest has undertaken a program to reduce dry season pumping from the Upper Mattole River and tributaries by subsidizing purchases of large storage tanks for willing landowners and facilitating forbearance agreements that strategically reduce water extraction from streams, thus improving low flows and relieving habitat stress for salmonids. This paper presents an analysis of low flows in the Upper Mattole River basin with the following objectives: 1) to put recent droughts into a longer-term perspective, and 2) to evaluate the effectiveness of the water storage and forbearance program in the Upper Mattole.

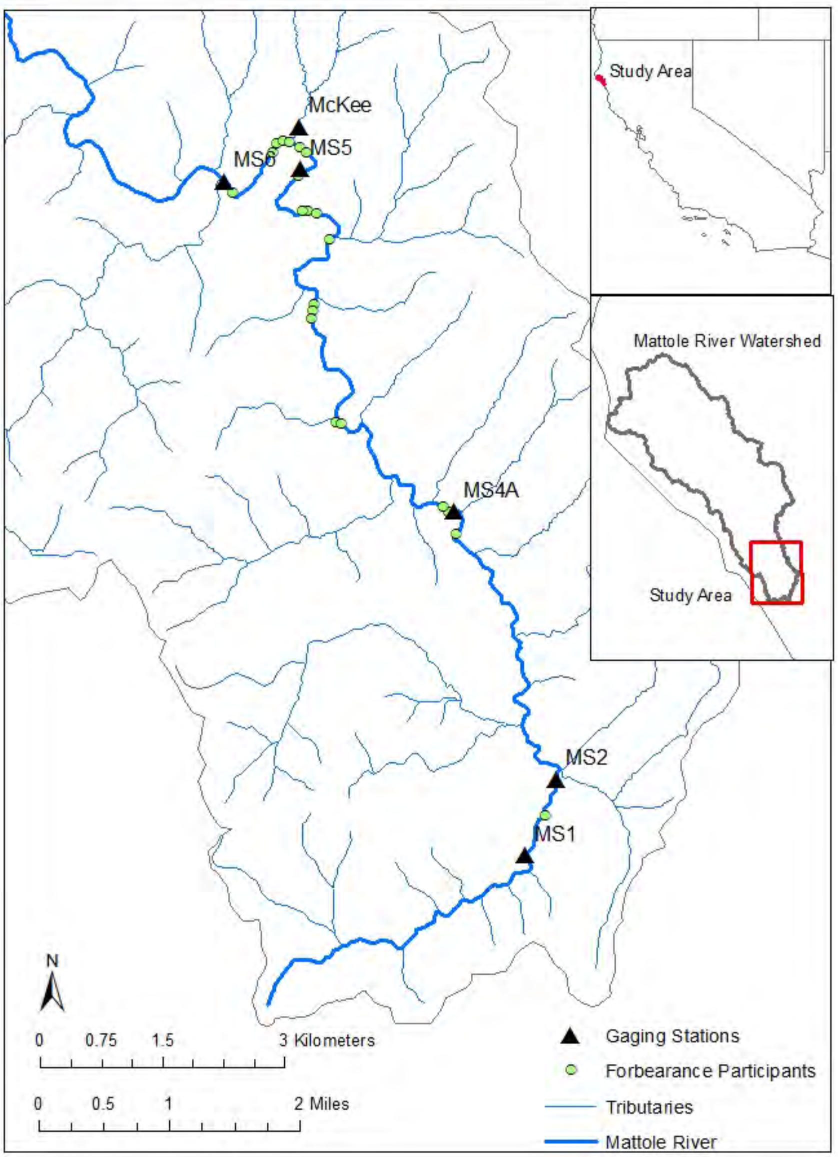

Figure 1—Upper Mattole watershed with monitoring sites and forbearance participants.

Study Area

The Upper Mattole River basin can be defined as that portion of the watershed upstream of the U.S. Geological Survey stream gaging station at Ettersburg (Sta. No. 11468900, drainage area = 151 km^2 (58 mi^2), fig. 1). The region has a Mediterranean climate, with virtually all precipitation falling as rainfall and averaging 191 cm/year (75 inches/year) occurring from October through the following June. Ranching and logging are the primary historical land uses, but marijuana cultivation has grown rapidly in recent years, and this so-called 'green rush' has become a focus for concerns over shrinking streamflows in salmonid-bearing streams (Bauer et al. 2015).

Since 2006, there has been a significant increase in awareness of the low flow problem and a community-wide response to increase water conservation as well as storage and forbearance agreements that reduce water extraction during the low flow periods. Starting in 2006, Sanctuary Forest implemented a streamflow education and outreach program that includes public service announcements and website alerts about streamflow conditions, water conservation and water storage educational materials, and community meetings. Additionally, Sanctuary Forest developed a water storage and forbearance program, with funding and implementation beginning in 2006. In 2007, storage systems were installed for the first two landowners along with legally recorded forbearance agreements to end all pumping at specified flow thresholds. By 2015 a total of 25 storage tanks had been installed totaling 5,700 m^3 (1.5 million gallons) in storage volume. All 25 participants have forbearance agreements, and an additional estimated 20 landowners have purchased tanks on their own and are participating voluntarily.

During summer drought conditions, landowners cease pumping when flows dropped to the cutoff of 20 l/s (0.7 cfs) at the downstream end of the Upper Mattole forbearance program area (MS6 monitoring site, fig. 1). Forbearance participants are notified of the cutoff date directly through phone and written communications from Sanctuary Forest and a flow alert roadside sign is also maintained to give a 2 week notice to the entire community. The low flow benefits of "turning off" the 25 pumps are analyzed in this report relative to flows measured at the MS6 monitoring site to show the cumulative benefits of the program.

Existing Data and Previous Studies

The "Northcoast Watershed Assessment Program" (NCWAP) Mattole River report (NCWAP 2001) provides a compilation of climatic and hydrologic data sources for the Mattole River. Appendix C of the NCWAP report, prepared by the California Department of Water Resources (DWR), lists all known official (government sponsored) data collection efforts in the Mattole and has assembled relevant data and performed some basic analyses, primarily of rainfall and streamflow. Sanctuary Forest staff has been collecting streamflow data since summer, 2004, and their data form the basis for most analyses contained herein. In addition, streamflow data collected by the U.S. Geological Survey (USGS) at stream gaging stations at Petrolia (1912 to 1913 and 1951 to present) and near Ettersburg (2001 to present) were used for assessing drought severity. Because water use was not a quantitative component of the present analysis, the reader is referred to the NCWAP (2001) study, which provides a listing of appropriative water rights granted within the Mattole River basin along with estimates of water use. Klein (2004) also summarized water use based on locally-derived estimates provided by Sanctuary Forest staff, but the accuracy of either of these estimates is unknown.

Stubblefield et al. (2012) measured water use by trees in the Mattole River, finding that although older and larger trees use more water, dense, younger tree stands use more per unit area. They project that, as forests are allowed to mature, the declining numbers of young trees will result in less total water use by forested areas. This, of course, assumes forests will be maturing despite ongoing timber harvest and future stand-replacing wildfires, should they occur.

Sawaske and Freyberg (2014) analyzed stream gaging records from Pacific coastal streams, finding that although spring discharge recession rates have remained relatively constant for the past 4 to 8 decades, summer recession rates have increased. Their results agree with those of Asarian and Walker (2016), who found that although precipitation-adjusted streamflow at pristine sites had not declined, September streamflow declined at 73 percent of un-dammed sites in northwest California and southwest Oregon in the latter part of the available record. They attributed this to water withdrawals and vegetation changes rather than precipitation or other climatic changes.

The burgeoning marijuana industry in north coastal California has been well-publicized. Bauer et al. (2015) conducted aerial inventories of 'grows' (signified by large greenhouses and outdoor gardens) relying primarily on Google Earth's high-resolution images with some level of verification derived from law enforcement activities on the ground. They speculated that for the redwood region, from 23 percent to 100 percent of summer flows may be withdrawn at the time of their study (2012) for use by this industry, but that deriving more accurate numbers was hampered by grows being typically clandestine operations located on private property. Whatever the true rates of water withdrawals for marijuana growing operations (and they have likely risen since 2012), their proliferation adds to the cumulative effects of the other human-caused decreases in streamflows mentioned above.

Methods

The data from the USGS gaging stations at Petrolia and Ettersburg were analyzed by first computing the minimum 7-day low flow discharge. This was accomplished by scanning the daily average flow data for the dry months (July to October) each year for the 7-day period with the lowest flow of the season. The exceedence probability (the likelihood of a specific flow being equaled or exceeded in any given year) was then computed for each annual 7-day low flow, allowing an assessment of whether or not low flows have declined in recent years.

Beginning in August, 2004, flows were measured by Sanctuary Forest staff and volunteers at selected sites in the Upper Mattole River basin on both the main stem and selected tributaries. Main stem sites are numbered in a downstream direction (MS1 is at the upper end of the monitoring reach, MS6 is at the lower end; see fig. 1). Site descriptions are listed in Table 1, which also includes the USGS gaging stations used here (note that tributary data are not included in this paper due to length limitations, but can be found in Klein (2012).

Table 1—Mattole River mainstem monitoring sites and characteristics

Site name	Dist. upstream from mouth (km)	Drainage area (km^2)	Description
MS1	68.8	8.4	downstream of Big Alder Creek
MS2	68.3	10.3	upstream of Lost River confluence
MS4A	66.3	31.9	downstream of Gibson Cr. confluence
MS5	61.7	59.9	upstream of McKee Creek
MS6	60.6	66.4	upstream of Bridge Creek
USGS Ettersburg	48.7	150.5	near Ettersburg
USGS Petrolia	5.8	634.6	at Petrolia

The data collected by Sanctuary Forest enabled a more detailed assessment of Upper Mattole low flows than was possible solely using USGS gage data. Measurements were made by collecting the flow at a confined section of the channel in a 19 liter (5 gallon) bucket and timing how long it took to fill the bucket (volumetric method), or with an electromagnetic current meter (Marsh-McBirney), depending on prevailing flow and site conditions. Data loggers were used to provide a continuous record at three mainstem sites (MS4A, MS5 and MS6) during some years.

Results

Low Flow Exceedence Probability

Table 2 shows the 7-day minimum low flows and exceedence probabilities for both Petrolia and Ettersburg gages for the entire period (2001 to 2015). In addition to the exceedence probabilities for the 7-day low flows plotted in fig. 2, table 2 includes the inclusive dates of the 7-day low flow period each year. In every case but one (2015), the minimum flows occurred in September and/or October.

Table 2—Low flow discharges (cms), dates and exceedence probabilities for Petrolia (PET) and Ettersburg (ETT) stream gages, 2001-2015

Water year	Petrolia (cms)	Petrolia (cms/km^2)	Petrolia dates	Exc. Prob. (%)	Ettersburg (cms)	Ettersburg (cms/km^2)	Ettersburg dates
2001	0.49	0.0008	10/5-11/2001	81	0.14	0.0009	9/3-9/2001
2002	0.40	0.0006	9/26-10/2/2002	90	0.11	0.0007	10/7-13/2002
2003	0.74	0.0012	10/25-31/2003	72	0.15	0.0010	10/23-29/2003
2004	0.51	0.0008	9/28-10/4/2004	88	0.12	0.0008	9/7-13/2004
2005	1.16	0.0018	10/7-13/2005	4	0.28	0.0019	10/8-14/2005
2006	0.64	0.0010	9/22-28/2006	70	0.15	0.0010	9/24-30/2006
2007	0.56	0.0009	9/24-30/2007	52	0.17	0.0011	8/31-9/6/2007
2008	0.31	0.0005	9/5-11/2008	97	0.09	0.0006	9/4-10/1/2008
2009	0.56	0.0009	9/27-10/3/2009	57	0.17	0.0011	9/25-10/12009
2010	0.96	0.0015	10/15-21/2010	10	0.28	0.0018	10/15-21/2010
2011	0.65	0.0010	9/8-24/2011	23	0.24	0.0016	9/18-24/2011
2012	0.55	0.0009	9/30-10/6/2012	64	0.16	0.0010	9/28-10/4/2012
2013	0.45	0.0007	9/10-16/2013	46	0.18	0.0012	9/9-15/2013
2014	0.40	0.0006	9/6-12/2014	93	0.11	0.0007	9/11-17/2014
2015	0.43	0.0007	8/22-28/2015	84	0.13	0.0008	10/4-10/2015

The NCWAP (2001) report presented and analyzed rainfall in the Mattole based on two long-term rain gages and streamflow records in the lower Mattole River near Petrolia (USGS Gaging Station No. 11469000, drainage area = 635 km^2 (245 mi^2)). Based on their analyses, they concluded there were no discernible long-term trends in annual precipitation and that there was only a modest decline in annual yields over the previous 25 years, which was attributed to increasing water withdrawals from streams.

Sixteen additional years of data have been collected at the two USGS gages currently in operation (Petrolia and Ettersburg) since the NCWAP (2001) analyses were done, some among the driest on record. An exceedence probability analysis of 7-day low-flows was done using the full period of record (fig. 2). Because the Ettersburg gaging station lacks sufficient record length to perform low flow frequency analyses, discharge was estimated for the pre-record (1912 to 2000) period by down-scaling the Petrolia data by drainage area ratio.

As shown in fig. 2, the 2008 7-day low flow was the lowest on record for Petrolia, and many other years since 2001 are clustered near the high end of exceedence probabilities (lowest flows) portion of the data shown. At 86 percent, the median exceedence probability of this latter period is much higher than that of the entire record, indicating a distinct shift to lower summer flows. The wettest years of the recent period were 2005 and 2010 to 2011. Although not as dry as 2008, 2014 was the second-driest on record at both Petrolia and Ettersburg.

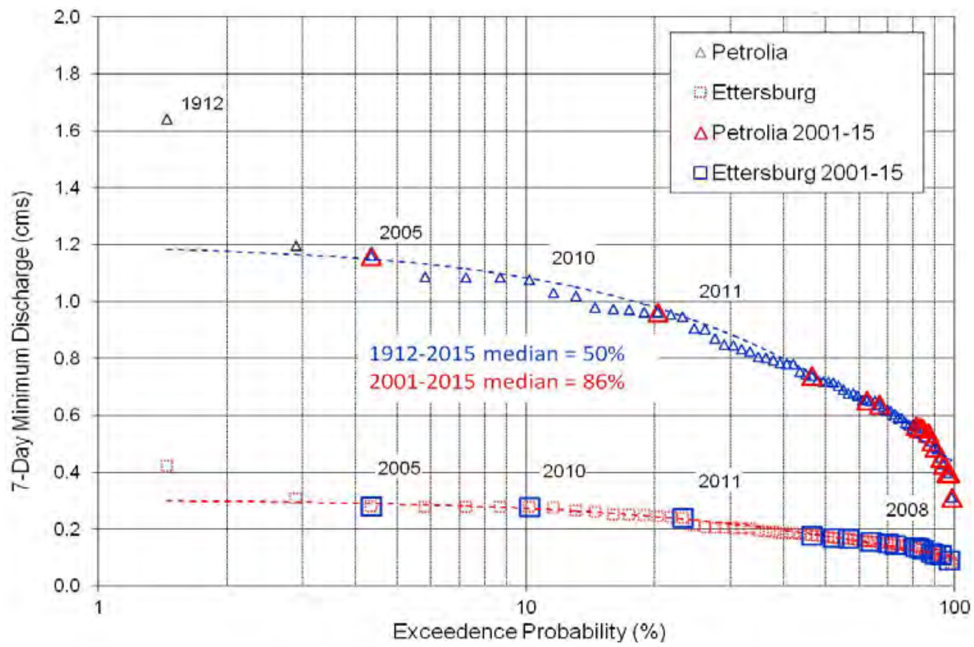

Figure 2—Exceedence probabilities for the annual 7-day low flow discharges for Mattole River at Petrolia and Ettersburg with recent data (WY2001-16) highlighted (note that data for Ettersburg prior to 2001 were estimated by down-scaling by drainage area from Petrolia records).

Forbearance Program Monitoring and Implementation

Figure 3 presents discharge measurements taken at the mainstem site MS6 for 2004-15. This is an important monitoring site because it is used in the forbearance program as the index site for determining when water withdrawals should cease when the established discharge threshold of 20 l/s (0.7 cfs) is attained. The beginning of the summer flow recession varied widely among the years shown, with the 2008 recession (driest year) starting in early June and not until early August in 2005 (the wettest year). Flows dropped to zero in 2004 and 2008. Flow recovery in all years did not occur until rains began in mid-September at the earliest and November at the latest.

The effects of late spring rainfall can be readily seen in the delayed recessions in wetter years (e.g., 2005, 2010, and 2011). The year 2008 was by far the driest, with the MS6 pumping cessation threshold (cutoff) of 20 l/s (0.7 cfs) reached in late July. Although 2014 was the second driest year at Ettersburg, fig. 3 shows that flows rebounded after pumping was ceased on July 20, 2014, and MS6 minimum flows were higher than in several other dry years.

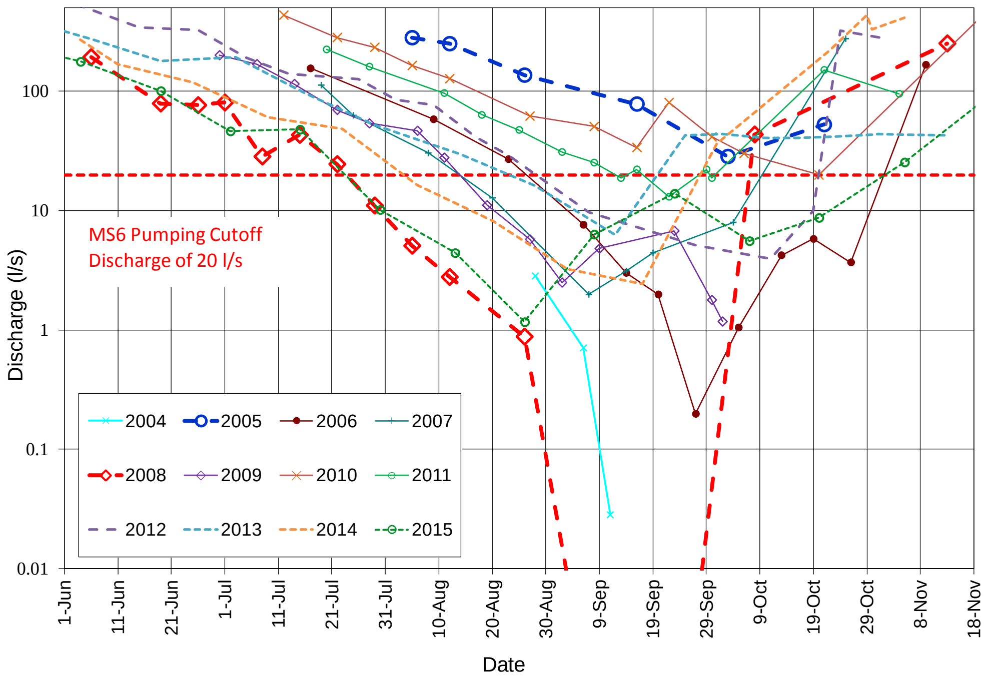

Figure 3—Discharge measurements at MS6, 2004-15 (all dates adjusted to 2016 for plotting). The discharge threshold for cessation of pumping (forbearance) for MS6 (20 l/s, or 0.7 cfs) is also shown. The two heaviest lines indicate flows during 2008 (driest year, dashed red line) and 2005 (wettest year, dashed blue line).

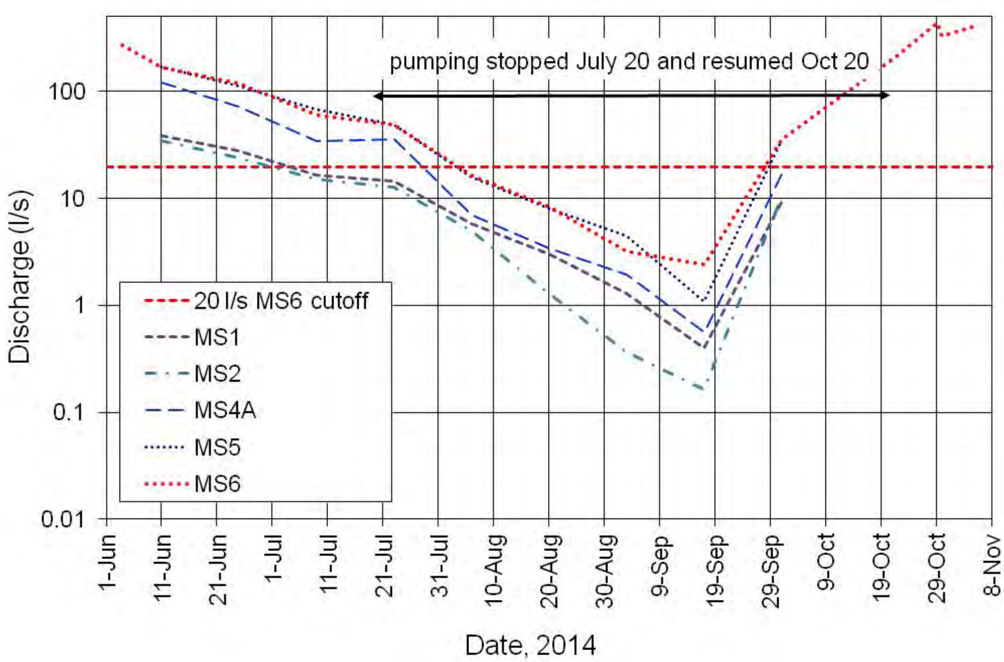

Figure 4—Discharge measurements along the Upper Mattole mainstem in 2014. Pumping cessation period (July 20 through October 20) is noted. Sites are numbered from upstream to downstream order.

Figure 4 shows discharge measurements for five mainstem gaging sites in 2014, with the forbearance pumping cessation period (July 20 through October 20) noted. Predictably, discharge generally increases with increasing drainage area (sites are numbered from upstream to downstream order). Landowners participating in the forbearance program were notified to cease pumping beginning July 20, ahead of the impending flow recession to the MS6 threshold (20 l/s). Pumping restrictions for participants were lifted in October after when rains finally alleviated the seasonal drought. Without forbearance, one or more of the three mainstem sites would almost certainly have gone dry for a lengthy period of the dry season.

Forbearance Program Effectiveness

Table 4 lists the annual progress of the forbearance program upstream of MS6 through 2015. Maximum instantaneous annual and cumulative pumping rates for participants are listed, and can be taken as the maximum reductions in water withdrawals when forbearance is invoked. As of 2015, the maximum potential pumping reduction achieved thus far is 13.6 /s (table 4), or about 0.5 cubic feet per second.

Table 4—Participation in the forbearance program from 2007 to 2015 for the Upper Mattole upstream of MS6 with estimated instantaneous pumping capacities of participating landowners

Year	No. of landowners turning off pumps	Annual storage installed (m^3)	Max instantaneous pump capacity (l/s)	Cum. instantaneous pump capacity (l/s)
2007-09	7	1,701	4.91	4.91
2010	4	567	1.26	6.17
2011	1	567	1.26	7.43
2012	4	699	0.63	8.06
2013	2	491	2.52	10.58
2014	2	680	1.26	11.84
2015	5	1,002	1.76	13.61

To estimate the hydrologic benefits from the forbearance program, fig. 6 plots MS6 discharge measurements for 2015 and the lower flows expected in the absence of the program (no restrictions on pumping) based on the cumulative instantaneous pump capacity reductions from table 4 (measured flows were reduced by the maximum instantaneous pumping capacity of 13.6 l/s (0.5 cfs) to estimate what flows would have been without the forbearance program). This hypothetical situation assumes all forbearance participants, had they not entered the program, would be pumping from Upper Mattole streams at full pump capacity at some point in time. Use of the maximum instantaneous pumping capacity might overestimate the benefits of the forbearance program, but there is some merit to the assumption that all pumps may be withdrawing water simultaneously during the driest part of the summer when water users lacking adequate storage would most likely be pumping from streams.

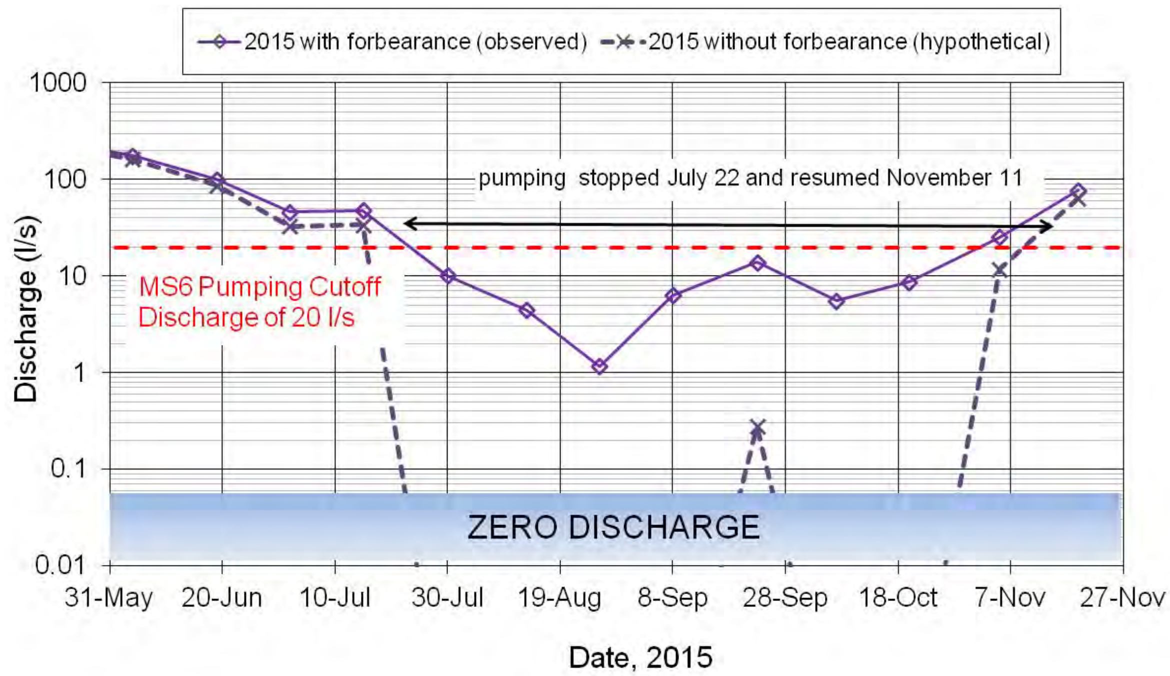

Figure 6—Discharge measurements at MS6 in 2015 (observed) and estimated (hypothetical) discharge without forbearance.

Predicting Discharge at MS6

Ettersburg provisional flow data are available online through the USGS website (http://waterdata.usgs.gov/ca/nwis). Estimating MS6 flows using Ettersburg real-time data offers a considerable convenience for anticipating and invoking forbearance events. Figure 9 shows the relationship between Ettersburg and MS6 flow for WY2004 to 2014. The relationship is strong enough to use the Ettersburg real-time flow data to provide reliable estimates of MS6 flows within the range of flows analyzed. Sanctuary Forest has been using the relationship with real-time flow data from Ettersburg, along with direct observations of flows, as an important tool in managing the forbearance program. Daily checking of Ettersburg flow and applying the regression equation in fig. 5 to that flow allows estimation of MS6 flows from the office. As the forbearance threshold is approached, participants can be forewarned of an upcoming pumping cessation period.

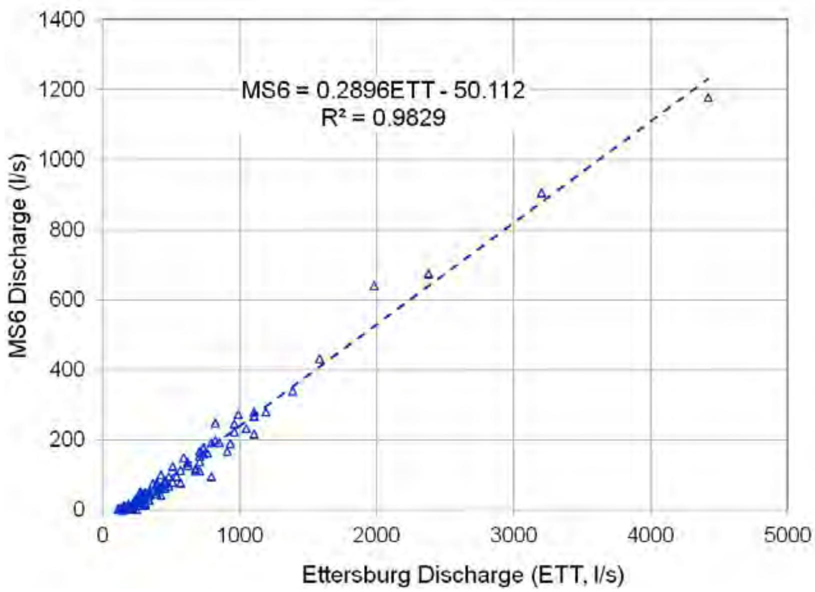

Figure 5—Relationship between discharge at the USGS Ettersburg gaging station and those taken at MS6, 2004-2015.

As an aside, the regression was also performed just using the 2012 to 2014 data to evaluate whether or not the relationship had shifted in a manner consistent with the increase of upstream storage and reduced pumping (i.e., increasing in MS6 flows relative to Ettersburg). No differences were found between the full dataset and the more recent data. At nearly twice the contributing drainage area of MS6, the low flow improvements due to the forbearance program would have to be higher than that computed below (13.6 l/s, see table 4) to be detected by a shift in the MS6-Ettersburg relationship.

Conclusions

1. With the exception of 2005, 2010 and 2011, drought conditions have been unusually severe since 2001 in the Upper Mattole River, with 2008 being the driest of the 67-year discharge record for the Mattole River near Petrolia. Median exceedence probability during this later period was 86 percent, compared to 50 percent for the entire record, indicating a distinct shift to lower summer flows.
2. Real-time Ettersburg flow data, available online, was demonstrated to be a useful tool for predicting flows at MS6 and will help facilitate implementation of the forbearance program.
3. Sanctuary Forest’s forbearance program has caused measurable increases in low flows in the Upper Mattole River since 2009, elevating extreme low flows, reducing the number of days MS6 was below the cutoff, and almost certainly improving low flow conditions for fish than would be the case in the absence of the program.
4. With growing participation in the forbearance program, additional benefits to summer low flows will continue to accrue and improve conditions for fish. If the recent trend of worsening droughts becomes a longer term trend, maintaining and augmenting forbearance in the Upper Mattole River, and indeed the entire redwood region, will become increasingly important for juvenile salmonids.

Acknowledgments

The Upper Mattole River monitoring program benefitted from the work of many volunteers over the years, and we thank them for their service. The storage and forbearance program was been funded primarily by the

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Evaluating the Ecological Trade-Offs of Riparian Thinning for Headwater Stream Ecosystems in Second-Growth Redwood Forests[1]

David Roon,[2] Jason Dunham,[3] Bret Harvey,[4] Ryan Bellmore,[5] Deanna Olson,[6] and Gordon Reeves[6]

After decades of intensive timber harvest and land use change that removed forests from the landscape, recent satellite data show that forest cover has increased in North America (Liu et al. 2015). However, these regenerating forests differ greatly in structure and composition than the forests that preceded them (McIntyre et al. 2015). This has been especially evident on the Pacific coast, where only 3 to 5 percent of old-growth coast redwood (*Sequoia sempervirens* (D. Don) Endl.) forest remains (Russell 2009). To address this, land managers are actively thinning second-growth forests to restore old-growth conditions (O'Hara et al. 2010, Teraoka and Keyes 2011). Whereas most thinning has taken place in upland forests, thinning also could accelerate recovery in second-growth riparian forests (Keyes and Teraoka 2014).

Riparian forests are highly connected to adjacent streams and rivers (Baxter et al. 2005, Gregory et al. 1991). For example, riparian forests shade stream channels, keeping temperatures cool for cold-water adapted species and contribute leaf litter and terrestrial insects that act as the primary sources of energy for aquatic macroinvertebrates, amphibians, and fish (Baxter et al. 2005, Vannote et al. 1980). Historical timber harvest practices which clearcut riparian forests disrupted these ecological processes, altering in-stream conditions with adverse effects on some sensitive species (Campbell and Doeg 1989). In response, contemporary forest management practices now require buffers to protect riparian forests (Marczak et al. 2010). Though such practices are intended to protect riparian forests, dense growth of young trees and early successional species can become dominant. In these cases, the question of actively managing these forests to more quickly restore late-successional forest structure and composition has been raised (Keyes and Teraoka 2014, Russell 2009).

It has been long understood that riparian forests provide inputs of organic matter that support aquatic species (Vannote et al. 1980); in addition, evidence shows that light is also an important driver of in-stream productivity (Kiffney et al. 2004). Previous studies have documented that increased light associated with opening riparian canopies catalyzed in-stream productivity at multiple trophic levels (Bilby and Bisson 1992, Wilzbach et al. 2005, Wootton 2012). However, this increase in aquatic productivity is often at the expense of increased stream temperature (Beschta and Taylor 1988). This ecological trade-off has caused some to hypothesize that a more subtle change in riparian forest cover, like those associated with thinning, could strike a balance by providing some increased light without substantially increasing stream temperature (Wilzbach et al. 2005).

As streams and the biota they support can be sensitive to terrestrial disturbances (Welsh and Ollivier 1998), it is essential that we understand how streams respond to changing riparian forest conditions (Warren et al. 2016). Therefore, before thinning treatments are applied to second-growth riparian forests it is essential that we understand the effects on headwater stream ecosystems. As a result, we are evaluating the effects of experimental riparian thinning treatments on: 1) canopy cover,

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light, and stream temperature; 2) stream-riparian food webs; and 3) growth and bioenergetics of stream amphibians and fishes. We also examine the extent to which site-level responses to riparian thinning are evident at larger spatial scales, including further downstream and across entire watersheds.

Our study is taking place in three headwater stream networks in second-growth redwood forests of coastal northern California. Two streams, the west and east forks of Tectah Creek, are located on private timber land owned by Green Diamond Resource Company and flow into the lower Klamath River. The third stream, the middle fork of Lost Man Creek, is located in Redwood National Park and flows into Redwood Creek. These streams drain two distinct land ownerships, but proposed riparian thinning treatments on both are motivated by many similar objectives and are conducted under a common set of regulatory requirements.

Riparian thinning treatments are occurring at seven locations distributed across these three watersheds. Riparian thinning treatments consist of a reduction to 50 percent canopy cover over the stream channel along a 100 to 200 m reach. To evaluate the effects of riparian thinning, data are collected following a Before-After-Control-Impact (BACI) study design. Data are collected immediately upstream and downstream of experimental treatment reaches to understand the potential responses in context to local conditions and if those responses extend further downstream. Data are also being collected seasonally to understand how thinning affects streams during different times of year.

In order to evaluate the effects of riparian thinning on streams, first we are measuring how changes in canopy cover and light affect stream temperature. Second, we are determining how the food webs in these streams are currently structured (riparian vs. freshwater pathways of productivity) and how that may change with thinning. To do this, we are collecting macroinvertebrates in the diets of coastal giant salamander and coastal cutthroat trout and using stable isotopes to discern if freshwater or riparian pathways support these food webs. Next we are examining how stream fish and amphibian communities respond to the thermal and trophic responses associated with the thinning treatments by measuring seasonal growth rates. We are also modeling bioenergetics for coastal cutthroat trout using the combined temperature and diet data. Finally, a food web systems dynamics model will assemble the composite abiotic and biotic data to provide a comprehensive perspective of how these streams are responding to riparian thinning.

As riparian forest conditions continue to change, it is likely that freshwater ecosystems will be affected (Warren et al. 2016). Our studies focus on the potential changes in thermal and trophic conditions that are most likely to interact in supporting important aquatic species that inhabit headwater stream ecosystems. By combining empirical data collection with contemporary approaches in spatial stream network, food web, and bioenergetics modeling, we hope to provide a more comprehensive understanding of how these stream ecosystems are responding to riparian thinning. We hope that data collected by this study will not only address existing knowledge gaps, providing crucial information for multiple stakeholders, but will also help inform future riparian forest management in redwood ecosystems.

Acknowledgments

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Effects of Logging Road Removal on Suspended Sediment Loads and Turbidity[1]

Randy D. Klein[2] and Vicki Ozaki[3]

Abstract

Poorly designed and unmaintained logging roads pose serious risks to aquatic ecosystems through sediment delivery from stream crossing failures and landslides. Redwood National Park (RNP) in northern coastal California has been implementing a restoration program for almost four decades, focused primarily on removing (decommissioning) abandoned logging roads on former commercial timberlands that were acquired by expansion of RNP onto large areas of cutover timberlands in 1978. Road decommissioning reduces sediment threats over the long term, however there are shorter-term impacts arising from ground disturbance that occurs when roads are removed. To better understand the magnitude and duration of sediment impacts, RNP conducted both onsite and offsite monitoring in a small watershed, Lost Man Creek, where nearly all legacy logging roads were removed from 2000 through 2010. Onsite turbidity increases were initially high at some locations, but diminished rapidly with time. Annual maximum peak discharge explained most of the variability in suspended sediment loads and turbidity at offsite gaging stations. Although restoration-driven increases in offsite turbidity and suspended sediment loads were likely detectable for part of the study period, legacy logging and natural sediment sources, triggered by larger storms, tended to confound the ability to quantify offsite effects with confidence. The year-to-year variability in road treatment intensity was high, and 2 consecutive years with high treatment intensity (2007 and 2008) likely caused concomitant, albeit brief, increases in suspended sediment loads. Since completion of restoration in 2010, sediment loads and turbidity have diminished rapidly despite the occurrence of the largest peak discharges of the study period, suggesting that the elimination of potential legacy sediment sources far outweighs sediment increases arising from road decommissioning.

Keywords: logging roads, road decommissioning, turbidity, suspended sediment.

Introduction

For almost 4 decades, Redwood National Park (RNP) has conducted an erosion control and prevention program to reduce long-term sediment delivery from logging roads into streams in the Redwood Creek basin, northern California. Lands acquired in the 1978 park expansion included logged over timberlands and an extensive network of poorly designed and often failing logging roads and skid trails that posed risks to the aquatic ecosystem, including threatened species such as coho salmon (*Oncorhynchus kisutch*). The goal of the ongoing program is to reduce logging road-related erosion and sedimentation of streams and thereby speed the recovery of the watershed and native ecosystems toward pre-disturbance conditions.

Logging road removal (decommissioning) in RNP typically consists of outsloping unstable road benches and excavating stream crossing fill material, thereby restoring channels close to their original courses, and placing excavated fill in stable locations to mimic pre-disturbance topography and drainage patterns. Preventing erosion and turbidity increases following road removal is physically impossible, but minimizing erosion and the resultant effects on downstream water quality is an

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important goal of the program. To that end, RNP has monitored post-decommissioning erosion and sediment responses using a variety of methods.

Studies of erosional responses to road removal work (Bloom 1998, Flanagan et al. 2012, Keppler et al. 2007; Klein 1984, 2008; Madej 2001, PWA 2005) and over 25 years of observations in RNP indicate that: 1) erosional responses are skewed, i.e., most sites will likely generate low to moderate erosion while a few generate relatively large volumes of erosion, 2) the greatest erosional responses occur within the first several years following road removal, and greatly diminish with time, 3) although crossing site conditions (e.g., slope steepness, contributing drainage area) exert strong control on post-decommissioning erosional responses, inexperience and/or inattentiveness on the part of those conducting road decommissioning can have a large effect as well (PWA 2005).

This paper departs from most previous work by emphasizing downstream (offsite) water quality response (continuous turbidity and suspended sediment loads), whereas most previous studies focused on onsite responses. Over an 11-year period (2000 to 2010), nearly all unmaintained legacy logging roads were decommissioned in Lost Man Creek (fig. 1). Post-road removal monitoring began in 2003, 2 years into the restoration project. The objectives of the monitoring program were to: 1) assess the effects of road removal on downstream turbidity and suspended sediment loads, and 2) quantify onsite erosion and turbidity responses from crossing removal.

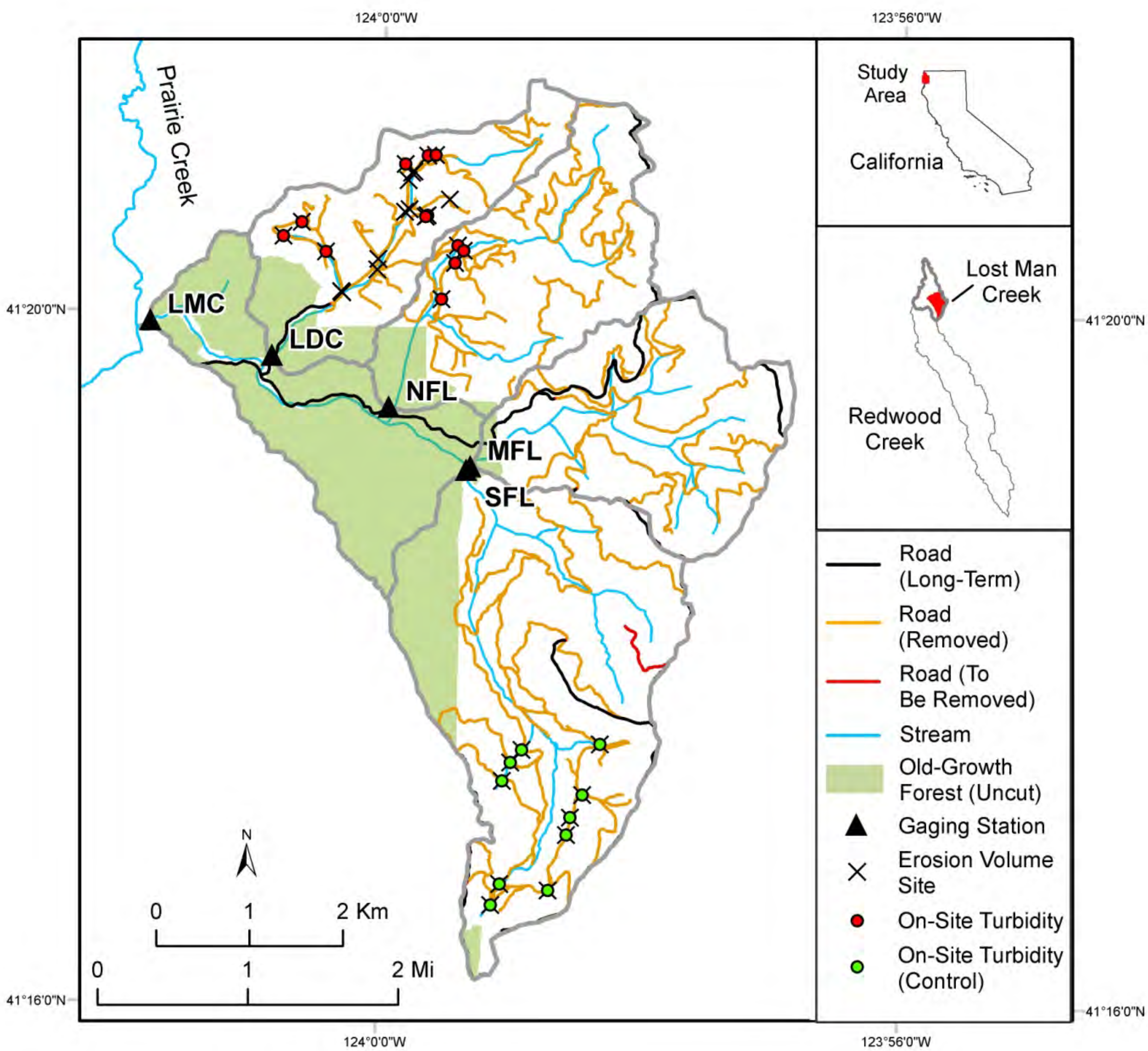

Figure 1—Lost Man Creek watershed and surrounding areas.

Study Area

Lost Man Creek is a major tributary to Prairie Creek which joins with Redwood Creek about 10 km (6 mi) upstream from its mouth at the Pacific Ocean (fig. 1). Mean annual precipitation in the study area is 1750 mm (69 inches). Coastal northern California has a Mediterranean climate, with most precipitation falling as rain from December through May. Streamflow varies greatly from wet to dry season with summer lows flows approaching zero and winter storm peaks commonly varying from 0.5 to 2 cms/km^2 (46 to 183 cfs/mi^2). Prairie Creek is recognized for its vital role in sustaining salmonid runs in Redwood Creek. Recent fish trapping in Prairie Creek and Redwood Creek just upstream from the mouth of Prairie Creek revealed that 1+ juvenile coho salmon production from Prairie Creek was “exponentially higher” than that of the larger Redwood Creek watershed area upstream (M. Sparkman, California Dept. of Fish and Wildlife, 2012, personal communication). A main objective of road decommissioning in Lost Man Creek is to sustain and fortify its role as a stronghold for regionally-declining salmonid species.

Like the vast majority of timberlands in north coastal California, most of Lost Man Creek was heavily logged and roaded during the 1950s to early 1970s, prior to implementation of state forest practice rules in 1975. Forest land use during this period consisted of largely unregulated road and landing construction and large areas were clearcut with intense ground disturbance from tractor yarding. Lost Man Creek experienced severe erosion and sediment delivery to streams during the 1955, 1964, and later floods and was highly disturbed by 1978, when the entire watershed became parkland.

Table 1 gives physical characteristics of Lost Man Creek and its sub-watersheds along with two other Prairie Creek sites used as controls. Figure 1 shows the Lost Man Creek watershed hydrography, stream gaging stations that provided data for this study, and logging roads removed between 2000 and 2010. A significant portion (about one-third) of Lost Man Creek is underlain by the Prairie Creek Formation (*PPpc*), lying mostly in the upper slopes of the tributary watersheds. Almost all of the rest of the watershed is underlain by the Coherent Unit of Lacks Creek (Franciscan Assemblage, *Kjfl*). Cashman et al. (1995) describe the *PPpc* as composed of ‘weakly consolidated shallow marine and alluvial sediments’ and the *Kjfl* as composed of interbedded sandstone and mudstone, with sandstone being the dominant rock type. Anecdotal observations indicate that hillslopes within some areas of the *PPpc* are particularly susceptible to surface erosion following disturbance due to low cohesion and coarse fragment content. Shallow landsliding is commonly associated with hillslopes underlain by rocks of the *Kjfl* (Cashman et al. 1995).

Table 1—Attributes of study watersheds (Lost Man Creek is located at the treatment basin outlet, thus data listed for this site integrates all upstream attributes)

Watershed (station code)	Drainage area (km^2)	Mean basin slope (%)	Basin relief (m)	%Prairie Creek Fm. (PPpc)	Area in old growth (%)	Pre-treatment road density (km/km^2)[a]
Lost Man Creek watershed treatment sites						
Larry Damm Creek (LDC)	4.8	43	475	70	19	2.8
North Fork Lost Man Creek (NFL)	5.7	44	471	48	10	3.6
Middle Fork Lost Man Creek (MFL)	5.8	45	518	38	2	2.7
South Fork Lost Man Creek (SFL)	10.6	43	731	6	14	2.6
Interfluves	4.7	41	431	15	96	1.0
Lost Man Creek at Hatchery (LMC)[b]	31.5	43	731	32	24	2.6
Control sites						
Little Lost Man Creek (LLM)	9.3	43	679	5	89	0.1
Prairie Creek above Boyes (PAB)	20.2	46	442	90	93	0.2

[a] Unpaved roads only; [b] Integrates all of the above sites.

Two long term stream gaging stations located elsewhere in Prairie Creek were used as controls for this study (LLM and PAB, table 1). PAB is underlain almost exclusively by *PPpc* and drains less steep terrain than Lost Man Creek and LLM is dominated by *Kjfl*. Both are predominantly pristine, old-growth redwood (*Sequoia sempervirens*(D. Don) Endl.) forest (table 1), but have minor influences from past road building.

Road Decommissioning History

Table 2 lists annual lengths of roads decommissioned and numbers of stream crossings excavated in Lost Man Creek by subwatershed. Approximately 95 km (59 mi) of logging haul roads existed in the watershed prior any treatments, with a total of 81.4 km (50 mi) removed over the life of the project, reducing road density in the watershed from about 3 to 0.4 km/km^2 (4.8 to 0.7 mi/mi^2). The remaining roads consist of a service road traversing the longitudinal axis of the watershed along the main channel and a short segment slated for future removal with low risk of failure (fig. 1).

Table 2—Road decommissioning history in Lost Man Creek and tributaries as of fall 2010 when work was completed (both lengths (km) of roads treated and number of stream crossings excavated (# exc.) are shown)

Year	LDC		MFL		NFL		SFL		LMC	
	# exc.	km	# exc.	km	# exc.	km	# exc.	km	# exc.	km
2000	0	0	7	3.6	18	6.6	18	3.9	43	14
2001	0	0	5	2.7	11	2.2	1	0.5	17	5.4
2002	23	4.7	0	0	16	4.7	0	0	39	9.4
2003	35	7.6	0	0	0	0	0	0	35	7.6
2004	3	0.8	0	0	0	0	0	0	3	0.8
2005	0	0	0	0	0	0	0	0	0	0
2006	0	0	0	0	0	0	23	1.8	23	1.8
2007	0	0.4	3	2.0	4	1.6	38	14	45	18.3
2008	0	0	13	4.5	23	7.1	16	3.8	52	15
2009	0	0	2	2.4	0	0	5	2.8	7	5.2
2010	0	0	2	1.5	0	0	3	1.9	5	3.4
Total	61	13.5	32	16.7	72	22.2	104	29.0	269	81.4

Figure 2 shows examples of road decommissioning at RNP and ground conditions following completion of the work. Despite often heavy mulching of bare soil and placement of large volumes of coarse woody debris in channels after earth-moving, erosion and sediment delivery to channels and downstream elevation of turbidity and suspended sediment loads is inevitable.

Figure 2—A. stream crossing being excavated; B. outsloped road with heavy mulching; C. recently excavated stream crossing with large woody debris placed within the exhumed channel; D. excavated stream crossing with a relatively large volume of woody debris added.

Methods

This study included both onsite monitoring at selected stream crossing excavation sites and offsite monitoring at a network of continuous stream gaging stations. Onsite monitoring occurred for 2 consecutive years (2002 and 2003) and was discontinued after erosional activity appeared to have ceased and it became clear that the results were consistent with other studies. Offsite monitoring was performed the entire length of this study, however, the four gaging stations at the sub-watershed scale were discontinued in 2011, 1 year after road removal work was completed. Offsite monitoring now consists of just the station at the watershed outlet (LMC, fig. 1), which will be continued indefinitely.

Onsite Monitoring at Excavated Stream Crossings

Onsite monitoring at crossings excavated in 2002 consisted of both turbidity sampling during three winter rainfall events during the first wet season, and estimates of eroded volumes after the first and second wet seasons, following excavation. The sites were selected to represent the range of crossing excavation sizes in Lost Man Creek, with additional consideration for the relative ease and safety of winter storm access. Eroded volumes were measured at 20 selected stream crossing excavations (fig. 1) following each of two consecutive wet seasons (2003 and 2004) by measuring erosional voids and tallying volumes.

A subset (n = 12) of the crossing excavations monitored for erosion volumes were selected for storm turbidity sampling in Larry Damm Creek (LDC) and North Fork Lost Man Creek (NFL) (fig. 1). Upstream and downstream stormwater samples were collected at these sites and turbidity was measured upon return to the lab. In addition, 11 untreated stream crossings in South Fork Lost Man Creek (SFL, fig. 1) were similarly monitored to compare with the excavated crossings, bringing the total number of onsite turbidity sampling locations to 23.

Offsite Monitoring

Offsite monitoring consisted of collecting continuous stage and turbidity data at gaging stations at the mouths of four main tributaries to Lost Man Creek and at the mouth of Lost Man Creek (fig. 1). Two of the sites (LMC and SFL, fig. 1) were set up as recording stations in water year 2003 (WY2003; a water year extends from October 1 from one year until September 30 of the following year) with automated equipment (data logger, stage sensor, and turbidity sensor) recording at 10-minute intervals. LMC is just upstream of its confluence with Prairie Creek, while SFL is the tributary basin farthest upstream from the confluence with Prairie Creek (fig. 1). The three tributaries between SFL and LMC (LDC, NFL, and MFL, fig. 1) were manual sampling sites in WY2003, but equipment for monitoring continuous turbidity and stage was installed in WY2004.

Suspended sediment samples at the Lost Man Creek stations were taken manually during storm periods at the five gaging stations. Most samples were depth- and width-integrated samples (DIS), but occasional point samples were also taken at a representative spot on the gaging section. Sampling at the control sites (PAB and LLM, table 1) was accomplished using an automated pumping sampler, with sampling increasing in frequency at higher turbidities, a method known as turbidity threshold sampling, or TTS (Lewis and Eads 2008).

Suspended sediment samples were transported to the laboratory soon after collection and analyzed for suspended sediment concentration (SSC) using filtration methods. Depending on the rainfall and storm magnitudes for a given year, 20 to 50 samples were taken at each gaging station and processed for suspended sediment concentration (SSC); more during the wetter years. Stage readings and discharge measurements were also taken manually, and discharge rating curves were constructed to convert recorded stage data to discharge. Equations for converting recorded turbidity to suspended sediment concentration (SSC) were developed by regressing SSC from the samples against simultaneous turbidity measurements. SSC to turbidity fits were always strong, with R-squared values always greater than 0.90 and often above 0.95. The regression equations were used to estimate 10-minute SSC values that were then multiplied by the associated 10-minute discharge volume to compute incremental suspended load. Annual suspended sediment loads were computed by summing 10-minute loads for each stormflow season.

As with flow duration analyses, the continuous turbidity data were sorted from largest to smallest and the percentages of time each value was equaled or exceeded (i.e., exceedence probability) were computed for December through May each year. The turbidity level exceeded 10 percent of the time provides a single value summarizing conditions for the entire season.

Results

Annual Maximum Peak Discharge

Instantaneous peak discharges and their recurrence intervals at LMC are shown in fig. 3. The largest annual maximum, about a 14-year event, occurred in WY2013. The three largest peaks occurred within the latter 4 years of the study (WY2003 to 2016).

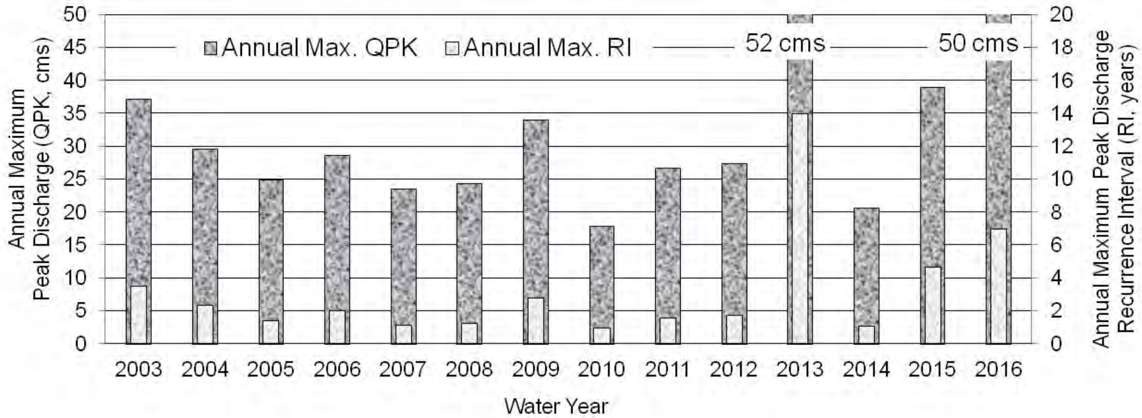

Figure 3—Annual maximum peak discharges and recurrence intervals at the Lost Man Creek (LMC) gaging station for 2003 to 2016.

Onsite Results

Figure 4 shows upstream-to-downstream turbidity increases during three storm samplings in the following rainy season at a subset of stream crossings excavated in 2002. Downstream turbidity ranged to over 5000 NTU during the first storm sampled (Dec. 14, 2002). Percent increases in turbidity through the crossings ranged from zero to nearly 2000 percent, with half the sites exhibiting increases exceeding 500 percent and the other half not exceeding 200 percent. Because no information existed for comparing hydrograph position (rising limb, peak, falling limb), an important context for the turbidity observations is lacking. Nonetheless, there was a strong trend for the most turbid sites (LDC2 to NFL7, fig. 4) to exhibit rapidly declining turbidity responses as the winter progressed, with the exception of Site 1 for which the largest turbidity response occurred in the second storm (Dec. 16, 2002). Turbidities at intact stream crossings exhibited virtually no increases from upstream to downstream.

Cumulative eroded volumes from excavated stream crossings after 2 years following treatment were distributed similar to many previous studies, with a strong skew toward low volumes and a small number with high eroded volumes. Figure 5 plots these results in order of decreasing eroded volumes. Mean for the sample was 19 m^3 (25 yd^3) ranging from zero (four sites) to 204 m^3 (267 yd^3).

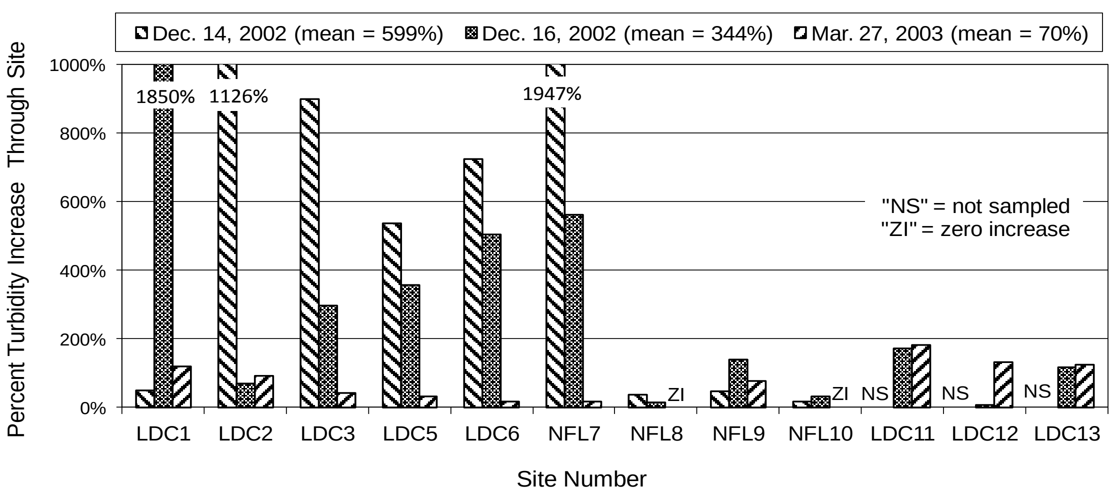

Figure 4—Turbidity increases through selected stream crossing excavations in Lost Man Creek, WY2003.

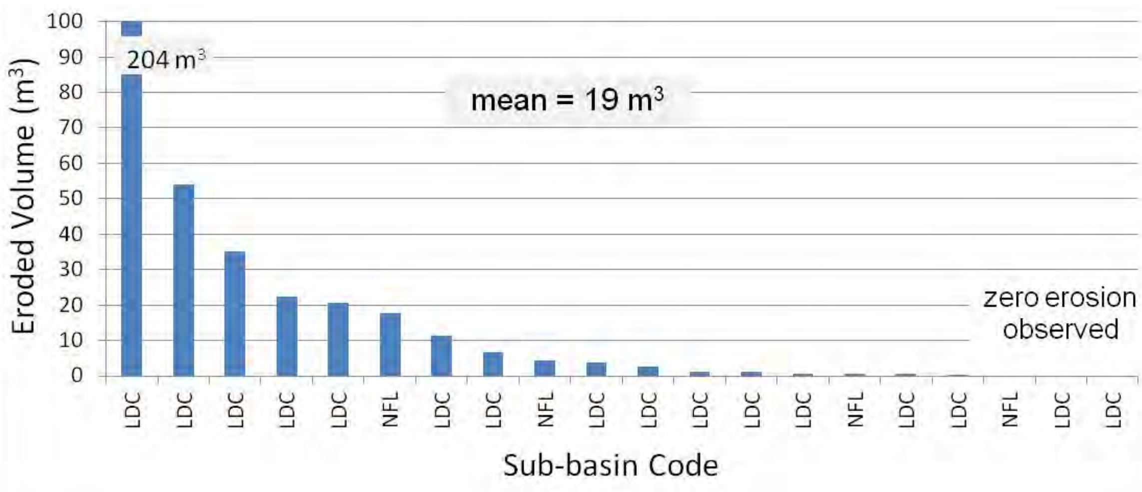

Figure 5—Eroded volumes at stream crossing excavations through selected stream crossing excavations in Lost Man Creek sub-basins (NFL = North Fork Lost Man Creek; LDC = Larry Damm Creek; note that site numbers denote sub-basin and correspond to the same numbering in fig. 4, above).

Offsite Results

Winter stormflow turbidity varied widely among sites and years (table 3). Turbidities at the 10 percent exceedence level were far lower at the control sites (PAB and LLM) than at any of the Lost Man Creek treatment sites, often by an order of magnitude or more, owing to the near pristine conditions of the contributing watershed. Turbidities among the Lost Man Creek sites were inconsistent from year to year, partly because of legacy logging and natural erosion processes triggered by larger storms.

Table 3—10 percent turbidity exceedence for Lost Man Creek and control sites (PAB and LLM), WY2003-16

			Turbidity at the 10 percent exceedance level				
WY	**SFL**	**MFL**	**NFL**	**LDC**	**LMC**	**PAB**	**LLM**
2003	37	---[a]	---	---	58	---	8
2004	14	21	19	25	19	4	4
2005	19	19	15	14	18	2	16
2006	30	25	28	19	33	4	8
2007	35	19	21	6	22	3	8
2008	29	16	15	12	21	1	8
2009	55	17	21	13	27	1	9
2010	28	16	18	10	16	1	7
2011	30	18	18	17	19	1	8
2012	---	---	---	---	20	1	9
2013	---	---	---	---	16	1	7
2014	---	---	---	---	11	4	6
2015	---	---	---	---	17	5	6
2016	---	---	---	---	35	7	9
Mean	31	19	19	15	24	3	8

[a] = Station not operated.

Annual suspended sediment loads per unit area for WY2003 to 2016 are shown in table 4 for the Lost Man Creek sites and the two control sites (PAB and LLM). The downstream-most site in Lost Man Creek (LMC) integrates all loads originating upstream, including MFL, NFL, LDC, SFL, and approximately 2.8 km^2 (1.1 mi^2) of ungaged interfluve areas. WY2003 and 2006 stand out as exceptionally high sediment load years, and at least some of the reason for this is explained by non-road related erosion and sediment delivery. In WY2002, the year before the first year of data collection, a debris torrent contributed high sediment volumes to Larry Damm Creek, which accounted for the high load downstream at LMC. A field reconnaissance in fall 2006 (near the end of WY2006) revealed the likely reason for the elevated LMC loads. Several hundred meters upstream of LMC, a concrete dam spillway was blocked by a large log causing the channel to breach the earthen walls and flow around the structure. Large volumes of sediment formerly composing the channel banks and bed and flood terraces had recently eroded as the channel diverted around the dam, and the suspendable (finer) portion of eroded sediment would have elevated WY2006 loads at LMC. After the effects of these two events subsided, annual loads were more consistent with peak discharges, although the relationship shifted over time (fig. 6).

Table 4—Suspended sediment loads (tonnes/km^2) for Lost Man Creek sites and control sites (PAB and LLM), WY2003-2016

	Annual suspended sediment load tonnes/km^2)						
WY	**SFL**	**MFL**	**NFL**	**LDC**	**LMC**	**PAB**	**LLM**
2003	59	---[a]	---	---	349	---	81
2004	140	78	76	163	117	9	25
2005	81	73	43	75	83	4	6
2006	112	127	135	104	186	12	29
2007	63	84	49	40	61	6	35
2008	99	38	64	38	86	4	9
2009	208	69	74	55	150	11	44
2010	53	33	35	16	36	3	9
2011	118	30	61	31	58	6	15
2012	---	---	---	---	93	7	22
2013	---	---	---	---	104	11	23
2014	---	---	---	---	32	9	14
2015	---	---	---	---	73	18	13
2016	---	---	---	---	129	8	32
Mean	104	67	67	65	111	8	25

[a] = Station not operated.

Figure 6 shows a time series of annual results from LMC for the study period. The influence of the 2002 debris torrent is clear, with WY2003 sediment loads far above all other years despite the relatively moderate peak discharge. Except for WY2009, there is no discernible direct relationship between treatment intensity (road length treated) and either loads or turbidity. The high load in WY2009 relative to peak discharge is coincident with a high rate of road treatment in the two prior seasons. An apparent temporal shift in the sediment load response to peak discharge is suggested in fig. 6 with loads declining relative to peak discharge in the latter years (WY2010 to 2016), a time when 96 percent of the planned road removal work had been completed and many potential sediment sources had been treated.

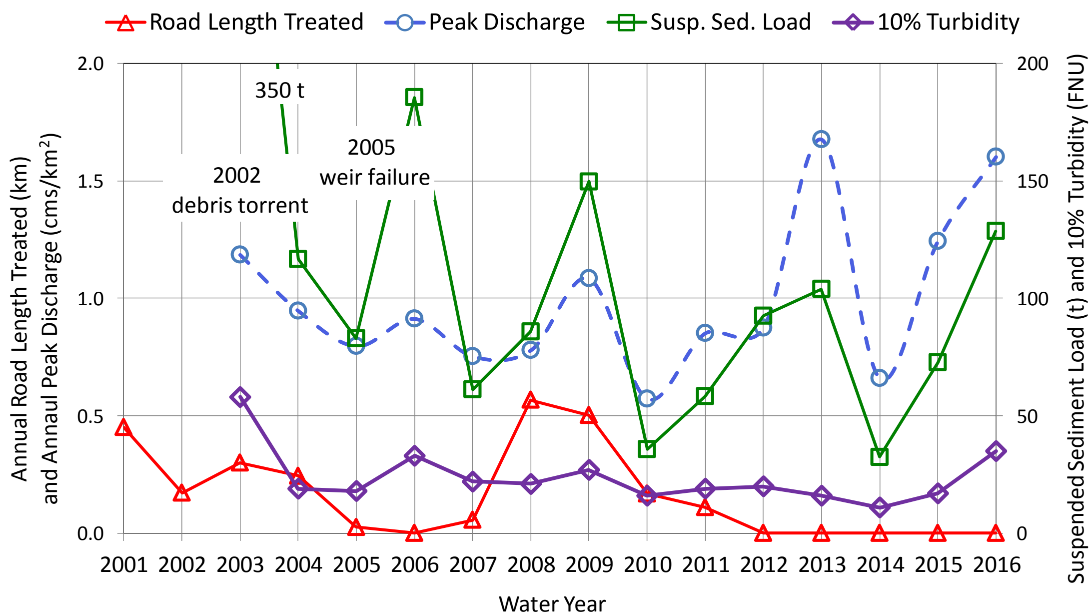

Figure 6—Upstream road lengths treated, peak discharges, sediment loads and turbidities for the Lost Man Creek watershed (LMC site), WY2001 to 20016.

Figure 7 plots these same data as scattergrams (with WY2003 and WY2006 omitted because of the two anomalous erosion events described above), partitioning the load and peak discharge data pairs into two groups (2003 to 2009 and 2010 to 2016). Simple linear regression on each period gave a line slope for the earlier period of over three times that of the later period, with both regressions yielding fairly high R-squared values. Road length treated (RLT) had no discernible relationship with loads.

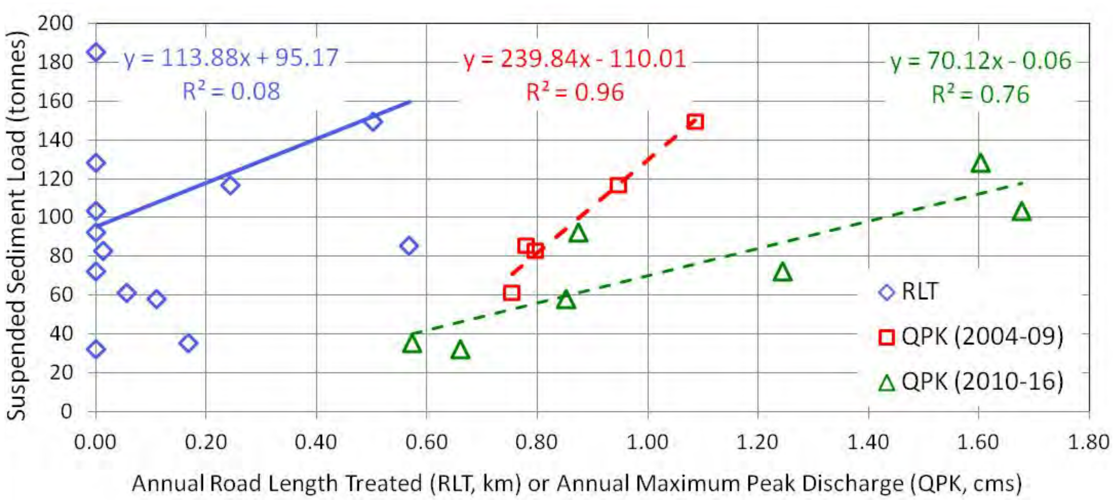

Figure 7—Road lengths treated (RLT) and peak discharges (QPK) versus annual loads for the Lost Man Creek watershed (LMC site), WY2004 to 2016.

Because of the differing geology, use of the control streams' (PAB and LLM) suspended sediment loads presented some difficulty: the treatment watersheds have mixtures of two geologic types with differing erodabilities in varying ratios (see table 1) while the controls are each dominated by one of the two (*Kjfl* in LLM and *PPpc* in PAB) and, as shown in table 4, had very different suspended sediment loads, with PAB far lower than LLM. Several analyses were performed to attempt to use data from the control sites to condition data from the treatment sites for examining road treatment effects. In one case, control site suspended sediment loads were subtracted from treatment site loads; in another, treatment site loads were divided by control site loads. The means of the loads for the two control sites also were compared with the treatment sites, but none of these analyses suggested a treatment effect.

Conclusions

Road treatments in Lost Man Creek were both extensive and, at times, intensive during the 11-year span of the restoration project. Erosion events unrelated to road treatments during the early years of this study (2003 and 2006) overshadowed and obscured any potential treatment effects during the subsequent high flow seasons. Later, peak discharge events exerted the dominant control over annual suspended sediment loads, with a shift in the relationship occurring after 2009 when loads declined relative to peak flows. By that time 96 percent of the roads planned for treatment had been treated, decreasing the number of unstable legacy sediment sources. Although some legacy and natural sediment delivery sources were likely still active in those years, a reduction in loads relative to peak discharges would be an expected outcome from road decommissioning. Thus, rather than providing clear evidence of water quality degradation because of disturbances associated with road treatment, this study's strongest conclusion is an apparent large improvement in water quality indicated by diminishing suspended sediment loads despite the three largest peak discharges of the study period occurring in within the most recent 4 years of the study (WY2013, 2015 and 2016).

In addition to the 2003 debris torrent in LDC and the dam failure upstream of LMC in 2006, failures on yet-to-be-treated roads (both road bench and crossing failures) were observed by RNP staff while driving and hiking areas of the Lost Man Creek watershed during the study period. Although not a complete inventory or systematic sampling of erosional features, it became clear that legacy sediment sources were a large, if unquantified, factor contributing to suspended sediment loads. In retrospect, confining monitoring to smaller areas in which complete inventories or sediment sources could have been made may have allowed a more complete understanding of the relative contribution of road decommissioning to suspended sediment loads in a legacy watershed.

The most definitive results of this study were those measured onsite, which showed sometimes very large erosional and turbidity responses at selected stream crossing excavations. Turbidity increased greatly passing through some stream crossing excavations during the first few storms of the season immediately following treatment and then dropped rapidly in later storms. The causes for differences among the monitored excavation sites are unknown, but the channel and side slope steepness, degree of mulching, and volume of coarse wood addition likely played key roles. Eroded volumes at excavated crossings exhibited the skew typical of previous work; most sites had low erosional responses, but a few were relatively large. Although intact stream crossings had essentially zero effect on stream turbidity during the brief 'snapshot' of monitoring done for this study, un-excavated stream crossings on unmaintained forest roads will eventually fail and degrade downstream water quality and habitat.

The relatively long duration of monitoring in Lost Man Creek (14 years as of this writing) allowed a sufficient length of time for the two opposing responses to road removal (increased sediment from ground disturbance, decreasing sediment from elimination of legacy sources) to play out. The gaging station at the watershed outlet (LMC, fig. 1) will be continued indefinitely and provide data to allow evaluation of the longer term effects of road removal. Future suspended sediment loads in Lost Man Creek are expected to continue to decline relative to peak discharges, punctuated by larger storms triggering episodic legacy and background erosion events and increasingly infrequent post-treatment adjustments from stream crossing excavation sites. A return to pristine conditions will not occur for decades, if not centuries, being dependent on a return to old growth forest conditions and exhaustion of erosion features from legacy logging. Nonetheless, road removal in this watershed has shortened that recovery time by permanently reducing the threat to the aquatic system from road-related legacy sediment impacts.

Acknowledgments

Funding for the Lost Man Monitoring program was provided by USDI, National Park Service, Redwood National Park (RNP). Staff with a private consulting firm, Graham Matthews and Associates, executed the monitoring program by contract with dedication and expertise during the first 8 years. RNP staff, including Tom Marquette, Carrie Jones, and Rachel Baker-De Kater among others, kept up data collection through 2016. Without their perseverance in carrying heavy loads through bad weather, and all the other, diverse jobs done by these people, this project would not have been possible, and we heartily thank them all.

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Long-Term Streamflow Trends on California's North Coast

J. Eli Asarian[1] and Jeffrey D. Walker[2]

Abstract

Using streamflow data from the U.S. Geological Survey, we assessed long-term (1953-2012) trends in streamflow on California's North Coast including many sites in the redwood region. The study area spans from the Smith River to the Mattole River and includes the Eel and Klamath-Trinity basins. Antecedent Precipitation Index (API) is a time-weighted summary of precipitation which provides high weight to recent precipitation and low weight to precipitation that occurred many months ago. We used a regression model of the relationship between API and streamflow to calculate "precipitation-adjusted streamflow", which statistically reduced the year-to-year fluctuations caused by variable precipitation and allowed evaluation of the underlying streamflow trend.

During the summer and early fall, streamflow and precipitation-adjusted streamflow have significantly declined in recent decades in many streams. With the exception of precipitation quantity, the methods used in this analysis do not allow individual quantification of the factors contributing to streamflow declines. However, the long-term streamflow gages include a diverse range of watershed and climate conditions. Thus, we can hypothesize about causal mechanisms by carefully examining the trends that have occurred in watersheds with different conditions and histories. The most pristine surface-runoff dominated watersheds within the study area showed no decreases in precipitation-adjusted streamflow during the summer months. This suggests that streamflow decreases at other sites are more likely the result of increased water withdrawals and/or changes in vegetation structure/composition than climate factors other than precipitation quantity (e.g., decreased fog and increased air temperature).

Our results appear to support the hypothesis that water withdrawals are an important factor, but not the only one, contributing to the declining trends in precipitation-adjusted streamflows. There were few declines in those watersheds with the least amount of diversions.

Because evapotranspiration is such a large portion of the annual water budget, small changes in evapotranspiration have the potential for large effects on summer streamflows. The North Coast's forests have undergone substantial changes in the past century as a result of timber harvest and fire suppression. Without fire or mechanical intervention, Douglas-fir trees are invading prairies and oak woodlands, potentially increasing evapotranspiration. Young dense forests within the region may be in a state of maximum evapotranspiration. Bull Creek (tributary to the South Fork Eel River) provides evidence supporting the hypothesis that vegetation change is contributing to streamflow declines, because despite lacking water diversions it had among the largest declines in precipitation-adjusted streamflow, coinciding with the regeneration of its forests following intensive logging that occurred prior to installation of the stream gage in 1960.

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SESSION 4 – Genetics and Restoration

Adaptation to Climate Change? Moving Coast Redwood Seedlings Northward and Inland[1]

Christa M. Dagley,[2] John-Pascal Berrill,[2] Forrest T. Johnson,[2] and Lucy P. Kerhoulas[2]

Abstract

Insight into genetic variation in trees may provide opportunities to select for genotypes that are better adapted to new locations and future climate conditions. We established a field test at two sites in Humboldt County, California to study the performance of coast redwood (*Sequoia sempervirens* (D. Don) Endl.) under assisted migration. Both test sites were near the eastern (inland) limit of coast redwood's range and had no naturally occurring redwood. Seed were collected from redwood trees on dry, hot ridges and upper slopes from the southernmost populations, and combined with redwood seed from Mendocino County and seed and tissue culture clones from Humboldt County. A total of 34 different clones, open-pollinated families, and commercial seedlots were planted in 27 replicates at each test site using an interlocking hexagonal design. Health, instances of damage, and total height of every seedling was recorded annually since planting in 2010. Caliper (basal diameter) was also measured annually three times beginning in spring 2014, giving basal diameter increment for each tree. Water stress was assessed for each young tree ($n \approx 2000$ trees) in the summer of 2015 using a pressure bomb. Performance of progeny planted at each test site varied among regions-of-origin, forest-of-origin, and among families of seedlings from individual open-pollinated parent trees. Results were counter to our expectation that seedlings originating from parents located at the warmer and drier southern extremes of redwood's range would perform best on the more extreme test site (higher elevation, no fog) in Humboldt County. However, high variances within families and clones suggested that genetic effects may have been obscured by other sources of variability at this early age.

Keywords: assisted migration, climate change adaptation, forest restoration, genetics, reforestation, seed collection

Introduction

Insight into genetic variation in trees informs forest conservation, restoration strategies, and assisted migration efforts in the face of a changing climate. Investigators have measured genetic variation through the use of molecular markers and analysis of data from common garden experiments in an attempt to divide phenotypic variability into genetic and environmental components. Genetic variation is hierarchical. Tree-to-tree variation among trees in the same stand is usually a major source of genetic variation. Many tree species also exhibit geographic variation among provenances, especially among distant populations in widespread species that occupy different climates (White et al. 2007).

Genetic variation among provenances may be a result of natural selection acting on local populations, making them adapt to local conditions. O'Brien et al. (2007) found that traits of jarrah (*Eucalyptus marginata* Sm.) varied over geographical scales; their findings suggested seed transfer from high to low rainfall sites would be accompanied by increased mean growth rate for the site. However, under drier conditions the high growth rates may be unsustainable and result in increased mortality over longer time intervals. This suggested that seed sourced from different locales may be better or worse suited for conservation and/or restoration projects. Millar and Libby (1989, 1991) also

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noted the importance of considering the environmental conditions and provenance variability in conservation planning and restoration activities.

Coast redwood (*Sequoia sempervirens* (D. Don) Endl.) is a species with restricted natural range. The current geographic range is confined to a relatively narrow coastal strip from the southwestern corner of Oregon (42°09' N. latitude) to Monterey County in central California (35°41' N. latitude) (Burns and Honkala 1990). In general, along this north-to-south gradient, rainfall declines and potential evapotranspiration increases, yet trees of similar size can grow at similar rates in tall forests (Sillett et al. 2015). Less coastal fog and warmer growing season temperatures are found inland, further away from the Pacific coast, creating another potential evapotranspiration gradient. Throughout this range, redwoods are found with different assemblages of associated species and different upper limits of stand density in terms of leaf area index and total live above-ground biomass (Van Pelt et al. 2016). For example, the northern redwood forests of Del Norte and Humboldt counties often have coast redwood, coast Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*), grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.), Sitka spruce (*Picea sitchensis* (Bong.) Carrière), western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh), and Pacific madrone (*Arbutus menziesii* Pursh). In comparison, the drier southern redwood forests often comprise a mix of redwood, Douglas-fir and hardwoods including California bay (*Umbellularia californica* (Hook. & Arn.) Nutt.), coast live oak (*Quercus agrifolia* Née), and tanoak (Olson et al. 1990, Zinke 1988). Redwood forms a more continuous belt north of the Sonoma-Mendocino County line, and becomes more isolated in its disjunct populations further south (Douhovnikoff and Dodd 2011).

Unknown is the genetic variation within or among coast redwood populations that make them better or worse adapted to different climates. For example, if drought tolerance and water-use efficiency are found to be heritable traits, we can select for these characteristics and raise seedlings for reforestation on marginal sites or in areas where adverse changes in climate are forecast. Most conifers have only 20 to 24 chromosomes. However redwood has 66 chromosomes making it hexaploid (Saylor and Simons 1970, Sclarbaum and Tsuchiya 1984). As a result, within a single individual it is possible for allelic variation to occur (alternative forms of the same gene). The wide variation of within-family genetic variability was clearly demonstrated by Rogers (1994) by showing gametes to have one, two, or in some cases three different alleles of a gene coding for a particular enzyme. Therefore, any redwood provenance might be capable of adapting to new sites and climates.

We implemented a large field experiment to determine if progeny of redwood trees growing in the hottest/driest parts of their natural range may be better-adapted to hot, dry conditions and hold promise for restoration and resistance to climate change. We designed this study to answer the following questions about the progeny of coast redwoods originating from different climates: 1) Do progeny growing at test sites differ in growth and/or water stress, and 2) Do progeny exhibit differences between and within regions of origin in terms of growth and/or water stress? When planting redwood on an extreme site (hot, dry summers; cold winters), we hypothesized that progeny originating from the southernmost populations where redwoods experienced hotter and drier summers would outperform seedlings or clones originating from the more northern, moister coastal locations. Conversely, we expected seedlings adapted to moister locales to outperform the southern redwoods moved northward and planted on a northern site with some coastal fog. This paper describes the implementation of the experiment and presents preliminary data on tree size, growth, and water stress at two test sites for redwood propagules from different regions, forests, and open-pollinated families.

Methods

Coast redwood seed cones were collected from southern and inland locations where redwood experiences hotter and drier conditions (fig. 1). These southern redwoods were mixed with shrubs or hardwoods and showed characteristics of thick, flaking bark with sparse crowns. In the autumn of 2009, shallow increment cores were taken from candidate parent trees of dominant or co-dominant crown position exhibiting favorable form characteristics (i.e., straight, without excessive forking; small horizontal branching). A subset of these trees had increment cores exhibiting favorable radial growth and wood properties, defined as relatively wide growth rings (rapid growth), darker heartwood coloration, and higher density (resistance of core sample to fingernail pressure). These superior trees were selected for seed collection.

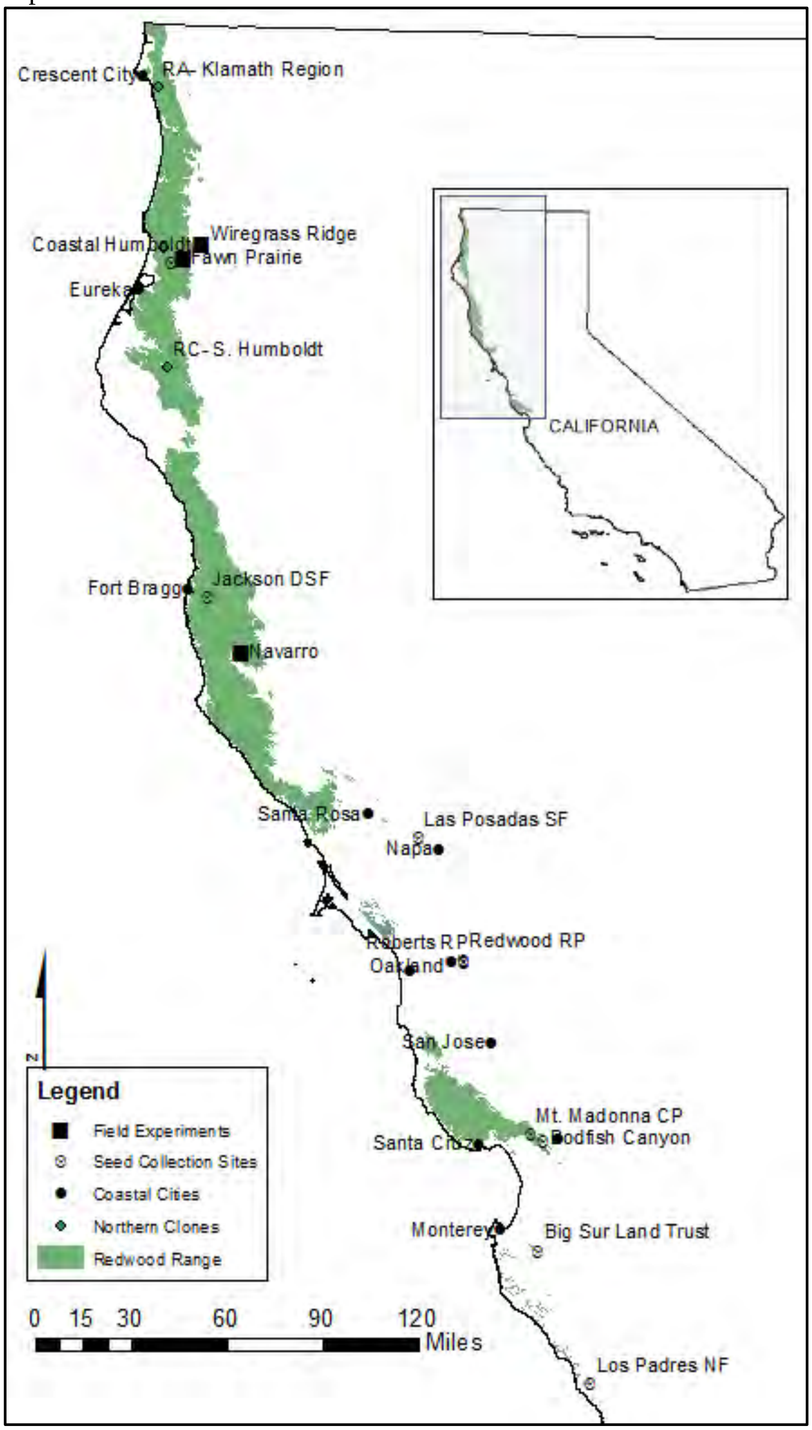

Figure 1—Location of seed cone collection locations, clone origins, and field test sites.

Cones were concentrated at the tops of the selected trees. Tree climbers used pole pruners to clip branchets bearing clusters of cones. Individual trees yielded 3 to 18 liters of cones with each cone having up to 50 seeds. Seed cones from each parent tree were kept separate. The cones were processed by the CAL FIRE L.A. Moran Reforestation Center. Open-pollinated families (separate seedlot for each parent tree) were tested for germination rate, cold stratified for 1 month, and planted at a rate of four seeds per styro-15 container. Seed from a commercial seed collection (multiple open-pollinated families) on Jackson Demonstration State Forest in Mendocino County were also sown in styro-15 blocks at the same time and location. The 1-0 seedlings were raised in a commercial nursery at a hot, dry location outside of redwood's natural range at Cottage Grove, Oregon. Additional planting stock for the field tests were open-pollinated seedlings and tissue culture clones from Humboldt County, raised in styro-15 containers at Green Diamond Resource Company's Korbell Nursery, Humboldt County, California.

The seedlings from Mendocino and Humboldt counties and tissue culture clones from Humboldt County were labelled and planted in winter 2010-2011 at two test sites in Humboldt County using the replicated interlocking hexagonal experimental design of UC Berkeley Professor Emeritus William J. Libby (fig. 1). Both test sites were Douglas-fir/tanoak sites with no naturally occurring redwood and were approximately 20 km (12.5 miles) from the Pacific coast, near the eastern (inland) limit of coast redwood's range. The first test site, Fawn Prairie (FP), is at 560 m (1,847 ft) elevation and receives some summer fog. The second test site, Wiregrass Ridge (WG), is on an exposed ridge at 1000 m (3,269 ft) elevation. We considered the WG test site to be the more extreme test site because it does not receive coastal fog and soon after planting was covered by 30 cm of snow. The following year, a third test site was established at a hot dry site near Navarro in Mendocino County. This site suffered drought and was abandoned due to widespread mortality in the first growing season.

The experimental area at each test site covered 1.2 ha (2.7 ac) and included 980 planted seedlings. A total of 34 genotypes were planted at each test site which included 27 replicates (nine replicate blocks with three replicates within each block), each containing the 34 genotypes. Two outside border rows surrounded the experimental area at each test site. The hexagonal design provides for two future thinning operations (each removing one of the three complete replicates, i.e., cut 33 percent of the trees at each thinning) that preserve even spacing between neighbors. Site preparation included spraying tanoak stumps and/or sprouts with triclopyr. Manual weeding in the vicinity of each planted redwood was undertaken annually.

Health, instances of damage, and total height to the nearest centimeter (cm) were recorded for every seedling annually since planting in 2010. Health status was coded as healthy, sick, moribund, or dead based on visual assessment. Instances of damage recorded included deer browsing, dead tops, or broken tops. Early in the spring of 2014, a form pruning on seedlings with multiple stems and/or basal sprouting occurred. At this time, basal diameter (termed 'caliper') was also measured to nearest 0.1 mm using digital calipers immediately above any swelling associated with the root collar. Caliper was re-measured at end of the 2014 and 2015 growing seasons. The difference between repeat measurements gave annual growth in terms of caliper increment. Converting caliper data to cross-sectional stem area gave basal increment.

Data were sorted and summarized by test site, forest-of-origin and region-of-origin, and by open-pollinated family or individual clone. Assuming that the slowest growing trees had problems such as poor genetics, poor establishment, or damage, we excluded from the analysis the shortest 25 percent of trees from each family or clone based on height in 2016.

Water potential was measured using a Scholander pressure bomb during the dry summer months of 2015. Samples were taken repeatedly throughout the day. Water potential data were plotted to demonstrate how water stress changed over the course of a day at the test sites inside and outside redwood's range.

Results and Discussion

Regional differences were examined based on the average tree height after five full growing seasons (hereafter referred to as age-5 years), and 2-year caliper increment and basal increments for both test sites. Growth was much lower on average (41 percent, 28 percent, and 55 percent less in height, caliper increment, and basal increment, respectively) at the more extreme test site (WG) versus FP (table 1).

Table 1—Regional averages of age-5 height, and 2-yr caliper increment and basal increment, for Fawn Prairie (FP) and Wiregrass Ridge (WG) test sites (standard deviation in parentheses)

Test Site	Region	Height (cm)		Caliper increment (mm yr^{-1})		Basal increment (cm^2 yr^{-1})	
FP	Humboldt	243.2	(80.8)	11.52	(5.1)	7.81	(7.9)
FP	Mendocino	233.0	(89.3)	12.70	(6.3)	8.67	(6.3)
FP	Southern	232.4	(79.4)	12.15	(6.5)	8.12	(7.7)
WG	Humboldt	136.7	(52.4)	9.49	(4.7)	4.12	(3.1)
WG	Mendocino	140.6	(41.4)	8.42	(5.0)	3.52	(3.4)
WG	Southern	137.7	(52.9)	8.25	(4.7)	3.42	(3.5)

As a group, clones and seedlings from the northern region (Humboldt) that were planted at FP showed slightly better performance in height, however, the central region (Mendocino) had better performance in caliper and basal increment. When the redwoods were planted on the harsher test site (WG) the opposite result occurred; the central region seedlings showed the best performance in height and the northern region had the highest caliper and basal increment (table 1).

Table 2—Fawn Prairie test site showing averages of age-5 height, and 2-yr caliper increment and basal increment, and top rankings in basal increment (standard deviation in parentheses)

Forest	Height (cm)		Caliper increment (mm yr^{-1})		Basal increment (cm^2 yr^{-1})		rank
GD Clone (N. Humboldt)	228.5	(65.6)	13.07	(4.3)	8.80	(5.0)	
Central Humboldt Seedlot	238.8	(71.1)	11.40	(4.4)	6.43	(5.1)	
GD Clone (C. Humboldt)	245.9	(84.2)	12.96	(5.2)	9.13	(5.9)	3
GD Clone (S. Humboldt)	265.0	(102.2)	12.88	(6.7)	9.70	(9.0)	1
Mendocino Seedlot (JDSF)	233.0	(89.3)	11.52	(6.3)	7.81	(7.9)	
Inland Napa (Las Posadas SF)	228.8	(78.7)	8.79	(4.2)	4.80	(4.4)	
Roberts Park (Oakland Hills)	243.7	(81.1)	12.67	(6.9)	9.35	(9.1)	2
Redwood Park (Oakland Hills)	238.8	(57.6)	12.39	(6.2)	8.00	(7.0)	
Mount Madonna (Watsonville)	229.3	(76.4)	13.13	(6.5)	8.70	(7.0)	
Gilroy (Bodfish Canyon)	243.3	(90.0)	12.63	(7.3)	9.07	(8.7)	4
Big Sur Land Trust	219.6	(89.5)	13.12	(6.6)	8.46	(7.1)	
Los Padres National Forest	204.6	(71.8)	10.58	(4.9)	5.77	(5.3)	

Table 3—Wiregrass Ridge test site showing averages of age-5 height, and 2-yr caliper increment and basal increment, and top rankings for basal increment (standard deviation in parentheses)

Forest	Height (cm)		Caliper increment (mm yr^{-1})		Basal increment (cm^2 yr^{-1})		rank
GD Clone (N. Humboldt)	140.6	(49.0)	8.99	(5.0)	3.84	(3.2)	
Central Humboldt Seedlot	129.7	(67.5)	7.26	(4.0)	2.90	(2.4)	
GD Clone (C. Humboldt)	137.1	(48.6)	8.50	(4.2)	3.51	(2.7)	
GD Clone (S. Humboldt)	135.4	(53.7)	8.21	(5.3)	3.46	(3.9)	
Mendocino Seedlot (JDSF)	140.6	(41.4)	9.49	(5.0)	4.12	(3.4)	2
Inland Napa (Las Posadas SF)	134.3	(53.6)	6.57	(3.0)	2.41	(1.9)	
Roberts Park (Oakland Hills)	143.4	(54.0)	8.39	(5.2)	3.55	(3.9)	4
Redwood Park (Oakland Hills)	148.7	(53.8)	10.12	(4.7)	4.59	(3.8)	1
Mount Madonna (Watsonville)	145.6	(54.8)	8.89	(4.9)	3.96	(3.7)	3
Gilroy (Bodfish Canyon)	133.4	(56.0)	7.91	(4.3)	3.17	(3.3)	
Big Sur Land Trust	125.5	(35.7)	7.75	(4.7)	2.99	(3.2)	
Los Padres National Forest	121.3	(43.6)	6.71	(4.1)	2.30	(2.5)	

Two year averages in basal increment and caliper increment, along with 5 year height averages were summarized for both test sites based on forest-of-origin. Basal increment rankings varied for both test sites showing differences in seedling performance when planted in different climates. Clones that were collected from southern Humboldt County showed the highest basal increment when planted at FP (table 2). In contrast, when planted at the more extreme site (WG), the greatest growth came from seed collected further to the south in Redwood Park, an area in the Oakland Hills (table 3).

Individual family or clone averages were also summarized for height, caliper increment, and basal increment at the test sites (tables 4 and 5). The two test sites showed differences in top performers within all three categories; however the high variances within families and clones suggested that genetic effects may have been obscured by other sources of variability at this early age.

Table 4—Redwood clones and O-P families (n = 32) listed in order of their place of origins from north to south and planted at Fawn Prairie (FP) test sites showing means and top rankings for age-5 height, and 2-yr caliper increment and basal increment (BI) (standard deviation in parentheses)

Clone*/	Height			Caliper increment			Basal increment		
family	(cm)		rank	(mm yr^{-1})		rank	(cm^2 yr^{-1})		rank
RA20*	215.9	(54.1)		14.53	(3.5)	4	10.08	(5.0)	
RA38*	192.5	(75.6)		8.21	(3.6)		3.99	(2.1)	
RA60*	273.7	(39.6)	2	14.70	(3.1)	1	10.57	(4.4)	
O92**	238.8	(71.1)		11.40	(4.4)		6.43	(5.1)	
RB1*	215.5	(86.9)		13.76	(4.9)		9.22	(4.3)	
RB2*	266.1	(78.1)		12.27	(5.1)		9.00	(5.9)	
RB54*	259.1	(85.8)		12.86	(6.1)		9.18	(7.8)	
RC59*	234.5	(66.7)		11.22	(5.0)		6.37	(3.4)	
RC64*	298.5	(126.0)	1	14.54	(8.0)	3	13.03	(11.6)	1
C**	233.0	(89.3)		11.52	(6.3)		7.81	(7.9)	
E3	228.8	(78.7)		8.79	(4.2)		4.80	(4.4)	
D1	256.4	(67.5)		13.68	(6.3)		9.55	(6.5)	
D2	248.4	(86.4)		12.67	(7.8)		10.34	(11.0)	
D3	215.9	(104.3)		12.80	(7.5)		10.05	(10.7)	
D5	266.2	(78.1)		14.14	(8.0)		10.83	(10.9)	3
D6	234.0	(62.1)		10.00	(4.5)		5.97	(5.1)	
R1	232.5	(72.5)		12.57	(7.6)		8.02	(8.6)	
R2	246.3	(50.8)		12.52	(5.1)		8.74	(6.4)	
R4	238.2	(47.8)		12.05	(5.8)		7.25	(5.9)	
M1	230.0	(78.6)		14.09	(6.2)		9.16	(5.9)	
M2	230.0	(74.0)		12.34	(5.3)		7.42	(5.2)	
M3	227.9	(80.2)		12.86	(8.0)		9.49	(9.6)	
B1	249.1	(94.6)		14.69	(8.7)	2	11.14	(10.8)	2
B2	241.1	(80.3)		12.05	(7.6)		7.99	(8.5)	
B3	268.2	(82.9)	4	14.51	(6.4)		10.59	(8.6)	4
B5	185.2	(85.6)		9.62	(6.3)		5.86	(6.8)	
B7	270.5	(86.8)	3	12.31	(7.3)		9.81	(8.4)	
BSL1	219.6	(89.5)		13.12	(6.6)		8.46	(7.1)	
LP1	209.1	(74.6)		11.06	(2.8)		5.66	(3.0)	
LP2	180.8	(80.7)		10.80	(6.2)		6.24	(6.7)	
LP3	200.4	(62.5)		8.89	(5.5)		4.96	(6.8)	
LP4	227.6	(66.2)		11.73	(4.8)		6.38	(4.3)	

**Seedlot from multiple parent trees in Humboldt County 092 seed zone or Jackson Demonstration State Forest (C).

Table 5—Redwood clones and O-P families (n = 31) listed in order of their place of origins from north to south and planted at Wiregrass Ridge (WG) test sites showing means and top rankings for age-5 height, and 2-yr caliper increment and basal increment (standard deviation in parentheses)

Clone*/ family	Height (cm)		rank	Caliper increment (mm yr^{-1})		rank	Basal increment (cm^2 yr^{-1})		rank
RA20*	154.4	(57.73)		11.16	(5.0)	2	5.52	(4.0)	2
RA38*	140.8	(42.49)		8.77	(2.1)		3.56	(3.1)	
RA60*	127.9	(49.04)		7.23	(4.4)		2.57	(2.0)	
O92**	129.7	(67.54)		7.26	(5.1)		2.90	(2.4)	
RB1*	137.6	(53.37)		9.01	(4.3)		3.94	(2.7)	
RB2*	115.8	(37.66)		6.14	(5.9)		2.14	(2.0)	
RB54*	155.8	(48.66)	4	9.83	(7.8)		4.12	(2.9)	
RC59*	131.5	(55.06)		7.74	(3.4)		3.14	(4.0)	
RC64*	138.7	(54.75)		8.63	(11.6)		3.74	(3.9)	
C**	140.6	(41.36)		9.49	(7.9)		4.12	(3.4)	
E3	134.3	(53.63)		6.57	(4.4)		2.41	(1.9)	
D1	151.3	(51.38)		9.41	(6.5)		4.03	(4.0)	
D2	131.7	(50.53)		7.37	(11.0)		2.97	(3.4)	
D3	140.4	(65.05)		8.63	(10.7)		3.81	(5.2)	
D5	157.1	(43.54)	3	7.68	(10.9)		3.10	(2.0)	
D6	137.8	(58.74)		8.76	(5.1)		3.77	(4.5)	
R1	169.2	(59.99)	1	11.22	(8.6)	1	5.55	(4.6)	1
R2	145.3	(42.73)		9.51	(6.4)		4.36	(3.3)	4
R4	131.3	(52.30)		9.63	(5.9)	4	3.80	(3.5)	
M1	132.1	(42.74)		8.72	(5.9)		3.72	(3.4)	
M2	167.0	(63.18)	2	10.20	(5.2)	3	4.80	(4.2)	3
M3	138.0	(52.62)		7.83	(9.6)		3.38	(3.3)	
B1	146.4	(46.66)		9.18	(10.8)		3.98	(3.3)	
B2	136.9	(51.40)		7.82	(8.5)		3.20	(4.1)	
B3	129.3	(55.42)		7.87	(8.6)		2.86	(3.5)	
B5	115.5	(58.37)		6.93	(6.8)		2.60	(2.7)	
B7	139.6	(67.34)		7.73	(8.4)		3.27	(3.0)	
BSL1	125.5	(35.74)		7.75	(7.1)		2.99	(3.2)	
LP1	117.0	(50.67)		6.00	(3.0)		1.87	(1.9)	
LP2	122.4	(37.82)		8.09	(6.7)		3.23	(3.5)	
LP3	124.3	(42.86)		6.09	(6.8)		1.85	(1.6)	

**Seedlot from multiple parent trees in Humboldt County 092 seed zone or Jackson Demonstration State Forest (C).

Water potential samples taken repeatedly throughout the day during the summer months of 2015 revealed how water stress changed over the course of a day. Both test sites were found to be more similar in midday values than anticipated (fig. 2). The FP test site showed slightly more relaxed predawn values, although midday values were similar to WG, which suggested that stomata were conservatively regulating plant water status by controlling the rate of water loss to the atmosphere. This implied that redwoods were water-use efficient (i.e., exhibit isohydric stomatal regulation). In contrast, Ambrose et al. (2015) described redwood as relatively anisohydric. This conclusion was reached in part because they found redwood seedlings exposed to severe drought conditions to reach below -5.3 MPa during the daytime. Their study took place in a greenhouse where they had control over soil moisture conditions and could impose a severe drought condition. In our field tests, we found redwood stomatal closure preventing daytime water potential from going below -2.5 MPa. Our findings may differ simply because neither of our sites reached a severe drought state during the sampling period.

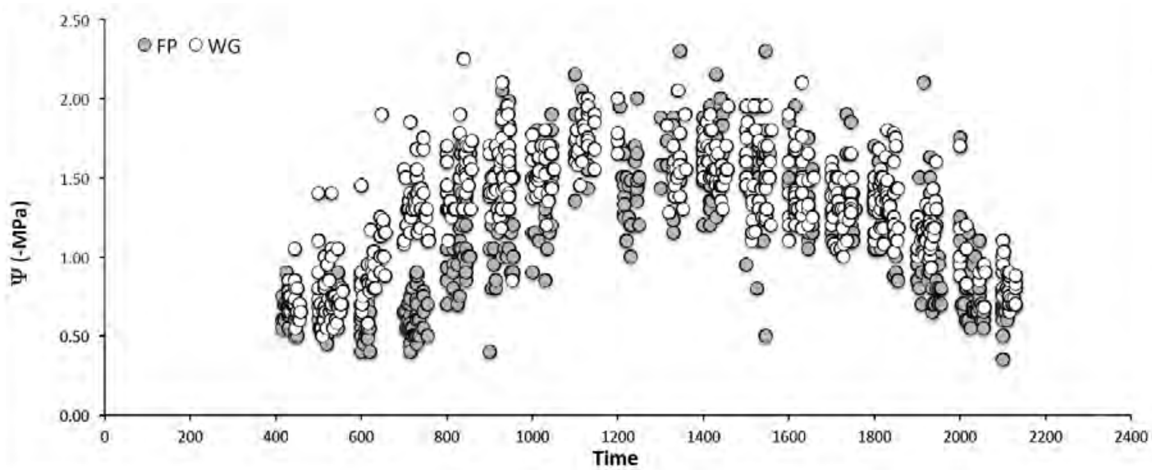

Figure 2—Diurnal water potential for redwood seedlings at two study sites Fawn Prairie (FP) and Wire Grass (WG) in mid-summer. Values show similar midday values at both sites.

Our next step is to analyze the summer 2015 water potential data we collected for every young redwood at both test sites (n ≈ 2000 trees) for among-region, among-forest, and among-family or among-clone differences in water stress. Ambrose et al. (2015) found no significant differences in water potential, hydraulic function, or growth and only minor differences in leaf gas exchange among redwood seedlings from Del Norte, Mendocino, and Santa Cruz counties. However, Anekonda et al. (1994) reported significant differences in metabolic response to temperature among redwood populations. Unlike these experiments performed under controlled conditions, our field test of seedlings from different populations appeared to have suffered from confounding environmental variations (e.g., microsite effects causing variability within families) which may dissipate with advancing age. Our preliminary findings appear to mirror field tests of redwood clones planted far outside its range (in New Zealand), where Meason et al. (2016) found no strong trends between growth, provenance, or provenance site characteristics. Therefore, we recommend continued monitoring and analysis of genetic variation in growth of our open-pollinated seedlings and tissue culture clones at field tests inside and outside redwood's range.

Conclusions

Progeny of some southern redwood families had high-ranking performance at both test sites. Different clones performed well at each test site. However high variability in growth among progeny from the same parent tree and progeny from the same forest suggested that other factors (e.g., microsite effects, weed competition) within each test site may have overshadowed genetic differences. Therefore, we recommend repeatedly re-measuring trees to later ages to get estimates of increment that may reveal within- and among-family differences in redwood performance after assisted migration northward onto different sites experiencing different climates.

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Comparing Growth and Form of Coast Redwood Selfs and Outcrosses[1]

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Abstract

We now report 22 years of new data and observations from the third of three small projects evaluating the effects of inbreeding in coast redwood (*Sequoia sempervirens* (D. Don) Endl.). We also briefly summarize previously-reported effects of inbreeding on redwood's cone production, seed set, germination percentage, nursery growth and survival, rooting of juvenile cuttings, and their growth and survival in field plantings. Offspring resulting from self-pollinations were compared to offspring from controlled outcrosses of the same seed-parents. Milder forms of inbreeding, likely among offspring of wind-pollinated redwoods, are likely to exhibit performances intermediate between those of selfs and outcrosses. Effects of self-level inbreeding were found to be small or absent in favorable conditions, increasing to serious or disastrous when the redwoods encounter competition or are grown in unfavorable or stressful environments. Deploying clones of outcrossed offspring will eliminate the possibility of inbreeding-caused poor performance of planted redwoods.

Keywords: form traits, genetics, inbreeding depression, *Sequoia sempervirens*, tree improvement

Introduction

With a few important exceptions, survival, growth and other conifer traits are adversely affected by inbreeding. In the 1970s, we and others wondered whether coast redwood (*Sequoia sempervirens* (D. Don) Endl.), uniquely hexaploid among conifers, might be protected from the effects of inbreeding by heterozygosity fixed or homozygosity slowed due to having three diploid sets of chromosomes. We first observed and measured control-pollinated selfs and outcrosses of a single redwood family of unknown origin, also including open-pollinated sibs, planted in 1965 on a high-quality mid-elevation redwood site near the Mad River in Humboldt County, California.

Shortly after joining the Berkeley faculty, one of us (WJL) noted in 1963 an easy-to-climb unusually fecund redwood of unknown origin (tree 10) growing on the University of California's newly acquired Russell Research Station. Using techniques recently learned at the Institute of Forest Genetics (Placerville, California), control crosses were made using its own pollen and pollens of trees 12 and 19, two nearby planted redwoods, thus producing families 10×10, 10×12, and 10×19. Open-pollinated cones (10×Wind) were also harvested when the cones matured. Seeds from these four families were germinated and grown in a greenhouse in 1964.

In spring 1965, five seedlings of each of the two outcrosses, four of the wind-pollinated family, and 14 of the selfs were planted at 3-m square spacing on available land within a deer-fenced Simpson Timber Company research area above the Mad River in Humboldt County, at 380 m elevation. The inbreeding-trial plot was on a high-quality northeast-facing redwood site. Selfs alternated with the other entries, such that the nearest four neighbors of each plot-interior self were four of the other entries, and the nearest neighbors of each of the other plot-interior entries were four selfs. The site had been well-prepared and there was no early mortality.

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All 28 seedlings survived the first year, and only two 10×19 outcrosses and two selfs had died by August 1979. Only modest competition from weeds and brush had occurred between 1965 and 1979. Average heights recorded after the first growing season were: family 10×19 0.15 m; 10×Wind 0.13 m; 10×12 and 10×10 both 0.09 m. Differences in relative size increased during the next 14+ years. By August 1979, average heights were: 10×19 4.7 m; 10×12 4.1 m; 10×Wind 2.7 m; and 10×10 1.8 m. Bole diameters were measured at 2 dm above ground: 10×19 8.5 cm; 10×12 7.3 cm; 10×Wind 3.5 cm and 10×10 2.2 cm. These observations were consistent with increasing adverse effects of inbreeding on growth with time. Differences in average numbers of main stems were also noted in 1979: 10×19 1.3; 10×Wind 1.5; 10×12 1.7 and 10×10 2.3. Surprisingly, 10 of the 12 surviving selfs were forked (> one main stem) in 1979. The trial was discontinued after 1979, because the selfs were beginning to be strongly suppressed by their neighbors (Libby et al. 1981).

During 1977 to 1979, we conducted controlled crosses employing a total of eight additional parents from Humboldt County. They had been cloned as mature cuttings, which continued to produce female strobili and/or pollen, thus allowing the crosses to be performed soon on very small ramets of the clones. The crosses were made on small field-grown ramets of the parent clones at Russell Research Station in 1977, and in 1978 and 1979 on even smaller ramets of the parent clones growing in a greenhouse or lathhouse.

We followed cone development of bagged female strobili protected from pollen, pollinated with self pollen, or pollinated with outcross pollen from different males. There were no consistent differences in cone abortion between selfs and outcrosses. This was less surprising as we learned that the female strobili protected from pollination also developed into normal cones. They developed normal-appearing seeds at about the same rates as pollinated cones, but apparently contained no viable embryos, since none of the 12,000+ seeds from the unpollinated strobili of 10 different parents germinated. Differences among selfed, wind-pollinated and outcrossed families in seeds per cone were highly variable and overall not statistically significant.

One clone (RB37) appeared highly self-compatible, with unusually high germination rates of selfs. But in most other comparisons, the germination rates of the outcrossed seeds exceeded the germination rates of selfed seeds of the common parent, in some cases substantially so. Furthermore, the germination rates of wind-pollinated seeds were lowest, probably because of insufficient windborne pollen in the Russell research planting where the 1977 control-pollinations were done and the wind-pollinated cones collected (Libby et al. 1981).

The first nursery run (December 1977), including selfed, outcrossed and wind-pollinated seedlings, was soon heavily infected with a *Phytophthora* root pathogen, and nursery survival percentages were: 56 percent for outcrosses, 44 percent for wind-pollinated sibs, but only 14 percent for the selfs ($p < 0.001$). In the second (June 1978) and third (December 1978 to June 1979) nursery runs, the root disease was greatly reduced and no wind-pollinated families were included. Survival was about 81 percent in the second and 89 percent in the third, with differences between the selfs and outcrosses small and statistically non-significant.

Ten 25-tree blocks of plants were organized in leach-tube racks from the second nursery run: each rack had 10 seedling selfs, 10 related seedling outcrosses, one ramet each of four ‘standard clones’ (random clones of Humboldt-County origin used in most of our experiments), and one seedling from select family RB17×RB23 from the Simpson breeding program. Three of the selfed parents (S series) were from the same stand on the upper Mad River, and thus the three parents of those three outcross families (S6×S3, S3×S1, & S1×S6) were possibly related. A fourth outcross was SI×ARC154. The ARC154 was at that time Earth’s tallest known tree, located in Redwood National Park, and thus not likely related to S1. Parents of the fifth outcross, RB17 and RB23, are similarly from different mid-elevation Humboldt County stands, and thus not likely related. The average heights of these families and the standard clones following 1 year’s growth in a lathhouse were, in rank-order: RB17×RB23 13.9 cm; S1×ARC154 12.8 cm; S6×S3, S3×S1, and S1×S6 11.6 cm; standard clones 11.0 cm; S1, S3 and S6 selfs 8.5 cm; and ARC154 selfs 4.6 cm ($p < 0.001$). Given a hypothesis of some inbreeding depression, this rank-order is what one might expect.

Dead plants in the racks were replaced by spares of the same identity and approximate size in March 1979. Half of the 25-tree blocks were then planted on a deep alluvial flat next to the Mad River in Humboldt County, at elevation 185 m. The other half were planted on a relatively xeric side-ridge near Philo, Mendocino County, at elevation 489 m.

After 5 summer months at the favorable Humboldt County site, all five RB17×RB23 outcrosses survived, only one of three planted ARC154 selfs survived, and the rest survived similarly: S1, S3 and S6 selfs 89 percent; S1×ARC154 81 percent; standard clones 80 percent; and S6×S3, S3×S1 & S1×S6 79 percent. These differences in early survival were not statistically significant. Early survival was much lower at the more-stressful Mendocino County site. RB17×RB23 survived 80 percent and two of three planted ARC154 selfs also survived. The rest were clustered: S6×S3, S3×S1 & S1×S6 28 percent; S1×ARC154 27 percent; S1, S3 and S6 selfs 26 percent; and standard clones 25 percent. These differences also were not statistically significant. Because of the low survival, this Mendocino trial was discontinued.

Average 1979 height after the March through August first-season growth on the Humboldt site, in rank order, was: RB17×RB23 17.5 cm; S6×S3, S3×S1, and S1×S6 15.8 cm; standard clones 14.6 cm; S1×ARC154 14.2 cm; S1, S3 and S6 selfs 12.3 cm; and ARC154 selfs 12.0 cm. Although average August 1979 height was in a fairly small range, the number of surviving plants allowed the differences to be statistically significant at the $p < 0.01$ level and the rank-order was consistent with some adverse inbreeding effect. In the more-stressed Mendocino trial, average August height was: S6×S3, S3×S1 and S1×S6 22.8 cm; RB17×RB23 15.3 cm; S1×ARC154 13.2 cm; S1, S3 and S6 selfs 11.7 cm; standard clones 10.0 cm; and ARC154 selfs 3.0 cm. With fewer surviving plants but a greater range in growth, these differences were barely statistically significant at $p \sim 0.05$, and the rank-order was consistent with a strong inbreeding effect.

As of the late 1970s, that was about what we knew about inbreeding effects in coast redwoods. We then launched the next small study. The 1977 to 1979-produced families were germinated and then a small sample of each was clonally propagated by rooting cuttings (Kirchgessner and Libby 1985). Planning to put clonal replicates of this third study on more than one site, we first observed and recorded rooting performance of the then-available selfed and outcrossed families, with selfs of one or both of their outcrossed parents included in the study.

In late December 1980, 5 to 15 juvenile cuttings had been taken from seedlings of the selfed and outcrossed families. After auxin treatment of fresh-cut bases, they were set in leach supercells that had potting mix topped with about 5 cm of a complex organic rooting mix. This eliminated the need to transplant recently-rooted cuttings from rooting mix to growing substrate as the cuttings grew into field-ready plantable rooted cuttings (i.e., stecklings). The labeled tubes were randomly placed in supercell racks, which were then set into shallow water maintained at 23 ^{o}C to supply warm moist air to the tubes. Mist was delivered five or more times per day, and the entire setup was partially enclosed in clear plastic to maintain heat and humidity. A cutting was declared ‘rooted’ when one or more roots emerged from the bottom of its supercell, and it was then moved in its supercell to a lathhouse to grow to steckling status.

Observations were noted frequently, and data on rooting and cutting mortality were taken weekly. The first roots emerged from the bottoms of the supercells after about 10 weeks, and the data were analyzed in three periods: early (mid-March to mid-April); middle (mid-April to mid-May); and late (mid-May to late July). In the early period, there was no recorded mortality of either self or outcross cuttings. The original 496 self cuttings had rooted 11 percent and the original 517 outcross cuttings had rooted 26 percent ($p < 0.001$). There was minor cutting mortality during the middle period: selfs 3 percent; and outcrosses 1 percent. The 443 surviving self cuttings had rooted 36 percent and the 382 surviving outcross cuttings had rooted 44 percent. Neither the mortality nor rooting percentages in the middle period were significantly different.

During the late period, the warm high humidity conditions had favored an algae growth that increasingly covered the remaining cuttings, a development we judged to be stressful to the cuttings. During this late period, the 271 surviving self cuttings had rooted 18 percent and the 211 surviving outcross cuttings had rooted 13 percent, not a statistically significant difference. But the 271 surviving self cuttings had 18 percent mortality while the 211 surviving outcross cuttings had only 7 percent mortality ($p < 0.001$).

By mid-July, we had sufficient rooted cuttings for our planned field trials, so the rooting trial was discontinued. Overall during mid-March to mid-July, 53 percent of the self cuttings and 64 percent of the outcross cuttings had successfully rooted. The self cuttings had suffered 12 percent mortality and the outcross cuttings only had 3 percent mortality. Differences in both rooting and mortality were statistically significant at $p < 0.001$. In summary, self cuttings rooted more slowly than related outcross cuttings, and self cuttings suffered greater mortality in conditions judged stressful (Kirchgessner and Libby 1985).

We were then ready to begin the third study, a larger comparison of selfs and related outcrosses in field trials, reported below. The questions addressed in this study were: 1) Is coast redwood, being hexaploid, buffered from inbreeding effects by having three copies of its genome? 2) Given that redwood suffers from inbreeding, how severe are such effects, and in what conditions might they be important?

Materials and Methods

Four nearly-identical clonal replicates were planted on:

(a) a flat valley-bottom site on 8 September 1981 as a hedge-orchard at the Russell Research Station (Latitude 37° 54’ 59” N, Longitude 122° 09’ 12” W, elevation 240 m);

(b) Simpson Timber Company site on 15 December 1981 just east of Maple Creek Rd, south of Snow Camp Rd and near Ward Rd (Latitude 40° 50’ N, Longitude 123° 53’ W, elevation 950 m);

(c) a ridgetop Masonite site on 26 January 1982, just south of the Navarro Ridge Rd (Latitude 39° 11’ N, Longitude 123° 38’ W, elevation 260 m); and

(d) 4 June 1982 at a second Russell Research Station site, described more fully below.

At each location, the fundamental research units were 2×2 squares, with selfs on one diagonal and their outcrossed sibs on the other diagonal (fig. 1). For data-taking purposes, these were mapped in five 25-tree blocks, but there was no additional space between the blocks, with 2×2 squares and two self-outcross single-pairs at the right-hand edge of each block. Ideally, a four-tree square contained two full-sibs on one diagonal and a self of each of the full-sibs’ two parents on the other diagonal. However, the available selfs and their outcrossed relatives did not permit that in most cases, so the relevant comparison was often between a paired self and its related outcross. Five ‘standard clones’ were planted in the same order down the center of each block. Spacing was square, with 3 m between trees except in (a) the hedge-orchard, where spacing was 1 m between hedges. Note that the same number of selfs and outcrosses occur as border trees, but that only two standard-clone trees are in the border.

All self and outcross stecklings were sourced from the rooting trial reported in Kirchgessner and Libby (1985). Stecklings of the five standard clones were sourced from the ongoing production of those clones for use as shared entries in various trials. With few exceptions, the same clone was planted in the exact same mapped location in each of the four plantings. Substitutions were made in the field trials when there was no ramet of the designated clone of proper size and condition available at the time of planting. While these substitutions may weaken paired comparisons of a self and its related outcross, it little affected analyses comparing averages of all selfs, outcrosses, and standard clones in or among trials.

Page 1

S	O	S	O	S
O	S	O	S	O
C	C	C	C	C
S	O	S	O	S
O	S	O	S	O

Page 2

O	S	O	S	O
S	O	S	O	S
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S	O	S	O	S

Page 3

S	O	S	O	S
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Page 4

O	S	O	S	O
S	O	S	O	S
C	C	C	C	C
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S	O	S	O	S

Page 5

S	O	S	O	S
O	S	O	S	O
C	C	C	C	C
S	O	S	O	S
O	S	O	S	O

Figure 1—Planting design showing five map pages of the (actually contiguous) self (S), outcross (O), and standard-clone (C) trees. Dark borders emphasize the locations of the 2×2 and paired arrangement of related selfs and outcrosses.

In the earlier seedling trials reviewed in the Background above, the seedlings germinated at about the same times and grew similarly in the nursery. There was no attempt to closely match the size of paired selfs and outcrosses at the time of field planting, and thus the (usually small) nursery-size differences in the selfs and outcrosses were carried into the field trials. By contrast, nursery steckling sizes were strongly influenced by when the cutting rooted, and because the self cuttings generally had rooted later, this confounded inbreeding effects on growth with post-rooting growth time in the nursery.

After completing the planting of the Simpson field trial and measuring steckling heights immediately following planting, we noted that the average height of the selfs was only 79 percent the average height of the outcrosses. We therefore attempted to match the sizes of paired selfs and outcrosses at the times they were planted in the Masonite and Russell field trials. Although not all pair heights were thus made closely similar, on average pre-matching was successful. Average newly-planted heights of the self, standard-clone and outcross stecklings were within 1 cm of each other in the Masonite and Russell trials.

The Russell Research Station hedge-orchard still exists, but has not been used to establish additional trials. The Simpson and Masonite trials have both been discontinued (See Results, below). The Russell field trial remains useful and had 22+ years' growth on-site when the most recent data were taken.

The Russell Study Site and Conditions

The Russell trial was established at the University of California's 115 ha Russell Research Station on a small alluvial terrace with a slight-to-modest north-facing slope. The site is 14 km east of the San Francisco Bay, 35 km inland from the Pacific coast, and has a Mediterranean climate characterized by cool wet winters and hot dry summers. Mean annual rainfall of 497 mm falls mainly between the months of November and March and the average daily high temperature is 31 ^{o}C in mid-summer (www.weather.com). Several below-freezing days occur in most winters, and temperatures near or somewhat above 40 ^{o}C occur in most summers. Although 20 km north-northeast of the nearest native redwood population, the good growth of previously planted redwoods in Russell Research Station's valley bottoms gave us confidence that redwoods could thrive on this site.

This site was well-prepared prior to planting and well-maintained during the establishment years following planting. Drip irrigation was delivered to each plant in rainless periods during the first 2 years following planting, at which time the roots of most of the then-established redwoods had apparently reached the generally reliable ground water flowing downhill in that watershed in and under the alluvial-terrace soil. The newly-planted redwoods thrived in this combination of abundant-to-sufficient water, warm-to-hot temperature, and abundant sunlight during the growing seasons.

Survival and growth during the 2 establishment years, before irrigation was discontinued, were both good. Growth of the selfs and outcrosses during that establishment period were, on average, nearly identical and not statistically significantly different. One self, one standard-clone and four outcross plants died shortly after planting. They were replaced by somewhat larger ramets of the same clones on 5 December 1982. Subsequent growth bias due to those late replacements would have probably favored the selfs. One self, one standard clone and one outcross were not replaced when other instances of early mortality were replanted, reducing sample size to 49 selfs, 49 outcrosses, and 24 standard clones. In subsequent years, encroaching brush was unusual and periodically cut back. Soon after each tree exceeded 2 m height, it was pruned to half-height. This practice continued until all the trees in the trial exceeding 6 m height had been pruned of branches from their lower 3 m.

As anticipated, as the trial developed and crowns began to close, outcrosses were first commonly and then generally taller than their paired selfs. We have analyzed and now present the most-recent data taken, in August 2004, not only on survival and growth, but on several other traits we thought might be of interest. By 2004, it was apparent that the smaller sizes of the selfs was increasingly due to suppression by their neighboring generally-larger outcrosses, rather than due solely or largely to their inbreeding status. Ten years later, in all surviving self-outcross pairs, without exception the outcross was larger, generally substantially so.

Russell Trial Data and Analysis

The Russell trial was assessed in August 2004, 22+ years after planting. Bole diameter at 1.37 m height (DBH) and bole height were measured on live trees. Incidence of forking and ramicorn branching (unusually large steep-angled branch) were tallied for each tree (undesirable for timber production objectives). New branches initiating on the lower bole after pruning (i.e., epicormic sprouts) were counted and measured for length. The frequency scale was: 0 = no epicormic sprouts (desirable); 1 = 1-3 sprouts; 2 = 4-10; 3 = > 10 sprouts. The development scale for longest epicormic sprouts present was: 1 = < 0.2 m length; 2 = 0.2-1 m; 3 = > 1 m (most undesirable). Vertically-oriented basal sprouts were also measured, with data similarly categorized on frequency and development scales: 0 = no basal sprouts; 1 = 1-3; 2 = 4-10; 3 = > 10 sprouts; and for the one tallest sprout among those present: 1 = < 0.2 m height; 2 = 0.2-1 m; 3 = > 1 m height. Bole straightness and symmetry – with emphasis on quality of the lowermost (maximum 6 m long) log - was assessed on a five point scale: 1 = unusable; 2 = bad; 3 = some defects; 4 = good; 5 = excellent; with penalties for sweep, sinuosity, crook, elliptical cross-section, and lower-bole (especially lowermost log) deformities. Four aspects of branching were assessed: branch diameter, length, angle, and uniformity of branch distribution and size. Ocular estimates of the typical branch diameter relative to bole diameter at the point of knot formation were assigned relative scores on a 1 to 5 scale : 1 = large; 5 = small (desirable). Typical branch length compared to tree height was assessed on a 1 to 5 scale: 1 = long branches; 5 = short branches (with narrow crown considered desirable). Branch angle was assessed in terms of the typical departure from a horizontal 90 degree angle: 1 = relatively steep droop; 2 = mild droop; 3 = near 90-degrees (preferred); 4 = mild upward angle; 5 = steep upward angle. Branch uniformity in terms of distribution and size was assessed on a 1 to 5 scale: 1 = highly variable in size or distribution; 5 = uniform size and distribution (desirable). Live tree crowns were examined for presence of cones.

Data were summarized and tabulated for each trait assessed: giving number of then-surviving plants observed or measured, means, standard deviation, minimum, and maximum values. Binary survival data for selfs, outcrosses, and standard clones were compared in chi-square tests. Paired t-tests (two-tailed) were constructed to compare live-tree diameter and height in surviving pairs between the adjacent selfs and outcrosses with a common parent. Tree diameter and height data were also compared between all surviving selfs and outcrosses using two-tailed t-tests (not paired).

Characteristics such as forking and epicormic sprouting were not present on all trees, and were compared among selfs and outcrosses using the non-parametric analysis of variance procedure NPAR1WAY of SAS statistical analysis software (SAS Institute Inc. 1989) for Kruskal-Wallis rank tests of different median values. The non-parametric Kruskal-Wallis rank tests were also used to compare lower-bole form and branching scores among all live selfs, outcrosses, and standard clones. Wilcoxon signed-rank tests of two related samples were constructed to compare average live-tree (standing dead trees were not assessed) bole form and branching scores between adjacent selfs and outcrosses with common parents.

Results

The Simpson and Masonite Trials

At the Simpson site, stumps of its recently-logged redwoods indicated that it was a suitable site for redwoods, and the site was well-prepared. But something apparently went wrong following the December 1981 planting of our trial stecklings. It was most likely aggressive regrowth of competing grasses and shrubs, or perhaps some other unobserved stressful events occurred. By March 1983, 64 percent mortality had already occurred. The expected pattern under stress was recorded: the outcrosses suffered 55 percent mortality, the standard clones 61 percent, and 75 percent of the selfs had died. In the following 2 years, an additional 23 percent of the trees surviving in March 1983 had died. The trial was then discontinued.

The Masonite trial, planted in January 1982, at first fared better. It was located on a similarly well-prepared redwood site. By June 1983, only 15 percent of the stecklings had died. Among these, the outcrosses had 10 percent mortality, the standard clones 16 percent and the selfs 20 percent. Prior to our next site visit, an ownership change had occurred, with replacement of the forestry staff. The new staff, unaware of the trial, located a skid road and landing on the trial. Only 33 of the original 124 planted stecklings remained alive around the edges of the skid road and landing, and some of those were damaged. The trial was then discontinued.

The Russell Trial

Following some initial post-planting mortality, soon replanted with larger ramets of the same clones, all stecklings survived the first 2 years, during which drip irrigation was supplied to each plant during rain-free periods. After irrigation was discontinued, the 49 outcross and 24 standard-clone stecklings survived similarly during 1984 to 2004, with mortality of 21 percent among standard clones and 27 percent mortality among outcrosses. The 49 self stecklings suffered much greater mortality of 61 percent. A chi-square test indicated that survival was significantly lower among the selfs than among the outcrosses ($p < 0.001$).

Trees in the southern six columns (map page 5 and adjacent column of page 4, fig.1) grew on a drier harsher region within the Russell trial, those conditions being apparent by a change in vegetation and slope. As of 2014, all 12 of those selfs had died, while four of those six standard clones and eight of those 12 outcrosses still survived, appeared healthy, and were growing reasonably well.

The summary data in table 1 are for all surviving self, outcross and standard-clone stecklings. By 2004 the outcrosses were on average 19 percent and 14 percent larger than the standard clones in DBH and height, respectively. By contrast, the outcrosses were on average 79 percent and 38 percent larger than the selfs in DBH and height. In the more sensitive analyses of self-outcross differences between live selfs paired with live outcrosses sharing a common parent (n = 16 data pairs), paired t-tests detected significantly lower diameters ($p < 0.001$) and heights ($p < 0.001$) of the selfs (figs. 2, 3). Higher coefficients of variation (s.d./mean) indicated less stable performance of selfs as a group.

Figure 2—RB22×RB37 outcross (left) vs. RB22 self (right), with age-22 DBH and height of 40.9 cm vs. 13 cm, and 24 m vs. 14.1 m, respectively, at Russell Research Station, Lafayette, California.

Figure 3—RB17×RB37 outcross (left) vs. RB17 self (right), with age-22 DBH and height of 69.0 cm vs. 20.5 cm, and 26.3 m vs. 10.9 m, respectively, at Russell Research Station, Lafayette, California.

Table 1—Summary data for DBH (diameter at breast height; 1.37 m) and total height among selfs, standard clones and outcrosses in the Russell Research Station field trial after 22+ years growth

Trait	Group	n	Mean	s.d.	min.	max.
Diameter (cm)	Selfs	19	23.7	10.2	5.4	42.3
	Outcrosses	36	42.4	9.1	21.2	69.0
	Std. clones	19	35.6	9.1	10.1	47.6
Height (m)	Selfs	19	15.9	5.7	2.0	23.6
	Outcrosses	36	21.9	3.0	13.5	27.8
	Std. clones	19	19.2	3.2	9.7	22.5

Mortality between the time of planting and assessment had reduced the number of adjacent pairs of related selfs and outcrosses available for assessment. However, results from analysis of all live selfs and outcrosses (n = 55) supported results of the paired t-tests (n = 16 pairs). Pairing the same self with two related adjacent outcrosses (and vice versa) gave 26 pairings of selfs and outcrosses including duplicate data for 10 trees used twice (in double pairings). Paired t-test analysis of these data also gave similar results: substantially lower diameter and height growth among selfs than among their related outcrosses over 22 years since outplanting.

Outcrossed progeny exhibited significantly greater frequency of forking ($p = 0.002$) and occurrence of ramicorn branches ($p < 0.05$) than selfs (table 2). Only one self had a fork and only one had a ramicorn branch. Epicormic sprouts were present on 75 percent of selfs, and on average these sprouts were larger and more prolific than sprouts on outcrosses or standard clones, but differences in epicormic sprout frequency and size were not statistically significant ($p = 0.28$ and $p = 0.12$, respectively). Basal sprouts were less frequent and smaller on selfs, on average, however these differences were not significant ($p = 0.22$ and $p = 0.34$, respectively). Cones were only detected on one outcross tree and two standard-clone trees. Comparisons between the outcrosses and standard clones, although generally not statistically significant, were consistent with the standard clones being at a less-juvenile maturation state than the outcross trees.

Table 2—Summary data for count data on presence of forking and ramicorn branching, and scores for frequency and size of epicormic and basal sprouting among selfs, outcrosses, and standard clones in the Russell Research Station field trial

		n[b]	Count[a]	Mean[b]	s.d.[b]	min.[a]	max.[a]
No. forks	Selfs	19	1	0.05	0.23	1	1
	Outcrosses	36	17	0.69	0.89	1	3
	Std. clones	19	3	0.21	0.54	1	2
No. ramicorn	Selfs	19	1	0.16	0.69	3	3
	Outcrosses	36	11	0.53	0.91	1	3
	Std. clones	19	2	0.16	0.50	1	2
No. epicormics	Selfs	19	15	1.68	1.25	1	3
	Outcrosses	36	25	1.33	1.10	1	3
	Std. clones	19	12	1.16	1.12	1	3
Size of epicormics	Selfs	19	15	1.53	1.12	1	3
	Outcrosses	36	25	1.06	0.98	1	3
	Std. clones	19	12	0.89	0.94	1	3
No. basal sprouts	Selfs	19	3	0.21	0.54	1	2
	Outcrosses	36	11	0.47	0.81	1	3
	Std. clones	19	4	0.42	0.90	1	3
Basal sprout size	Selfs	19	3	0.47	1.12	3	3
	Outcrosses	36	11	0.67	1.12	1	3
	Std. clones	19	4	0.53	1.12	1	3

[a] Summary data for trees with forks, ramicorns, epicormics, or basal sprouts only.
[b] Summary data for all live trees, i.e., with/without forks, ramicorns, epicormics, or basal sprouts.

In terms of form characteristics (table 3), non-parametric Kruskal-Wallis tests indicated that branches on average were more horizontal ($p < 0.0001$) and evenly distributed and of uniform size among the standard clones ($p < 0.05$) than in the selfs or outcrosses whose branching attributes were not significantly different from one another. No other statistically significant differences were detected among these three groups. Among pairs of selfs and outcrosses sharing a common parent (n = 16 pairs), Wilcoxon signed-rank tests indicated that average branch diameter was marginally larger among outcrossed ramets ($p = 0.06$). No significant differences were detected between the selfs and outcrossed progeny in terms of bole straightness and symmetry or branching traits.

Table 3—Summary data for redwood bole straightness and symmetry (Bole SS), and branching characteristics: branch diameter, length, angle, and uniformity of branch distribution and size among selfs, outcrosses, and standard clones in the Russell Research Station field trial

		n	Mean	s.d.	min.	max.
Bole SS	Selfs	19	2.8	0.9	1.0	4.0
	Outcrosses	36	3.0	1.0	1.0	5.0
	Std. clones	19	2.9	0.8	2.0	4.0
Branch diameter	Selfs	19	2.6	1.3	1.0	5.0
	Outcrosses	36	3.2	1.1	1.0	5.0
	Std. clones	19	3.2	1.1	1.0	5.0
Branch length	Selfs	19	2.7	1.3	1.0	5.0
	Outcrosses	36	2.9	1.1	1.0	5.0
	Std. clones	19	3.0	1.6	1.0	5.0
Branch angle	Selfs	19	2.3	0.7	1.0	4.0
	Outcrosses	36	2.5	0.8	1.0	4.0
	Std. clones	19	3.2	0.5	2.0	4.0
Branch uniformity[a]	Selfs	19	3.0	1.3	1.0	5.0
	Outcrosses	36	3.0	1.4	1.0	5.0
	Std. clones	19	4.1	0.9	2.0	5.0

[a] Branch uniformity in terms of distribution and size, assessed on a 1 to 5 scale: 1 = highly variable in size or distribution; 5 = uniform size and distribution (desirable).

Discussion

Inbreeding can occur at various levels, and inbreeding depression manifests proportionally to the level of inbreeding. For simplicity, both in the creation of inbreds and in the interpretation of inbreeding depression as causing observed differences, we chose an inbreeding level of 50 percent, associated with selfing of non-inbred parents. We employed selfs of seven different parent redwoods in the third set of inbreeding trials. Three parents, S1, S3 and S6, were from an earlier study investigating the distribution of maturation state in large redwoods. The S-series parents had been left as seed-trees in a redwood harvest at mid-elevation near the Mad River in Humboldt County. Parents RB17, RB22 and RB37, were selections from Simpson's breeding-zone B, in mid-elevations of second-growth redwood stands in Humboldt County. The seventh selfed parent, ARC154, was from low elevation next to Redwood Creek in what is now Redwood National Park in Humboldt County. The RB-series selections were made as the best-formed large tree in two hectares, and each came from a different area within the 'B' breeding zone. While not verified by heterozygosity analyses of DNA or allozymes, it seemed likely that none of these seven parents were inbred, and thus their selfs would average 50 percent inbred.

We used some of those same seven parents as pollen-parents in crosses to produce full-sib families. We also used pollen from an additional four parents from the RB selections (RB2, RB5, RB9 and RB23) to produce additional full-sib families related to the seven parents we selfed. Since none of the RB-series selections were from the same stand, it is unlikely they were related to each other, or to ARC154 or the S-series parents. Thus, the full-sib outcross families from crosses involving RB or ARC154 pollens were likely to have zero or near-zero percent inbreeding. However, S1, S3 and S6 were from the same stand and might have been related at the sib or cousin level. Thus full-sib families, S1×S3, S1×S6 and S6×S3, might have had some degree of inbreeding, possibly as high as 25 percent if the parents were full-sibs. We saw no evidence of this. Their performance in terms of average DBH and height (41.3 cm; 22.2 m; n = 5) was similar to the average for all other surviving outcrosses (42.5 cm; 21.8 m; n = 31).

The five standard clones had been chosen much earlier, originated as five random seedlings from a production run of redwood planting stock in Simpson's Korbel nursery. Being from wind-pollinated cones collected in natural stands, which mostly have some family structure, it is possible, even likely, that some or all of these five standard clones were inbred to some currently-unknown degree. Their performance in these trials lends some support to that idea, but the fact they were mostly interior trees also leads to the expectation that their average growth would be less than the outcrosses, which occupied edge positions more than half the time (fig.1).

In nature, or in redwood plantations employing wind-pollinated seedlings, the expectation is that trees with modest-to-strong inbreeding would lose out in competition as the stand develops. That might even be a good thing in plantations in need of thinning, or in overstocked natural regeneration. But one can have too much of a good thing, and most silviculturists likely prefer to manage stands or plantations not suffering from inbreeding depression.

At Russell Research Station, the incidence of forking and ramicorn branching was infrequent and highly variable, but lower among selfs on average (table 2). However, we could not separate spontaneous forking and ramicorn branch incidence from occurrences incited by prior woodrat damage. Woodrats may have favored larger trees, and thus had less impact on selfs that were generally smaller than outcrosses and standard clones. In the large 200 clone Kuser trial, woodrats caused upper bole girdling in a majority of the trees. We followed recovery from that damage. Some formed basket whorls, some forked, and some grew a single sprout from below the girdle, thus keeping a single bole. It is a valid question whether ability to recover from girdling is different between selfs and outcrosses. While it would be nice to know, our small experiment and our available data did not allow us to determine cause. Future studies, if performed, should include additional replicates, preferably on high-quality redwood sites within the native range, to better detect differences in traits occurring infrequently, or in highly variable traits such as epicormic sprouting.

Conclusions

Compared to other conifers, coast redwood has many outstanding features, but hexaploid protection from inbreeding depression is not one of them. Although cone set and seeds per cone were little affected by pollinations that produce inbred offspring, from germination on, inbred redwoods survived at lower rates and grew more slowly than non-inbred redwoods, particularly on harsh sites or as competition or other stresses occurred or intensified.

An earlier reported observation that inbred redwoods had a higher frequency of forking was not supported by this trial. It was observed, however, that pruned inbreds had more and larger epicormic sprouts than did related outcrosses or random standard clones. If this observation is sustained, it could be important if pruned redwoods are grown to produce knot-free outerwood. Inbreds had fewer and smaller basal sprouts than did outcrosses, possibly an effect of the inbreds' smaller size and lower vigor. Other redwood traits of interest, namely bole form and several branch characteristics, seemed little affected by inbreeding.

As redwood silviculture advances, it seems likely that few serious growers will purposely plant inbred redwoods, except possibly for research purposes. Increasingly, clones from outstanding and apparently adapted native trees will be used, as will increasingly tested and characterized outcrossed clones from redwood breeding programs.

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Variation in Genetic Structure and Gene Flow Across the Range of *Sequoiadendron giganteum* (giant sequoia)[1]

Rainbow DeSilva[2] and Richard S. Dodd[2]

During this century, climate warming and altered precipitation patterns will lead to habitat changes that may be beneficial to some long-lived tree species and detrimental to others. Paleoendemics, with limited and disjunct distributions will face the greatest challenges, as migration rates will be too slow to keep pace with rapid environmental change and populations at the receding edges are eroded through mal-adaption. Giant sequoia (*Sequoiadendron giganteum* (Lindl.) Buchholz) is an iconic Sierra Nevada tree species with populations that tend to be small and highly fragmented (particularly in the northern range), making them especially vulnerable to environmental change. Maintenance of genetic variation is an important determinant of population persistence that, in part, depends on gene flow within and between populations. The research presented here describes: 1) the distribution of genetic diversity among population pairs distributed across the range of giant sequoia, and 2) the effective rates of gene flow across a highly fragmented habitat.

In 2015 and 2016, DNA was extracted from foliage collected from eight groves distributed across the range of giant sequoia, with the exception of Mariposa grove (leaf tissue from this grove originated from a clonal orchard at the University of California's Russell Research Station). For Placer rove, DNA was obtained from foliage of all trees and from seeds after germination. We used 11 microsatellite loci (DeSilva and Dodd 2014) to compare allelic diversity between population pairs, using the software STRUCTURE (Pritchard et al. 2000). We also used STRUCTURE to infer levels of historic gene-flow through assessing the degree of admixture between spatially separated population pairs. We inferred dispersal distances and size of effective mating groups within groves using SPAGeDI (Hardy and Vekemans 2002).

We found a prominent north-south divide at the Kings Canyon in spatial connectivity across *S. giganteum* groves. For the northern populations pairs, assignments of individuals to genetic clusters closely follow the grove of origin, indicating strong genetic structure (fig. 1). Rates of admixture (individuals showing evidence of shared genotypes from the two clusters) were low. This is surprising given the close proximity of members of each pair (i.e., ~4 km between Calaveras groves, ~3 km between Merced and Tuolumne , and ~7 km between Nelder and Mariposa groves. This indicates that even over short distances, groves in the northern range act as independent genetic units, with limited inter-grove gene flow. This is in contrast to populations in the southern part of *S. giganteum* range. For example, Long Meadow and Freeman Creek exhibited weak genetic and high rates of admixture (all individuals show equal ancestry from both assigned clusters) (fig. 1). These data indicate that these two southern groves exhibit little or no genetic divergence indicating recent or ongoing gene flow. This finding confirms the pattern noted in a range-wide study (Dodd and DeSilva 2016) that southern populations form a continual genetic unit. Given the distance between these two groves (about 20 km), we believe that genetic connectivity in the southern S. *giganteum* range may be maintained through a stepwise model of gene-flow.

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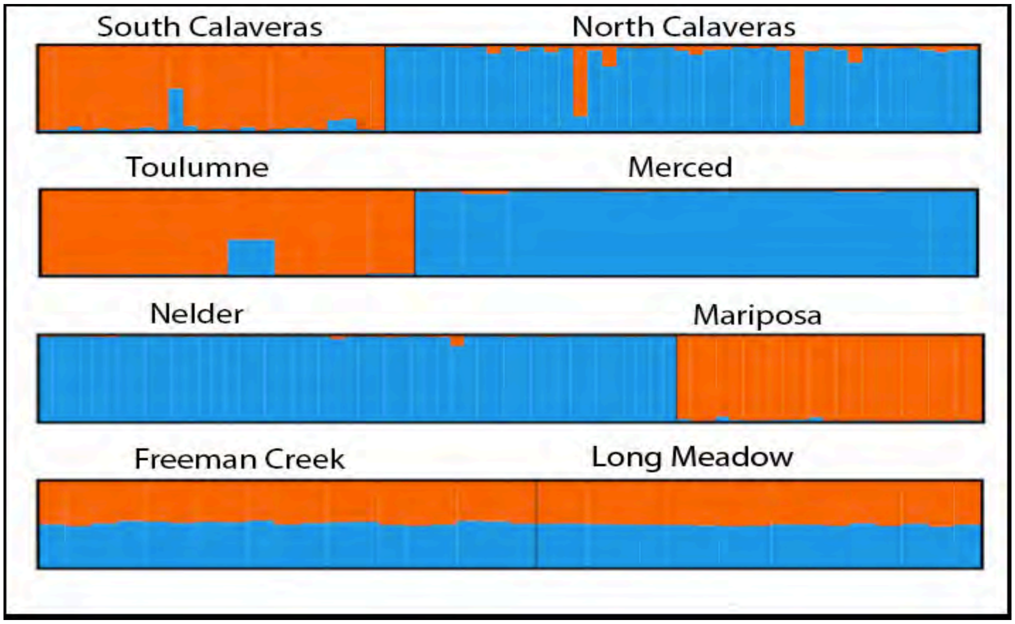

Figure 1—STRUCTURE (Pritchard et al. 2000) clusters for population pairs: Vertical bars represent a sampled individual, color-coded for assigned cluster. Northern range of giant sequoia represented by: North–South Calaveras, Tuolumne–Merced, and Nelder–Mariposa. Southern range: Freeman Creek–Long Meadow.

We estimated the fine-scale genetic parameters of neighborhood size (the number of effectively reproducing individuals occupying a deme around a parent tree) and sigma (the bulk dispersal distance of genes). We found dispersal dynamics to be highly variable across populations studied so far (table 1). In North Calaveras, gamete dispersal (sigma) likely occurs over a spatial scale of (0.22 km) and mating groups are not dramatically smaller than the mature census population. For Nelder, average dispersal distance is 0.29 km; however, the neighborhood only contains 10 individuals. We believe this difference is due to the variation in density across these two groves and the fact that Nelder grove is spread over a large area, with many areas where giant sequoia is absent. Thus, Nelder exhibits increased grove edge and spatial separation of demes. In contrast, within Giant Forest, a large grove in the southern section of *S. giganteum* range, the genetic neighborhoods are large in both area and number of individuals. The larger dispersal distance found in Giant Forest can be accounted for, in part, by the larger grove area. Larger groves contain more overlapping genetic neighborhoods and less edge, which allows gene flow to occur in all directions.

Table 1—Neighborhood size for three populations distributed across the range of giant sequoia

Population	Number of individuals within neighborhood	Radius (km) of neighborhood (sigma)
North Calaveras	51	0.22
Nelder	10	0.29
Giant Forest	174	1.6

Placer is the northernmost *S. giganteum* grove, and is separated from the nearest natural grove at North Calaveras by approximately 90 km. There are six mature individuals in this grove and two young individuals, which we determined by genetic analysis to be a result of human planting, rather than natural reproduction. Placer grove exhibited extremely low genetic diversity; of the 11 loci, five were monomorphic and three others had only two alleles each. Interestingly, we found evidence of an influx of exogenous pollen at Placer; preliminary results from seed arrays showed some seeds that were heterozygous with an allele that did not occur in any of the mature trees in the Placer grove. This cannot occur without the introduction of an outside pollen source. Considering that the nearest grove to Placer is ~90 km to the south, we determined that the likely source for this exogenous pollen is either one of the two young individuals (within the grove of planted origin) or the nearby plantation of *S. giganteum* (planted by the Auburn Lions Club in 1951 approximately 15.2 m to 30.5 m from the grove edge).

In conclusion, our research suggests that the northern populations of S. *giganteum* should be considered a high conservation priority, due to their small size, unique genetic composition, and lack of genetic continuity with other groves. In contrast, southern S. *giganteum* groves will likely be more resilient to genetic diversity loss due to genetic exchange with adjacent groves. Our previous work also indicates that giant sequoia is suffering from a long-term decline in population size and exhibits low levels of genetic diversity (Dodd and DeSilva 2016), making it crucial to protect the remaining diversity within the species. The fine-scale spatial structure analyses presented here can be used to direct conservation efforts. For instance, groves like Nelder that have strong within-population structure, are highly vulnerable to loss of genetic diversity. Moreover, incorporating genetic neighborhood size into ongoing seed collection strategies can maximize the capture of genetic diversity across the range. With the onset of climate change, genetically diverse seed banks may become invaluable if and when land managers begin to undertake widespread S. *giganteum* planting or assisted migration.

Acknowledgments

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SESSION 5 – Silviculture

A Comparison of Stand Structure and Composition Following Selective-Harvest at Byrne-Milliron Forest[1]

Amy K. Petersen[2] and Will Russell[2]

Abstract

The effects of selective-harvest on forest composition and structure in the southern range of the coast redwood (*Sequoia sempervirens* (D. Don) Endl.) forest have not been well documented. This case study focused on the Byrne-Milliron Forest in Santa Cruz County, California where selective-harvest is currently the primary method of timber extraction. The purpose of this research was to determine how forest structure and composition varied in regard to harvest intensity and management goals. We sampled 100 plots in the Byrne-Milliron Forest across five harvest sites. All sites had been essentially clear-cut in the late 19th or early 20th century, and subsequently selectively harvested in the late 20th and early 21st century. Four of the five sites have been managed primarily for timber production, while the fifth site, the Late Successional Unit (LSU), has been managed for old-forest conditions as well as timber production. We predicted the LSU would contain more late seral features, and that the presence of these features would be positively correlated to years since harvest, and negatively correlated to percentage cut and number of harvest re-entries. Data analysis procedures included one-way analysis of variance (ANOVA) for comparison between sites, and Pearson product-moment coefficient for correlations between variables. As expected, the LSU exhibited the most developed old-forest features, including the lowest stand density and exotic species richness among all sites evaluated. In addition, it contained the highest percentage of coast redwood associated herbaceous species and large woody debris (LWD). Results also indicated that percentage cut was the strongest predictor for canopy cover, stand density, LWD, and the cover of coast redwood associated herbaceous species. Our findings suggest that a lower percentage cut is more effective in maintaining conditions commonly associated with late seral forests such as snags, fire hollows, complex canopy structures and LWD, and these features can be present in selectively harvested stands if carefully managed.

Keywords: coast redwood, selective-harvest, *Sequoia sempervirens,* understory associates

Introduction

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). In coast redwood (*Sequoia sempervirens* (D. Don) Endl.) forests, higher yield harvests do not mimic this model. Previous studies have indicated that intensive harvesting of this forest type contributes to crown dieback (Russell et al. 2000), loss of habitat (Carter and Erickson 1992), loss of understory associated species (Jules 1998, Kahmen and Jules 2005, Reader 1987, Rivas-Ederer and Kjeldsen 1998) and an increase in exotic understory flora (Russell 2009).

In the Byrne-Milliron forest in Santa Cruz County, California, low intensity harvests were employed in some areas in an attempt to allow the continuity of late seral forest characteristics. These features, including snags, fire hollows, and complex canopy structures create habitat essential to a variety of wildlife species (Hunter et al. 1995, Overtree and Kitayama 2013[3], Ralph and Miller 1995).

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[2] Department of Environmental Studies, San Jose State University, San Jose, CA 95192.

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Large woody debris (LWD), which accumulates over time in late seral stands, provide habitat and also act as a stream buffer (Franklin et al. 1981, Lienkaemper and Swanson 1987). Understory floristic species associated with this forest type may take decades to recover following an intensive harvest event, and therefore a lighter cut is more beneficial in maintaining species diversity (Hageseth 2008, Halpern and Spies 1995).

Small group or selective harvest mimics the small patches created by tree fall gaps in late-seral coast redwood forests. Employing this method with regard to the aforementioned late seral forest characteristics enables forest managers to maintain a healthy forest ecosystem while actively managing for timber.

Methods

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 to 150 cm and morning fog is common during the summer months. Common coast redwood associate hardwoods in this region include California hazelnut (*Corylus cornuta* var. *californica* (A. DC.) Sharp), boxelder (*Acer negundo* L.), California bay (*Umbellularia californica* (Hook. & Arn.) Nutt.), Pacific madrone (*Arbutus menziesii* Pursh), big leaf maple (*Acer macrophyllum* Pursh), coast live oak (*Quercus agrifolia* Nee) and interior live oak (*Quercus wislizeni* A. DC.) (Cooney-Lazaneo and Lyons 1981, Lyons and Cuneo-Lazaneo 1988). Common coniferous associates include California nutmeg (*Torreya californica* Torr.), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and knobcone pine (*Pinus attenuata* Lemm.).

Selective-harvest has been the required method of timber extraction since clear-cutting was banned in 1970. This was in adherence to regulations applicable to the Southern Subdistrict under the California Forest Practice Rules (CAL FIRE 2016). The coast redwood forests in this region are predominately second-growth with sparse patches of old-growth occurring intermittently throughout the county.

Our research was conducted in five distinct harvest areas within the Byrne-Milliron Forest, located 8 km (5 miles) north of Corralitos. This region is composed of coast redwood, mixed chaparral, montane hardwood, coastal scrub and eucalyptus (CAL FIRE 2001). Clear-cut in the late 1880s, the property consists predominately of even-aged coast redwood and Douglas-fir. Byrne-Milliron spans 163 ha with elevations ranging between 125 to 500 m and predominately west facing slopes (see footnote 3). Physiographic characteristics including elevation and slope are similar between sites (table 1). 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. 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 (CAL FIRE 2001). Several small tributaries on the property lead to the nearby Browns Creek, although only one runs continuously throughout the year.

Table 1—Physiographic characteristics of five treatments in the Byrne-Milliron Forest in Santa Cruz, California (dates listed pertain to the years that harvest occurred)

	1987/2004		1990/2007		1996		2001		2007	
	Mean	*S.E.*	*Mean*	*S.E.*	*Mean*	*S.E.*	*Mean*	*S.E.*	*Mean*	*S.E.*
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

The Land Trust of Santa Cruz County purchased the original 130 ha Byrne Forest in 1984. The Milliron property, which added an additional 32 ha, was purchased in 2008 (see footnote 3). For the purpose of this study, the Milliron property has been excluded. There have been a total of seven selective harvests since the Land Trust took ownership. The property is now broken up into several management units (fig. 1) including the Central Unit (56 ha), the Early Successional Unit (22 ha), the Late Successional Unit (LSU) (15 ha), and the Southern Unit (24 ha). Harvesting has been conducted at a variety of intensities using tractor, skyline cable, or both methods combined (table 2).

The management goals for the units were similar, with the exception of the LSU (CAL FIRE 2001). Goals for the Central and Southern Units included implementation of an uneven-aged stand structure, which promoted new growth and provided space to aid the residual stand. Other goals included maintaining current species composition, with special consideration to coast redwood, as well as creating and maintaining structural features beneficial to wildlife. Co-dominant tree species were removed as needed to promote an increase of coast redwood size classes. Pruning of sprout clumps and reseeding where necessary were also objectives. Although the LSU had similar management goals, the main focus was the creation and preservation of late successional features including a complex canopy, shady microclimate, and structural elements such as snags and LWD. Minimal logging and an overall reduction in disturbance were promoted to conserve aesthetic value and support the desired habitat features.

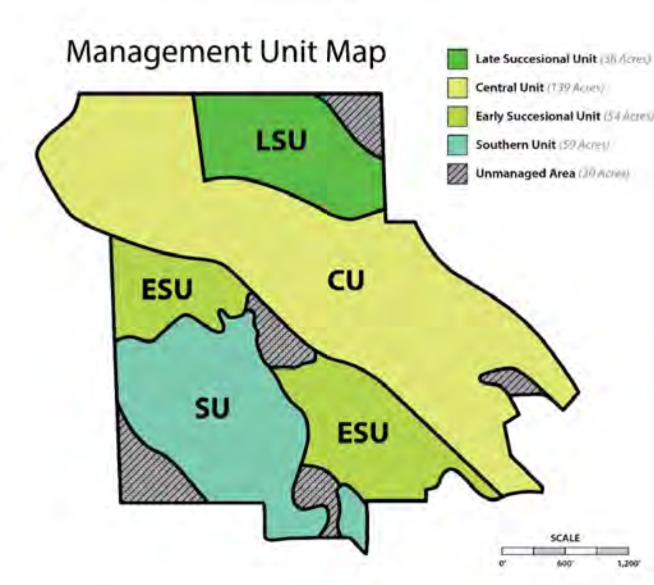

Figure 1—Management units at Byrne-Milliron Forest. (Image by Derrick Wynes)

Table 2—Harvest data for five treatments in the Byrne-Milliron Forest in Santa Cruz, California

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

[a.] CU = Central Unit, SU = Southern Unit, LSU = Late Successional Unit.
[b] T = tractor, S = skyline cable.
[c] Percentage cut reflects the most recent harvest re-entry.

Five of the selective-harvests 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 2001 (see footnote 3). All subsequent harvests 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 (CAL FIRE 2001). Other selective harvests completed include one in the Southern Unit, which was entered in 1996, as well as one harvest in the LSU, entered in 2007 for the first time since the clear-cut.

Plot 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. Twenty 0.032 ha (20 m diameter) circular plots were randomly selected within each of the five sites, for a total of 100 plots. 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 quadrants to determine relative herbaceous cover.

Data was collected between May 2012 and July 2013. Physiographic variables recorded at each plot included slope, aspect and elevation. All tree species > 10 cm were measured using a diameter at breast height (DBH; 1.37 m) tape. Canopy cover was determined using a convex spherical densiometer, with readings taken in each of the four cardinal directions from plot center. The LWD was also counted and circumference and length were recorded for each occurrence.

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 percentage cut per hectare) and dependent variables (aspect, elevation, slope, canopy cover, stand density, size classes of coast redwood, LWD density, coast redwood associated species, native species richness, exotic species richness, and relative dominance of coast redwood, which was calculated using the basal area). Size class tallies were conducted using the follow classes: < 10 cm, 10 to 24 cm, 25 to 49 cm, 50 to 99 cm, 100 to 149 cm, 150 to199 cm and > 200 cm DBH (Giusti 2007). Herbaceous species present throughout each plot were identified and visual cover 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 (Baldwin et al. 2012). In addition, coast redwood associated herbaceous species were selected for further analysis. These included hooker's fairy bells (*Prosartes hookeri*), modesty (*Whipplea modesta*), Pacific starflower (*Lysimachia latifolia*), Pacific trillium (*Trillium ovatum*), redwood violet (*Viola sempervirens*), and redwood sorrel (*Oxalis oregana*). One-way analysis of variance (ANOVA) with Bonferroni post hoc analysis was used to analyze differences between treatments. All statistical analyses were conducted using IBM SPSS Statistics 21 and Microsoft Office Excel 2013.

Results

Forest structure and composition varied significantly between treatments, with differences found in canopy cover, stand density, density of LWD, coast redwood associated herbaceous species, and several other metrics (table 3).

Table 3—Results for structural and compositional variables on five treatments in the Byrne-Milliron Forest in Santa Cruz, California

	Years since harvest		Number of re-entries		Percentage cut	
	p-value	*r^2 value*	*p-value*	*r^2 value*	*p-value*	*r^2 value*
Stand density (all species)	0.28	0.052	0.001	0.15	0.001	0.27
Coast redwood density	0.8	-0.00036	0.007	0.1	0.02	0.26
Tanoak density	0.05	0.15	0.32	0.13	0.76	-0.007
Redwood basal area	0.91	-0.036	0.44	-0.04	0.76	0.1
Coast redwood dominance	0.18	-0.097	0.61	-0.13	0.3	0.08
LWD density	0.004	-0.32	0.99	0.052	<0.001	-0.36
Coast redwood associates	0.014	-0.25	0.68	-0.04	<0.001	-0.47
Native species richness	0.86	-0.13	0.32	0.1	0.052	0.21
Exotic species richness	0.82	-0.04	0.1	0.26	0.013	0.27
Canopy cover	<0.001	-0.04	0.2	0.06	<0.001	-0.04

Stand Density

Stand density varied between the five treatments (table 4). The 1990/2007 harvest site had the highest density of all sites, followed by the 1987/2004 site, the 1996 site, the 2001 site, and the LSU (fig. 3). Pearson's product-moment correlation coefficient indicated that there was a positive relationship between stand density and number of re-entries (r = 0.15) as well as percent cut (r = 0.27). Variation between sites in regard to individual tree species density was particularly pronounced for redwood and tanoak (table 4). These species had a much lower density in the LSU compared to the other four sites. Variation for other tree species was all within the margin of error.

Table 4—Comparison of mean stand density (trees/ha) on five treatments in the Byrne-Milliron Forest in Santa Cruz, California

	1987/2004		1990/2007		1996		2001		LSU	
Stand density	*Mean*	*S.E.*	*Mean*	*S.E.*	*Mean*	*S.E.*	*Mean*	*S.E.*	*Mean*	*S.E.*
All tree species	11.90	0.96	12.85	1.41	11.15	0.98	9.15	1.02	7.05	0.68
Coast redwood	10.10	0.81	11.25	1.24	9.10	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	0.60	0.50	0.10	0.07	0.30	0.15	3.10	1.44	1.25	0.95
Oak	2.35	1.25	4.60	1.22	5.40	3.12	0.55	0.26	0.65	0.25
Douglas-fir	0.40	0.28	0.40	0.28	0	0	0.60	0.29	0	0
Big-leaf maple	0.05	0.05	0.05	0.05	0.40	0.35	0	0	0.35	0.30

Size class distribution of coast redwood also varied between treatments (fig. 2). Overall, the LSU had the most even distribution as well as the lowest stand density with the fewest stems < 49 cm and the highest density of stems 150 to 199 cm. Sites entered twice, including the 1987/2004 and 1990/2007 harvest areas had the highest number of small stems and the most skewed ratio of small to large trees. There was a significant correlation between stems < 10 cm and years since harvest (p = 0.001, r = 0.18) as well as percentage cut per hectare (p = 0.004, r = -0.05) (table 5). When analyzed with number of harvest re-entries, stems 10 to 24 cm were significantly lower in sites with one re-entry compared with two re-entries (p = < 0.001, r = 0.22). In relation to percentage cut, the LSU had significantly fewer stems 10 to 24 cm in comparison with all other sites (p = < 0.001, r = 0.36). In addition, the LSU had a significantly larger amount of stems 150 to 199 cm in comparison with all other sites analyzed (p = 0.001, 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.

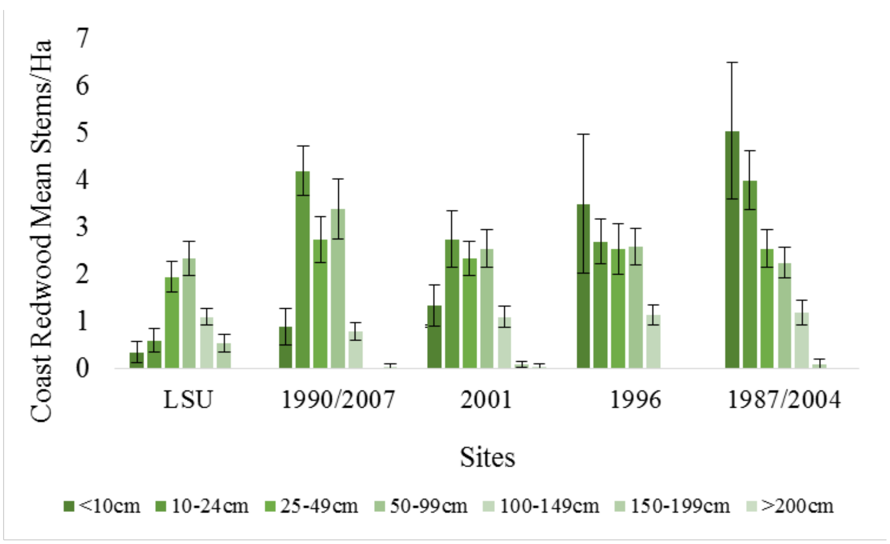

Figure 2—Coast redwood mean size class distribution on five treatments in the Byrne-Milliron Forest in Santa Cruz, California (n = 20 plots per treatment).

Coast Redwood Basal Area and Dominance

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 LSU (table 6). Relative dominance of coast redwood was highest in the 2001 site, followed by the 1990/2007 site, the LSU, the 1996 site and the 1987/2004 site. Analysis of these variables did not indicate a significant difference between the means.

Table 6—Comparison of stand structure variables on five treatments types in the Byrne-Milliron Forest in Santa Cruz, California

	1987/2004		**1990/2007**		**1996**		**2001**		**LSU**	
	Mean	*S.E.*	*Mean*	*S.E.*	*Mean*	*S.E.*	*Mean*	*S.E.*	*Mean*	*S.E.*
Redwood basal area	20.80	3.65	27.81	6.22	22.15	4.77	20.85	4.05	20.42	3.34
Coast redwood dominance	90.65	5.21	96.85	1.60	90.75	4.73	99.3	0.41	96.05	2.80
LWD density	4.60	0.61	4.05	0.54	3.15	0.56	3.20	0.57	6.60	0.42
Coast redwood associates	20.03	3.66	15.09	2.89	16.20	3.22	13.54	3.31	43.99	5.95
Canopy cover	0.97	0.004	0.98	0.002	0.93	0.019	0.98	0.004	0.98	0.002

Large Woody Debris

The occurrence of LWD varied among the five treatments (table 6). The LSU had the highest mean, followed by the 1987/2004 site, the 1990/2007 site, the 2001 site, and the 1996 site (fig. 3 ~~5~~). When analyzed with years since last harvest, LWD density was significantly higher in the LSU compared with the 1996 site and 2001 site (p = 0.015, p = 0.018). This relationship was negatively correlated (r = -0.32). Analysis of LWD density and the percentage cut per hectare also indicated a significant difference between the LSU in comparison with other sites (p = < 0.001, r = -0.36).

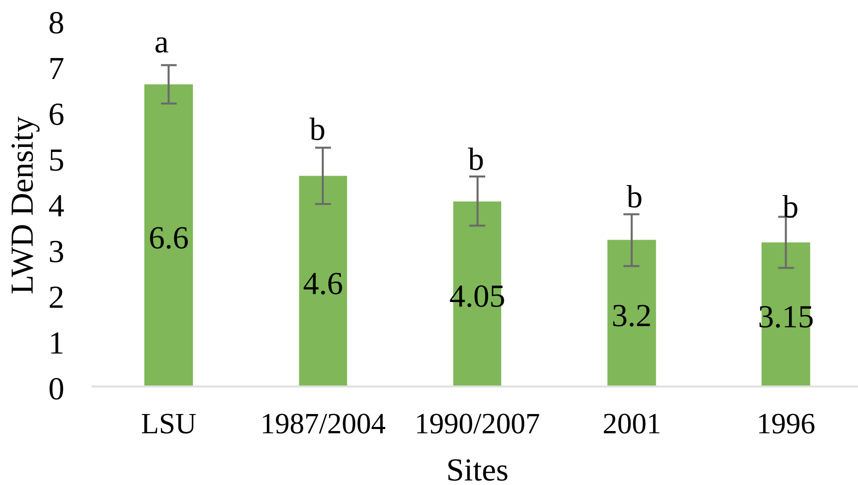

Figure 3—Mean large woody debris (LWD) among five treatments with 95% confidence intervals when correlated with percentage cut (sites with the same letter were not significantly different from one another).

Understory Species

Common coast redwood herbaceous associates including hooker's fairy bells, modesty, Pacific starflower, Pacific trillium, redwood violet, and redwood sorrel were further analyzed to determine potential differences among treatments. The LSU 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 (table 6, fig. 4). When analyzed with years since last harvest, results indicated that the percentage of coast redwood associates was significantly higher in the LSU compared with the 2001 site (p = 0.026, r = -0.25). Analysis of percentage cut also indicated the LSU had a significantly higher amount of these species in comparison with all other treatments (p = < 0.001, r = -0.47). The 2001 site had the highest native species richness, followed by the 1990/2007 site, the 1987/2004 site, the LSU and the 1996 site. When ANOVA was used to analyze this data with the independent variables, there was not a significant difference for species richness between treatments.

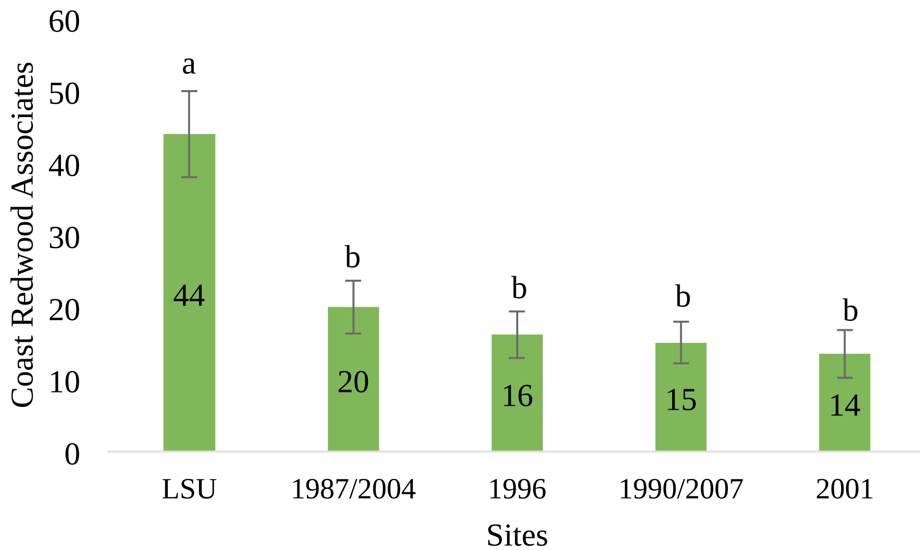

Figure 4—Mean percentage of coast redwood associated species among five treatments with 95 percent confidence intervals when correlated with percentage cut (sites with the same letter were not significantly different from one another).

The LSU had the lowest frequency of exotic species. This was followed by the 2001 site, the 1996 site, the 1987/2004 site, and the 1990/2007 site. There was a significant difference between the LSU and the 1990/2007 site when analyzed with percentage cut per hectare (p = 0.007, r = 0.27). Incidentally, the 1990/2007 site had the highest frequency of exotic species as well as the highest percentage cut, whereas the LSU had the lowest frequency and the lowest percentage cut.

Canopy Cover

The LSU appeared to have the highest canopy cover, followed by the 1990/2007 site, the 2001 site, the 1987/2004 site, and the 1996 site (table 6). The 1996 site had significantly lower cover than all other sites in relation to years since last harvest and percentage cut (p = < 0.001, 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).

Discussion

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

Mean stand density, which was lowest in the LSU, had the strongest correlation to the percentage cut per hectare. This 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 coast redwood. 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 LSU, which had the lowest percentage cut of all sites, had the lowest mean number of stems < 49 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. Although studies have promoted heavier thinning intensities to speed up the attainment of desirable structural features (Berrill 2009, Oliver et al. 1994), 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 (Floyd et al. 2009, Lutz and Halpern 2006; Russell et al. 2014, Sachs et al. 1993).

Density of LWD, which was considered relatively uniform among treatments prior to harvest, was highest in the LSU. This was not unexpected, as LWD is a characteristic feature of late seral forests. However, results indicated a decline in LWD in relation to the percentage cut. This could be due in part to a regulation specific to the Southern Subdistrict which requires that remaining slash not exceed 76.2 cm (30 inches) (CAL FIRE 2016). It is possible that some pre-existing LWD was also removed during the clean-up process in an effort to adhere by these guidelines.

In addition, the LSU had a significantly higher percentage of coast redwood understory associated species in comparison with the other treatments. The 2001 site, which had one of the highest harvest yields, incidentally had the lowest percentage of coast redwood associated species. These species were found to decline significantly in relation to percentage cut per hectare; this was the strongest correlation of all dependent variables analyzed.

The LSU, which experienced the lowest level of disturbance, also had the lowest exotic richness of all sites. Exotic species richness increased with the 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 site with the highest cut (1990/2007) had the highest exotic species richness. These results were not surprising since research has shown that exotic species are more likely to increase following a disturbance (Blair et al. 2010, Ebrecht and Schmidt 2003).

The LSU also had the highest canopy cover among treatments. Incidentally, the 1996 site, which had significantly lower canopy cover in comparison, had the lowest number of coast redwood associated species. 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).

In many respects, the LSU 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 complex canopy structures have allowed this area of the Byrne-Milliron Forest to maintain a healthy ecosystem amidst active management.

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Low Thinning and Crown Thinning of Two Severities as Restoration Tools at Redwood National Park[1]

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Abstract

Interest in the restoration of second-growth forests has continued to increase in the redwood region, which has further increased the importance of evaluating restoration-based silvicultural strategies. This study assessed the short-term effectiveness of four silvicultural treatments (two silvicultural thinning methods, low thinning and crown thinning, and two basal area retentions, 80 percent and 45 percent) as forest restoration tools via analysis of relative basal area growth at Redwood National Park. Prior to treatment, the second-growth stand had more than 1,600 trees ha^{-1} and 70.0 m^2 ha^{-1} basal area and consisted primarily of two species, Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) (the dominant species) and redwood (*Sequoia sempervirens* (D. Don) Endl.). Growth was enhanced for all treatments with 5-year net basal area gains of 28.4 percent for the low-retention crown thinning, 28.1 percent for the low-retention low thinning, 23.3 percent for the high-retention crown thinning, 19.1 percent for high-retention low thinning, and only 14.2 percent for the control. We conclude that all four thinning treatments improved tree growth; but among them, the low-retention treatments were most effective in accomplishing restoration objectives, while the high-retention low thinning was least effective. Increasing the array of silvicultural tools that Redwood National Park can use may prove helpful in accomplishing restoration goals in future projects.

Keywords: ecological restoration, forest stand dynamics, second-growth, *Sequoia sempervirens*, silviculture

Introduction

Approximately half of Redwood National Park is comprised of young, second-growth forest. Managers at Redwood National Park are using silvicultural treatments to adjust species composition, increase tree growth, and improve stand vigor to promote old forest characteristics (Teraoka 2012,Teraoka and Keyes 2011). Low thinning, the removal of trees in the lower crown classes to benefit trees in the upper crown classes (Smith et al. 1997), has been utilized by the park as the primary silvicultural method. Because the largest trees are retained, low thinning mimics the patterns of mortality in even aged stands (Chittick and Keyes 2007, Teraoka and Keyes 2011), thus accelerating a process that would naturally occur.

Many of the park's second-growth stands were initiated via aerial seeding of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) at high densities (Teraoka and Keyes 2011). Empirical growth and yield studies of second-growth stands have shown that Douglas-fir outpaces redwood (*Sequoia sempervirens* (D. Don) Endl.) in height growth during the early to middle stages of stand development (Lindquist and Palley 1963, Wensel and Krumland 1986), thereby promoting Douglas-fir to a dominant canopy position (Plummer 2008, Teraoka and Keyes 2011). The capacity of Douglas-fir to outpace redwood in height growth and sustain canopy dominance is an impediment to the restoration of redwood dominance that silvicultural restoration treatments should seek to mitigate. Analysis at a nearby site with similar conditions revealed that early thinning in young, dense, second-growth stands benefitted redwood proportionately more than Douglas-fir, thereby elevating its capacity for dominance (Plummer et al. 2012).

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Should Redwood National Park consider altering stand trajectories by utilizing silvicultural methods other than those currently being employed? Is simply accelerating the natural pattern of mortality via low thinning the best restoration-based silvicultural tool, or would another thinning method better facilitate restoration objectives? How does thinning severity interact with thinning method to promote old forest characteristics? To further investigate these questions, we analyzed the park's A-972 Forest Restoration Study, a thinning experiment that was conducted in 2008.

Methods

The study area is located approximately 4 km south of Orick, California and 2 km from the Pacific Ocean (fig. 1). The legal description is Rodger's Peak U.S. Geological Survey quadrangle SW1/4 NW1/4 and NW1/4 SW1/4 Sec.16, T. 10N. R.1E. H.B.M. Elevation is 300 m. Annual precipitation averages 178 cm. Slope ranges from 0 to 30 percent. The stand was clearcut in 1968 (prior to park acquisition), followed by broadcast burning and aerial seeding. Species composition is comprised of Douglas-fir, redwood, Sitka spruce (*Picea sitchensis* [Bong.] Carr.), western hemlock (*Tsuga heterophylla* [Raf.] Sarg.), and red alder (*Alnus rubra* Bong.). Understory vegetation is nearly absent, but small pockets of ferns, rubus, and ericaceous species occurred before treatment, in 2008, throughout the study site.

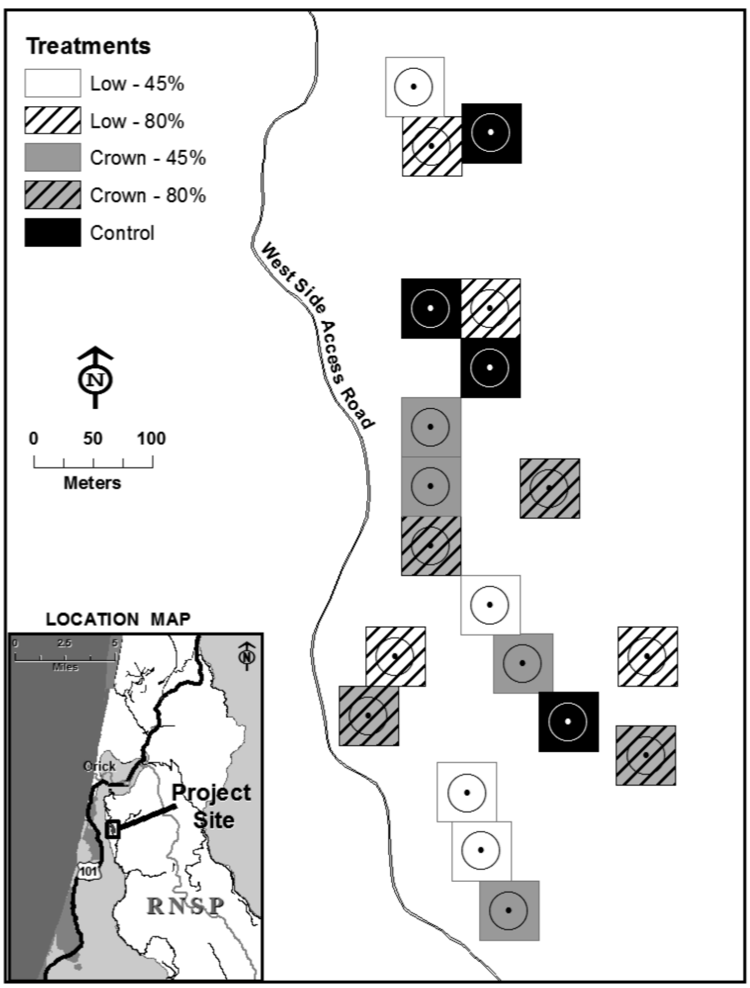

Figure 1—Location map and arrangement of treatment units and plots in the A-972 Study Area. Circles within the square units represent 0.08-ha monitoring plots.

In fall of 2007, 20 square units (0.25 ha) were established in the 40-year-old stand (fig. 1). Species and diameter at breast height (DBH; 1.37 m) of live trees were recorded for all stems greater than 10 cm DBH within a circular fixed-radius monitoring plot (0.08 ha) located at the center of each unit. Five groups were created: an untreated control and four thinning treatments. The four treatments consisted of two silvicultural thinning methods — low thinning (the removal of trees in the lower crown classes to benefit trees in the upper crown classes) and crown thinning (the removal of trees in the upper crown classes, primarily co-dominant trees, to benefit trees of the same crown classes) (Smith et al. 1997) — each with two basal area retentions of 80 percent and 45 percent. In meeting each unit's retention target, Douglas-fir was primarily targeted for removal, and clumps of redwood stump sprouts were thinned. All wood was lopped-and-scattered. The five groups are summarized as follows (abbreviated naming conventions in parentheses):

1. Low thinning with a 80 percent retention of basal area (L80)
2. Low thinning with a 45 percent retention of basal area (L45)
3. Crown thinning with a 80 percent retention of basal area (C80)
4. Crown thinning with a 45 percent retention of basal area (C45)
5. Control

Treatments and controls were randomly assigned to the units and replicated four times each (fig. 1). In the winter of 2007 to 2008, treatments were applied to the units totaling 4 ha of thinning.

Plots were resampled immediately after treatment, and again in 2013. Means of stand density (trees ha^{1} [tph] and basal area) were calculated for each measurement period. Relative stand basal area growth of the surviving trees and standard errors were calculated for all species combined, for redwood only, and for Douglas-fir only. Relative stand basal area growth used here was calculated as the total net stand basal area growth that occurred after thinning divided by the stand basal area immediately after thinning at the beginning of the 5-year growth period. Relative stand basal area growth was analyzed via one-way ANOVA and Tukey's post-hoc test using a pre-set alpha of 5 percent to determine whether differences in relative stand basal area growth among groups had occurred as a result of the treatments and the passage of time.

Results

Prior to treatment, there were no significant differences between groups for total number of trees (p = 0.725) or basal area (p = 0.674). Collectively, the stand averaged about 1,600 trees ha^{-1} and 70.0 m^2 ha^{-1} basal area; both stand density metrics were dominated by Douglas-fir with lesser amounts of redwood (fig. 2).

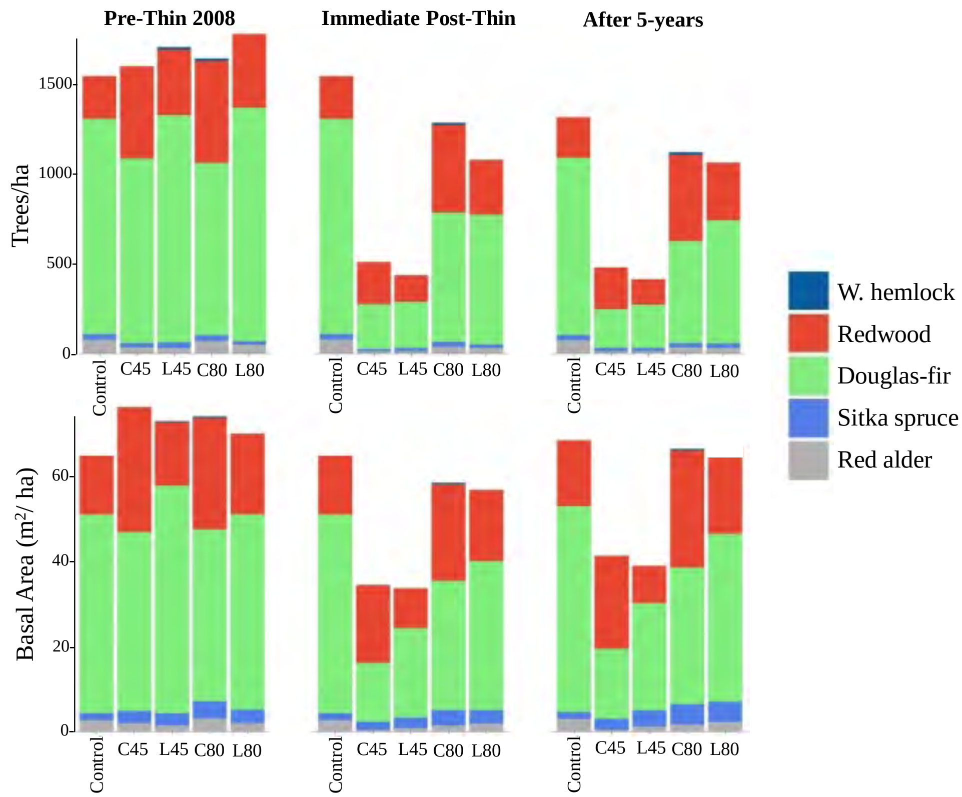

Figure 2—Number of trees per hectare (top) and basal area (bottom) for each group by species: before thinning (2008; left), immediately after thinning (2008; middle), and 5 years after thinning (2013; right).

Implementation of the low thinning treatments proved to be slightly more severe as compared to the crown thinning treatments at the same retention targets (fig. 2). Low thinning required the removal of relatively higher number smaller-diameter trees as opposed to crown thinning, which required the removal of a relatively lower number of larger-diameter trees.

Stand basal area growth for all species was apparent 5 years after treatment for all groups, with net basal area gains of 28.1 percent for L45, 28.4 percent for C45, 23.3 percent for C80, 19.1 percent for L80, and 14.2 percent for the control (fig. 3). Basal area growth for treatments C45 ($p < 0.001$), L45 ($p < 0.001$), and C80 ($p = 0.001$) were significantly greater than the control. Basal area growth for the L80 treatment ($p = 0.05575$) was not significantly different than the control. There were no differences in terms of basal area growth between treatments of the same basal area retentions (C45 - L45, $p = 0.9999$; C80 - L80; $p = 0.2140$). Both C45 ($p = 0.00454$) and L45 ($p = 0.00568$) were significantly different than L80.

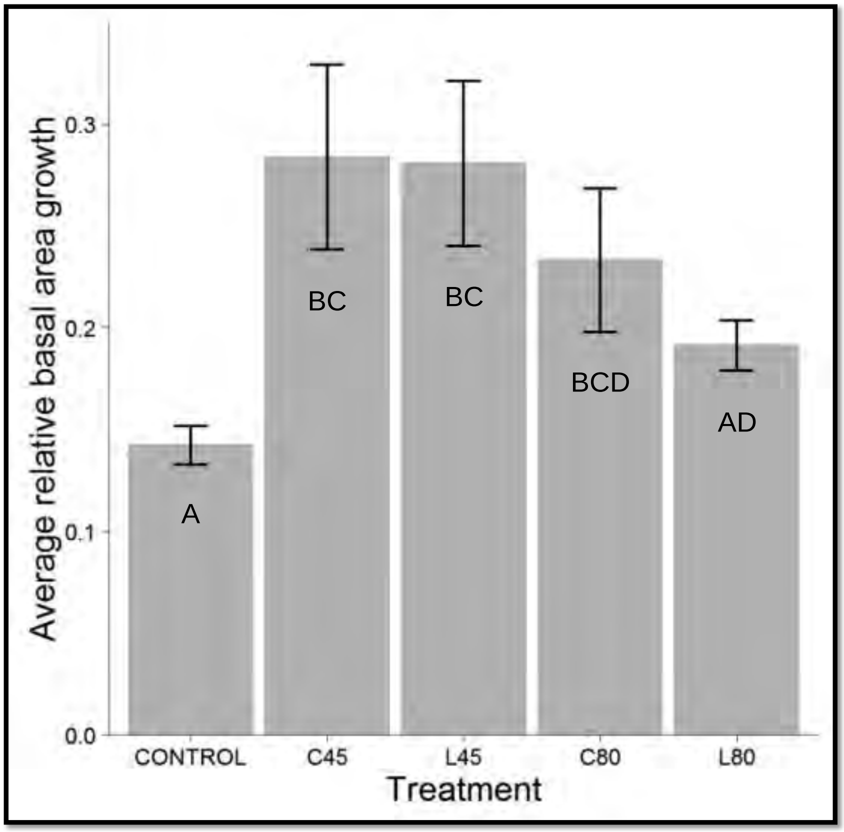

Figure 3—Average relative stand basal area growth and one standard error per treatment for trees of all species for the 5-year post-treatment period. Different upper case letters within the bars represent significant differences among the means when one-way ANOVA and Tukey's post-hoc test were performed at a pre-set alpha of 5 percent.

Basal area growth for Douglas-fir had net 5-year gains of 25.6 percent for L45, 24.2 percent for C45, 17.4 percent for C80, 16.6 percent for L80, and 12.9 percent for the control. Redwood had more relative basal area growth than Douglas-fir over the 5-year period, with net basal area gains of 31.3 percent for L45, 30.2 percent for C45, 26.9 percent for C80, 24.6 percent for L80, and 17.3 percent for the control (fig. 4). Pairwise comparisons between groups for individual tree species produced results similar to that of all species combined. Redwood basal area percent increase appeared to respond more positively to thinning compared to Douglas-fir ($p < 0.001$) across all treatments, yet it also out-paced Douglas-fir in the untreated control (fig. 4).

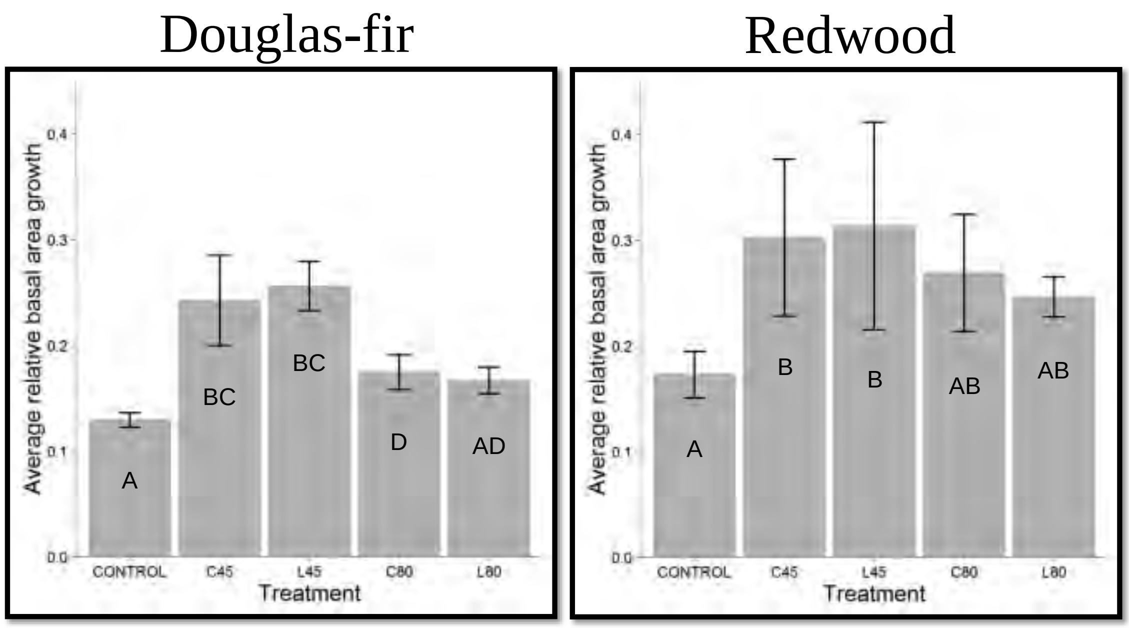

Figure 4—Average relative stand basal area growth and one standard error per treatment for Douglas-fir (left) and redwood (right) for the 5-year post-treatment period. Different upper case letters within the bars represent significant differences among the means when one-way ANOVA and Tukey's post-hoc test were performed at a pre-set alpha of 5 percent.

Discussion

All four thinning treatments improved stand basal area growth over the short-term period relative to the control. The more severe (45 percent retention) thinning treatments were generally more effective in maximizing stand-level growing conditions. Least effective was the low-severity, low thinning (L80) treatment, which produced growth that was just marginally greater than the control.

These results, particularly for the low-severity, 80 percent retention thinning treatments, were consistent with the findings and conclusions made earlier by Teraoka and Keyes (2011), who found that a low-severity low thinning, in the short-term, only weakly altered development patterns in the upper canopy. Our results from this more expansive study further suggest that crown thinning, or more aggressive thinning in general, could be more successful in altering stand development by effectively redistributing growing space to redwoods, and thus could better favor redwood dominance over the course of subsequent stand development.

The heavier thinning treatments were consistent with other growing stock studies that have shown that higher-severity thinning increased relative growth, but the increase was generally offset by the decrease in total yield associated with retention of less growing stock per unit area (Curtis and Marshall 2002, Lindquist 2004, Oliver et al. 1994).

These results lend support to utilizing higher severity thinning treatments, and indicate that other silvicultural methods— such as crown thinning—may be more effective at achieving redwood forest restoration objectives. Having a wide array of forest restoration tools at Redwood National Park’s disposal gives managers greater flexibility in achieving restoration goals. Utilizing higher severity low thinning, other traditional thinning methods, and alternative thinning methods (such as variable-density thinning; Keyes et al. 2010, O’Hara et al. 2010, O’Hara et al. 2012), may prove more effective. Identifying powerful, efficient silvicultural techniques will be key to achieving the park’s forest restoration objectives, as funding constraints and removal of roads makes multiple-entry restoration treatments unfeasible.

It is evident from existing long-term thinning studies in the Pacific Northwest that the main benefits of thinning are not necessarily increased basal area or volume production, but larger and more vigorous trees, enhanced stand stability, the enhancement of floral diversity, and increased aesthetic values and wildlife habitat (Curtis et al. 1997, Curtis and Marshall 2002, Keyes 2011, Webb et al. 2012).

Similar to other long-term thinning studies in the Pacific West, the A-972 study demonstrates the influence that restoration-based silvicultural treatments can have on retained tree growth over a relatively short period. Yet this study serves as a rare thinning study of mixed Douglas-fir-dominated redwood forest of moderate site quality; similarly-designed studies in the redwood region tend to have been located in high site quality, redwood-dominated stands (Lindquist 2004, Oliver et al. 1994, Webb et al. 2012). We intend to conduct further studies at this site to analyze other aspects of the growth-to-growing stock relationship—such as treatment-induced differences in understory dynamics, tree regeneration, and overstory species composition—to better inform forest restoration strategies at Redwood National Park.

Acknowledgments

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Long Term Results of Early Density Management of a Third Growth Redwood Stand[1]

Lynn A. Webb,[2] John-Pascal Berrill,[3] and James L. Lindquist[4]

Abstract

Precommercial or early thinning of regenerating redwood forests can support management objectives including maximizing yield, forest structure restoration, and promoting carbon sequestration. We present data collected over 30 years following a precommercial thinning (PCT) in a 19 year-old naturally regenerated and planted coast redwood (*Sequoia sempervirens*(D. Don) Endl.) and coast Douglas-fir (*Pseudotsuga* menziesii (Mirb.) Franco var. *menziesii*) stand. The study site is typical of most regenerating redwood forests with spatial variation in tree distribution from stump sprouting (coppice) redwood interspersed with planted and natural regeneration. Three replicates of six density treatments (100, 150, 200, 250, 300 trees per acre (TPA) and unthinned control) were established in 1981 and re-measured repeatedly. Plot size was reassessed in 2011. Post-thinning density had a lasting effect. Thirty years after PCT, average diameter at breast height (DBH) and stand density index (SDI) still differed among PCT treatment levels. When focusing on the largest trees (140 TPA), basal area (BA) and cubic foot volume results were significantly affected by post-thin TPA resulting in the highest production from treatments ranging from 120 to 250 TPA. A new method of characterizing site productivity by indexing redwood BA growth revealed that variations in species composition and site quality explained differences in growth and yield between plots. Multiple linear regressions revealed that post-thin TPA, species composition (percent redwood SDI), and site quality in terms of BA growth index all had a lasting influence on redwood productivity monitored over three decades after PCT.

Key words: precommercial thinning, *Sequoia sempervirens*, silviculture, site quality, stand density management

Introduction

Reducing stand density of redwood (*Sequoia sempervirens* (D. Don) Endl.) forests via commercial timber harvest is well understood to influence both residual tree growth and the forest's structure and subsequent development (Berrill and O'Hara 2014, Oliver et al. 1994, Webb et al. 2012). Tree growth can also be enhanced by precommercial thinning (PCT) to adjust the density of younger redwood stands (Lindquist 1998, 2004, 2007; O'Hara et al. 2015). PCT is also used to enhance representation of desired species or to favor improved planting stock by preferential removal of less desirable trees, to improve stand condition by removing unhealthy or malformed trees, and to modify the spatial arrangement of residual trees. A PCT study in young third-growth redwood stands in North Fork Caspar Creek Watershed on Jackson Demonstration State Forest (JDSF) found significant differences in individual tree attributes 12 years after the 8 to 11 year-old trees were thinned (O'Hara et al. 2015). Factors listed that can contribute to variability are inter- and intra-specific competition, animal damage, and differing vegetation management during stand establishment. Young redwood stands are inherently variable because of the clumpy nature of both redwood and hardwood stump sprout regeneration. Complex regional topography may contribute to fine scale variability in site quality and growth as well (Berrill and O'Hara 2016).

Lindquist (1998, 2004, 2007) reported findings from a PCT experiment designed to determine the response of coast redwoods to a variety of stocking levels following precommercial thinning and help determine optimum density for different management objectives. When contrasting the different PCT

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[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.

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TPA retention levels, Lindquist found that average diameter remained significantly different through time, but stand increment and measures of basal area or volume were not significantly different. The question as to when or how thinning young redwood stands affects the volume or basal area increment remains unclear. This paper examines patterns of stand development by combining Lindquist's data with recent data from the same experiment.

Methods

Study Design

The PCT study site was installed on JDSF at the site of the l960 Caspar Cutting Trials clearcut (Lindquist 1988) as a cooperative study by the U.S. Department of Agriculture Forest Service and the California Department of Forestry and Fire Protection. Three blocks were established on the east-facing mid to lower slope. The northern block had been partially burned. Prior to thinning treatment, the middle block had four times as many Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco as the northern block. Six density treatments based on number of trees retained (100, 150, 200, 250, 300 TPA and unthinned control) were implemented on adjacent 1.6 ha (0.4 ac) treatment areas. The 0.2 acre square measurement plots were installed within the treatment areas. One replicate was randomly assigned to each of the three blocks (fig. 1). Thinning was implemented in 1981 by individual plot quadrant; retaining targeted number of trees, with a preference for larger thrifty redwoods. The redwood stump sprouts were thinned to leave at least 0.61 m between stems. Plots were initially measured in 1981, and then remeasured in 1984, 1986, and 1998 by Lindquist and in 2011 by Green Diamond Resource Company inventory staff. The 2011 remeasurement used laser and hypsometer to more accurately measure plot size and map stems. We performed ANOVA tests for differences among plots, PCT treatment levels and replicate blocks (north, central, and south replicates of each PCT treatment level).

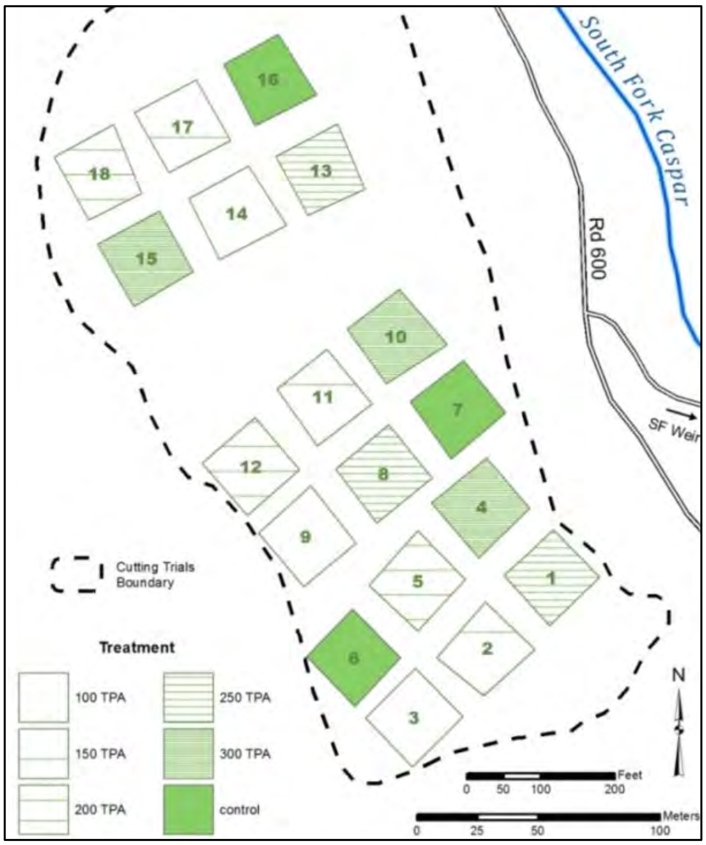

Figure 1–Plot layout with 1981 PCT treatment as TPA retention levels. Each of the 3 blocks has one each of the treatment levels.

The term 'crop trees' describes the largest or best-formed trees expected to become a component of a future commercial harvests or the dominant canopy trees in restored stands. Studying their growth separately is useful because trees of lower stature such as ingrowth and low vigor trees not expected to attain merchantable size are ignored. In redwood-dominated stands, stand density index (SDI) is expected to attain approximately 600 before competition-induced mortality of redwood trees is imminent (Long 1985, Reineke 1933). This threshold will not be surpassed until stands with 140 TPA have exceeded 61 cm average diameter at breast height (DBH; 1.37 m), when harvesting some these so-called crop trees would be of interest. Therefore data for crop trees only, defined as the largest-DBH 140 TPA in each plot was analyzed or for the 100 TPA PCT treatment all trees were analyzed.

Volume Estimates

A slightly different method was used to calculate volume than Lindquist (2004, 2007) used, since all heights were measured in 2011, whereas in the past, limited numbers of tree heights had been measured. For redwoods, height-diameter relationships were derived by measure year then used to estimate missing heights. Because very few Douglas-fir were present, a regression fitted to available height-diameter data for all measurement years was used to predict missing heights. Other volume calculations were consistent with Lindquist's (2007) methods and used Wensel and Krumland (1983) equations.

Assessing Site Quality and its Effect on Growth in Each Plot

SDI was calculated for the entire plot (all species) and for the redwood component in each plot. This gave species composition in terms of redwood percent SDI. We used this percentage to calculate the approximate plot area occupied by redwood. We then used this "SDI-adjusted plot area" as the expansion factor to calculate SDI and basal area increment (BAI) for all redwood trees in the plot, giving redwood SDI and BAI on a per acre basis. For each plot, we calculated two alternate site quality variables: (1) dominant redwood height (i.e., average height of 40 largest DBH redwood per acre at age 49 years), and (2) redwood BA growth index (RBAGI) (i.e., how much redwood BA growth deviated from an 'expected' modeled value for any plot SDI; Berrill and O'Hara 2014). To test for differences in average site quality among plots receiving different thinning prescriptions, we constructed ANOVA tests for dominant redwood height among treatments and BAI index among treatments. To investigate the possibility that species composition or site quality had influenced growth of redwood in each plot, we regressed redwood BAI for each plot against treatment (TPA after thinning) and candidate predictor variables: redwood composition, dominant redwood height, and RBAGI. The high TPA in control plots relative to all other treatments caused problems (skewness) when TPA was used as a covariate in the multiple linear regressions. Therefore we excluded data for the three control plots in these regressions. We compared Akaike's Information Criterion (AIC) among regressions (Burnham and Anderson 2002) to identify the combination of variables most likely to explain observed differences in redwood BAI between plots (after accounting for the thinning treatment effect by including post-thin TPA as a covariate in each regression).

Results

Ingrowth and Mortality

Implementation of the PCT treatments resulted in the actual TPA in each plot immediately after PCT was close to the prescribed TPA goal. Ingrowth was variable and consisted of Douglas-fir seedlings and redwood stump sprouts. Mortality occurred primarily in Douglas-fir with the exception of the unthinned plots where mortality occurred across all species (table 1).

Table 1—Ingrowth and mortality 30 years after precommercial thinning (PCT) treatment in 19 year old stand, and actual vs planned treatment TPA

Plot	Block	Treatment TPA 1981	Actual TPA 1981	Ingrowth TPA 2011	Species ingrowth[a]	Mortality TPA 2011	Species mortality
14	N	100	97	19	RW	15	DF
9	M	100	106	19	RW	0	
3	S	100	109	5	TO	0	
17	N	150	148	15	RW	0	
2	S	150	157	0		0	
11	M	150	159	5	RW	31	DF
12	M	200	199	5	RW	5	DF
5	S	200	208	0		0	
18	N	200	208	0		42	DF
8	M	250	255	0		15	DF
13	N	250	267	0		42	DF
1	S	250	292	0		0	
15	N	300	290	10	RW, DF	39	DF
4	S	300	296	0		10	DF
10	M	300	302	0		74	DF
16	N	Control	789	5	RW	368	RW,TO,DF
6	S	Control	891	0		443	RW,TO,DF
7	M	Control	935	0		455	RW,TO,DF

[a] RW = redwood, D = Douglas-fir, TO = tanoak.

Diameter

The 1981 PCT treatment had a lasting effect on DBH development. As measured in 2011 (age 49), thinning to lower densities significantly enhanced average DBH for redwood (p = 0.0002), but not Douglas-fir (p = 0.15) (fig. 2).

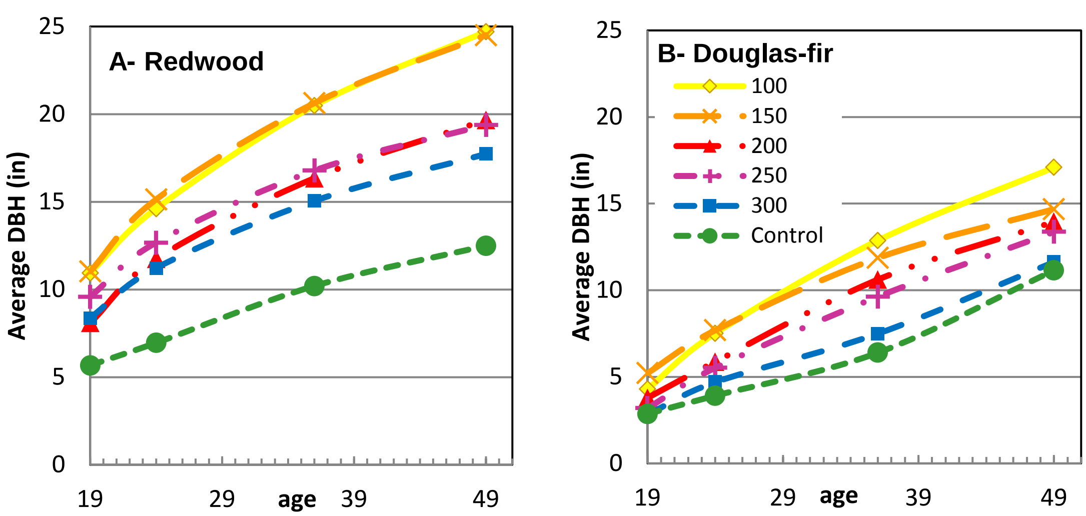

Figure 2—Change in DBH through time by species, A - redwood and B - Douglas-fir, and PCT treatment (TPA retained: 100, 150, 200, 250, 300 and unthinned control).

Basal Area

The 1981 PCT reduced TPA and BA in thinned plots (fig. 3). After PCT, BA development proceeded at slightly different rates, presumably as a consequence of differences in tree size, vigor, and site quality. The unthinned control and 100 TPA treatments showed relatively slow BA development, and the 250 TPA treatment had more rapid BA development relative to the other treatments. Thinning did not result in significantly different BA at age 49 among the three replicate blocks where each PCT treatment was repeated.

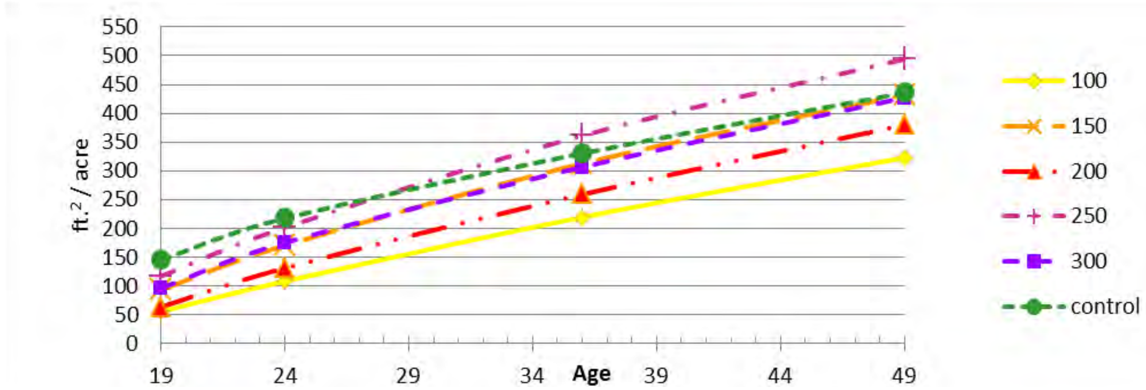

Figure 3—Average basal area (BA) over time by 1981 PCT treatment TPA retention levels.

When the tests were repeated for crop trees (i.e., 140 largest TPA), both the PCT treatment (p = 0.027) and the replicate blocks (p = 0.032) were significantly different. The 150 TPA thinning resulted in the greatest BA (table 2). BA of crop trees was lower three decades after PCT in the controls, which was expected, and in the 100 TPA treatment where the penalty of having fewer than 140 TPA counting towards BA was not overcome by more rapid individual tree growth.

Table 2—Evaluation of the 140 largest TPA (i.e., crop trees) by PCT TPA for the 49 year old stand

PCT TPA	BA ft^2/acre crop tree	ft^3/acre crop tree	ft^3 crop % total	range ft^3 crop tree%	Bf/acre crop tree	Bf crop % total[a]	range Bf crop tree%
100	322	11,358	100%	100%	61,777	100%	100%
150	432	15,213	99%	99-100%	82,051	99%	99-100%
200	338	11,035	91%	83-99%	56,255	92%	84-99%
250	388	13,657	79%	74-86%	72,242	80%	76-88%
300	325	10,962	80%	75-85%	56,168	82%	77-86%
control	289	9,628	71%	65-79%	48,521	75%	69-82%

[a] The volume “crop % total” refers to ratio of volume present in the 140 largest TPA to the total volume per acre expressed in percent.

Cubic Volume

Cubic foot volume increased exponentially through time (fig. 4A). The PCT treatment did not have a significant effect on cubic foot volumes in the 49 year-old stand. When only the 140 largest TPA were considered, both PCT treatment (p =.046) and replicate block (p =.023) were both significant determinants of crop tree volume per acre. The crop tree evaluation had different trends than total volume; the former had lowest volume for control (table 2) and the latter had lowest volume for 100 TPA PCT. This was expected as the result of relatively fewer large trees in the 100 TPA vs the abundant number smaller trees in the control.

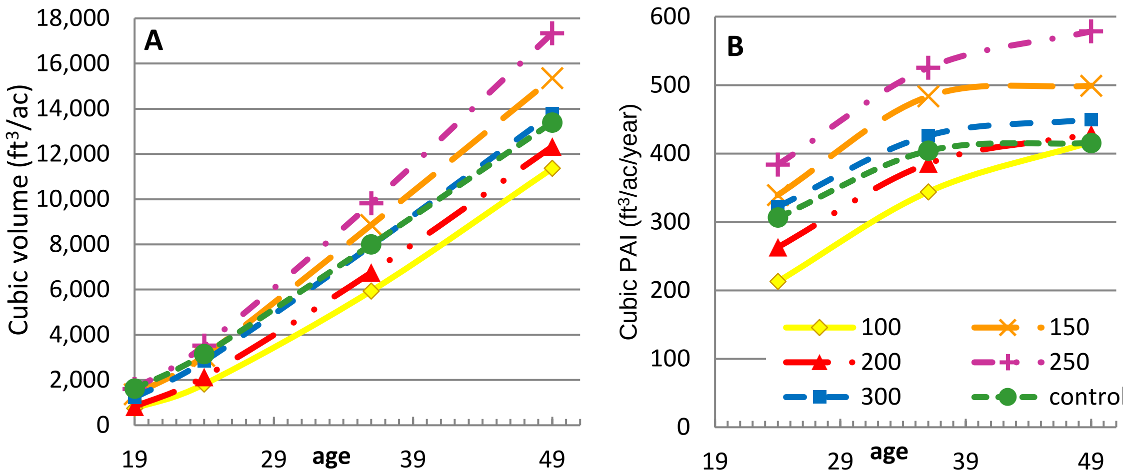

Figure 4—Average cubic foot volume per acre through time by 1981 PCT treatment TPA retention levels. A (left) is total volume per acre (ft^3/ac). B (right) is periodic annual volume increment (PAI) between measurement periods (ft^3/ac/year).

The cubic foot periodic annual increment (PAI) per year for each period between consecutive measurements indicated that stand growth rate was greater in later periods. However, counter to expectations, cubic volume PAI was not proportional to post-thin TPA (fig. 4B). Presumably, high variability among plots prevented detection of significant differences among PCT treatment levels and replicate blocks.

Merchantable Volume

Board foot volume (Bf) is an estimate that applies to the merchantable portion of the tree. On a per acre basis, merchantable volume exhibited the same trends as cubic foot volume (fig. 5). Thirty years after PCT to 250 and 150 TPA, these age-49 plots carried more Bf volume than any other treatment. The ANOVA test detected no difference among thinned TPA treatments or blocks. However, when the analysis of the 140 largest crop trees per acre was conducted, Bf volume differed among the replicate blocks ($p = .022$) and the PCT TPA ($p = .053$). The positive effect of PCT treatment focusing growth on the larger crop or canopy trees is evidenced as the Bf volume of crop trees at age 49 for the 150 TPA treatment had almost twice the volume as the control (82,051 vs 48,521 Bf/ac). Periodic annual increment (PAI, annual Bf growth over period between consecutive measurements) for Bf volume per acre had similar trends as PAI for cubic foot volume.

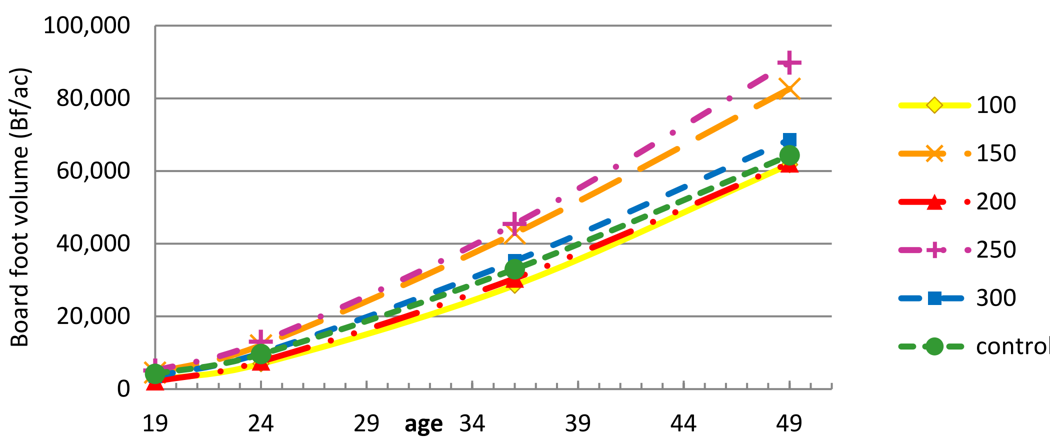

Figure 5—Average board foot volume (Bf) per acre by 1981 PCT TPA levels through time.

Species Composition and Stand Density Index

The proportion of other species to redwood can be characterized using SDI as a proxy for the amount of growing space occupied by each species. The proportion of redwood SDI to total SDI varied from 80 percent to 91 percent. Figure 6 shows the variation in stand density by treatment for the major species (fig. 6A) and displays total SDI development through time for the different PCT treatments levels (fig. 6B). SDI at age 49 years differed significantly among the thinning TPA treatment levels ($p = .014$). Much like stand BA, SDI did not follow the expected pattern of proportionality to post-thin TPA (actual SDI by PCT TPA ranked from lowest to highest: 100 < 200 < 150 < 300 < 250 < control). Immediately after PCT at age 19, all the plots were below full site occupancy for redwood. SDI of 350 is 35 percent of maximum SDI for redwood and over 50 percent of maximum SDI for Douglas-fir (Long 1985, Reineke 1933). The 49 year old stand has SDI greater than 600 for the higher retention PCT treatments. SDI of 600 is 60 percent of the maximum SDI for redwood and 100 percent of maximum SDI for Douglas-fir so the mortality shown in table 1 high density retention, seems to validate the concept of imminent mortality for this redwood - Douglas fir stand (Drew and Flewelling 1979, Long 1985).

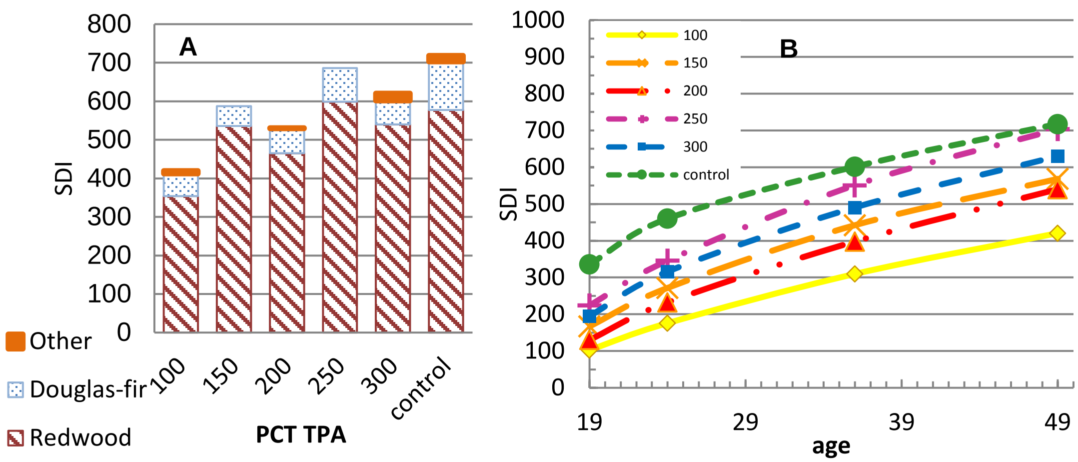

Figure 6–Stand Density Index (SDI) in 2011 (age 49) by PCT treatment, TPA retained and species, where “other species” mainly included grand fir and tanoak (A), and SDI for all species through time by PCT treatment (B).

Species Composition and Site Quality Affect Growth in Each Plot

Examining the development of redwood in each plot provided insight into factors affecting growth of the dominant stand component (i.e., redwood component). Plot-level redwood BAI was at or above the expected BAI curve developed for JDSF in a prior study (Berrill and O’Hara 2014), indicating that this was a very good redwood site. However, redwood BAI per acre was also highly variable among plots, as it ranged from 5 percent to 85 percent above the expect BAI for plot SDI (fig. 7A). Redwood BAI varied widely among plots and treatments (fig. 7B). Dominant redwood height at age 49 years also varied among plots (fig. 7C), and appeared to be lower in the 200, 300, and 500 TPA treatments, but did not differ significantly among the different thinning prescriptions ($p = 0.53$; fig. 7D). Species composition and site quality were correlated. Plots with a greater proportion of redwood SDI also had taller dominant redwood trees ($R^2 = 0.34$; $p = 0.007$; fig. 7C) and higher values of BA growth index for redwood (RBAGI) ($R^2 = 0.29$; $p = 0.01$; fig. 7E). Greater redwood BAI was found in plots with higher RBAGI ($R^2 = 0.80$; $p < 0.001$) and taller dominant redwood trees ($R^2 = 0.33$; $p = 0.007$). RBAGI and dominant redwood height in each plot were correlated, but not strongly ($R^2 = 0.33$; $p = 0.008$). The RBAGI values (describing site quality, in terms of deviation from the expected

BAI) varied among plots (fig. 7E) but did not differ significantly among thinning prescriptions (p = 0.33; fig. 7F). This was consistent with other basal area measurements where variation across the site (i.e., variation among plots and replicate blocks) may have obscured treatment effects.

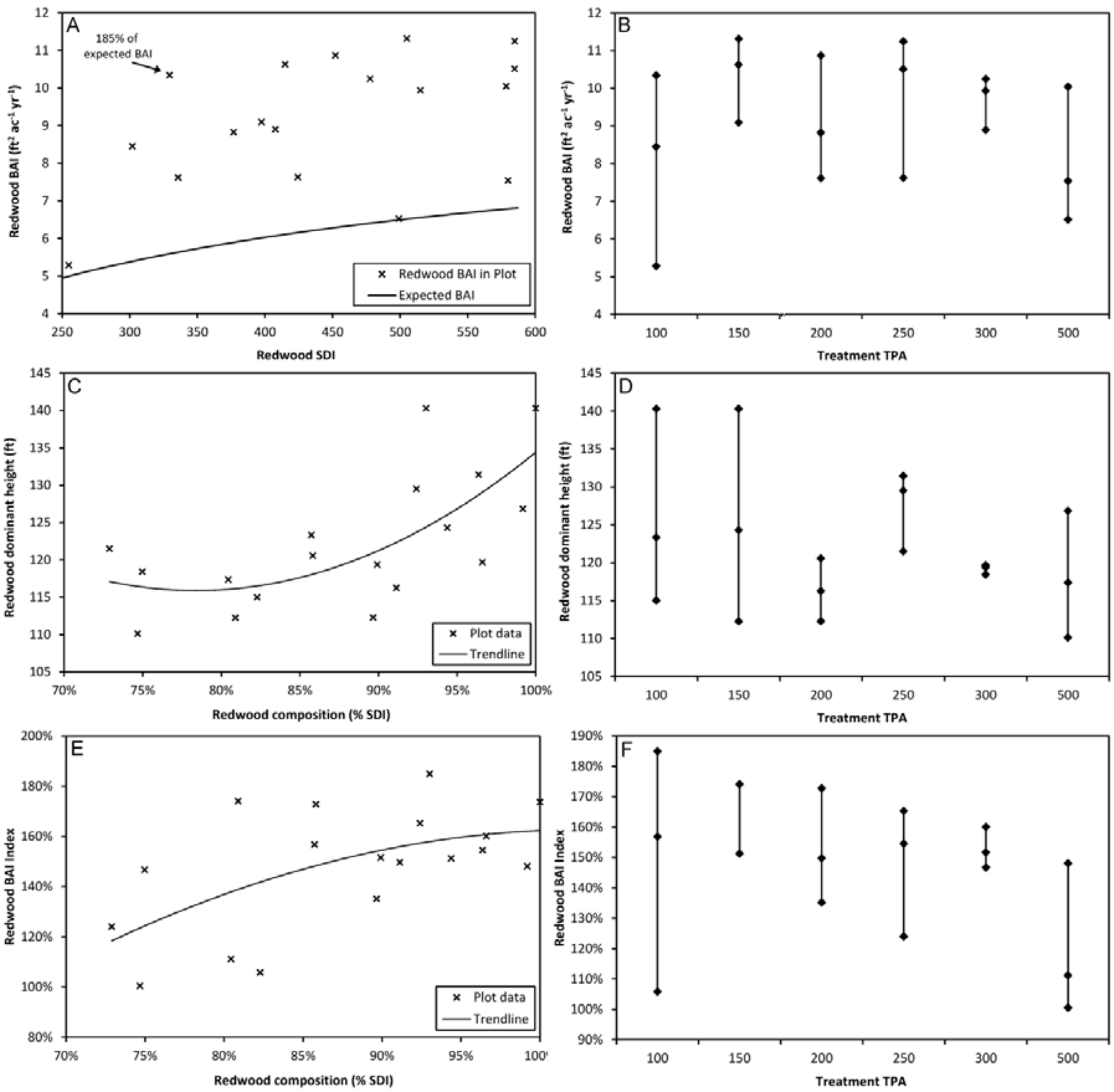

Figure 7—Plot data for redwood basal area increment (BAI) between 1998 and 2011 in PCT treatment plots, as a function of SDI for redwood component in each plot (A), and for each treatment post-thin TPA (B). Dominant redwood height (average age-49 height of 40 largest redwoods per acre in each plot) and composition in terms of redwood as a proportion of total plot SDI (C), and in each prescribed treatment in terms of post-thin TPA (D). Redwood BAI Index (RBAGI; redwood BAI indexed against expected BAI for SDI in each plot; Berrill and O'Hara 2014) and redwood composition (E) and in each PCT treatment (F).

After removing data for unthinned control plots (500 TPA prescription, actually ~900 TPA; table 1), there was no discernable treatment effect on redwood BAI between the two most recent measurements (1998 to 2011) across the range of post-thin densities (100 to 300 TPA) (ANOVA: p = 0.37). The thinning treatment effect on BAI was revealed only after accounting for differences between plots in terms of species composition and site quality. Species composition (percent redwood SDI) had a significant positive influence on redwood BAI. Plots with greater dominant redwood height also had higher BAI. The best combination of factors predicting redwood BAI in multiple linear regressions was post-thin TPA, species composition (percent redwood SDI), and site quality in terms of RBAGI (table 3).

Table 3—Comparing goodness of fit among redwood basal area increment (BAI per acre) models for experimental treatment plots between 1998 and 2011, including candidate predictor variables

Candidate models[a]	R^2	AIC	ΔAIC
$BAI=\beta_0+\beta_1 TPA$	0.062	-26.89	-
$BAI=\beta_0+\beta_1 TPA+\beta_2 RW\%$	0.344	-30.24	-3.35
$BAI=\beta_0+\beta_1 TPA+\beta_2 DomHT$	0.403	-31.68	-1.44
$BAI=\beta_0+\beta_1 TPA+\beta_2 RW\% + \beta_3 DomHT$	0.470	-31.46	0.22
$BAI=\beta_0+\beta_1 TPA+\beta_2\ RBAGI$	0.934	-64.61	-33.15
$BAI=\beta_0+\beta_1 TPA+\beta_2 RW\% + \beta_3 RBAGI$	0.944	-65.24	-0.63
$BAI=\beta_0+\beta_1 TPA+\beta_2 RW\% + \beta_3 RBAGI + \beta_4 DomHT$	0.949	-64.63	0.61

[a] TPA = post-thin trees per acre in each plot, RW% = redwood as a proportion of total plot SDI, DomHT = average age-49 height of 40 largest redwoods per acre in each plot, RBAGI = redwood basal area growth index, the actual plot BAI for all redwood indexed against expected BAI for SDI in each plot (Berrill and O'Hara 2014). Smaller AIC is better.

Site Quality Gradient Across Site

Lindquist calculated traditional site index (Lindquist and Palley1963; base age 100) for each block, finding significant difference in site quality among blocks. Analyzing site index calculated using the same equation applied to the 1998 height data with a two way ANOVA with block and PCT TPA found marginally significant differences in site quality among blocks (p = .053). Two way ANOVA of site index calculated using Wensel and Krumland (1986) equations with base age 50 years for 2011 data detected differences among blocks (p = .022) but not PCT treatments. The north block had higher site than the other blocks according to all aforementioned site index estimates and the index of redwood BA growth (fig. 8).

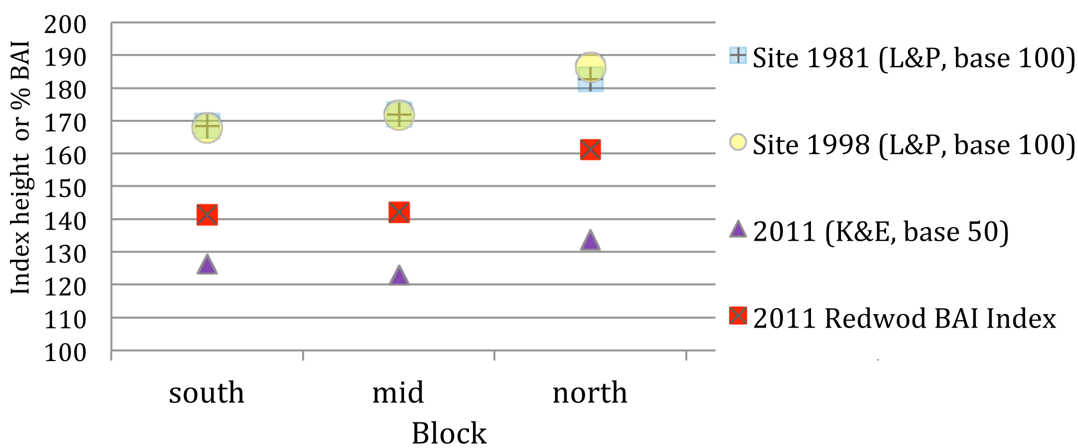

Figure 8—Comparison of the site index tabulations from stands at 19 and 36 year-old (Lindquist, 2004) as well as from 49 year-old stand. 1981 and 1999 calculations used Lindquist and Palley (1963). Wensel and Krumland (1986) and RBAGI (redwood BAI index) used 2011 data.

Discussion

As the stand age approached 50 years old, some trends from the stand establishment phase persisted and new trends were emerging. The significant differences among treatments continued for DBH and for replicate block's site quality. The north block's history of burning and resultant stimulation of *Ceanothus* appeared to have impacted the development of regenerating Douglas-fir seedlings, reflected in a significantly higher 'site quality' (i.e., productivity) for redwood. It was tempting to conclude that more rapid DBH development after heavier thinning appeared to compensate for the BA reduction with enhanced BA development in some thinned plots (fig. 3). Stand BA in the 150 and 250 TPA treatments appeared to “catch up” to BA in plots with the highest densities. However, the reduced BA development in 100 and 200 TPA is consistent with the expectation that reducing stand density by thinning reduces growing stock and therefore stand growth and yield in the near term. More likely was that BA growth was influenced by site effects that differed among plots and blocks.

The redwood BAI evaluation helped tease apart the influences of site, species, and density on productivity. By focusing only on per-acre BA growth of the redwood component (RBAGI) we bypassed the confounding influence of differences in species composition among plots. The RBAGI index of site quality also negated the confounding influence of stand density in each plot which can also obscure differences in site quality. This helped reveal differences in site quality between experimental plots thinned to different densities decades earlier. There was general agreement between RBAGI and redwood dominant height (equivalent to site index) which both indicated that site quality was poorer in the unthinned controls, and varied widely among plots. Agreement was also found when plots with higher proportions of redwood had both higher RBAGI and site index. This correlation between site quality and species composition had been suggested in previous studies (Lindquist 2004, 2007). The per-acre basal area growth of redwood in each plot was best modeled by the combination of thinning TPA, proportion of growing space used by redwood, and RBAGI. That RBAGI was a better predictor than site index is consistent with a different study on JDSF (Berrill and O’Hara 2014) which also demonstrated the utility of the RBAGI to account for the inherent variability in natural redwood stands not related to site index (Berrill and O’Hara 2014, 2016). In both studies, this disagreement between RBAGI and site index was found in plots where redwoods were relatively short but had above-average BA growth and vice versa. Site quality metrics can change over relatively short distances in redwood forests (Berrill and O’Hara 2012, 2015). Though the north block was only 91.4 m (300 ft) from the middle block, the effects of early stand management (i.e., control burning) may be influential as there was no apparent difference in abiotic factors that influenced site quality nor did aerial photos of the prior stand show differences that would explain the significant influence of site quality (in terms of site index and RBAGI) on growth of the redwood component decades after PCT. The weak correlation between site index and RBAGI indicated that height and BA growth proceeded somewhat independently, suggesting they may be controlled by a different suite of biophysical variables (Berrill and O’Hara 2016).

The most striking finding from this remeasurement was that thinning intensity had a lasting impact on average redwood DBH over 3 decades. Focusing on the 140 largest TPA revealed significant, long-lasting, effects of PCT intensity on BA and cubic foot volume. Prior analyses of earlier remeasurements of these plots (Lindquist 2004, 2007) had not identified a thinning effect on unit area basis.

Returning to the original intent of the study; it provided useful observations of the response of a precommercially thinned naturally regenerated redwood forest. When densities above 250 TPA are retained, a smaller proportion of the total volume is present in larger trees (69 to 86 percent crop trees as a percentage of total Bf). This means that a greater fraction of the stand capacity is being used by trees that may not reach the canopy and that provide a less efficient economic return because of their small size. Conversely if management goals were cubic foot carbon stock in the short term (stand less than 50 year old), the lowest PCT retention of 100 TPA did not fully utilize the growing space (fig. 6,) until age 30. Given that specific management goals may change as stands develop, choosing a thinning density in the 150 to 250 TPA range will provide for future flexibility. The unthinned control plots provide an example of when thinning is not undertaken or foregone in lieu of commercial harvest at/after age 50. In this case, even growth of the largest 140 TPA was negatively impacted and board foot volume production suffered.

The study site was one of few locations where PCT in redwood forests has been studied long-term. Maintaining this study will allow for another decadal measurement (year 2021). Remaining questions include; whether the lower stocked (TPA) plots will increase growth rate as they fully occupy the growing space, and how higher density plot growth rate and mortality will proceed given that SDI is now in excess of 600 (fig. 6).

Conclusions

Precommercial or early thinning focused growth on fewer TPA which grew larger up to the latest measurement at age 49 years. Stand production was not proportional to post-thinning density (TPA) alone. Stand history, species mix and site quality were important. Stand production was higher in certain plots and blocks with higher site quality and proportionally more redwood. PCT enhanced development of dominant trees which are expected to be important stand components for both economic and restoration goals.

Given the confounding factors of site history, site quality, and species mix, the study did not identify one optimum density for precommercial thinning. Moderate thinning (150 to 250 TPA retained) did provide reasonable individual tree growth, utilization of growing space, and options for future management. Control plots showed that deferring thinning resulted in ongoing mortality and smaller tree size than any of the PCT treatments.

Acknowledgments

Long term studies do not happen without the commitment of many people through time. The 2011 remeasurement by the Green Diamond Resource Company inventory team was a valuable contribution which included surveying to obtain more accurate plot dimensions and stem mapping, and was greatly appreciated. Thanks are due to Jeff Leddy, Demonstration State Forest Biometrician, for volume tabulation. Lastly but not least, thank you to Jim Lindquist and the past and present staff at Jackson Demonstration State Forest who maintained this study through the decades.

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Growth of Coast Redwood and Douglas-fir Following Thinning in Second-growth Forests at Redwood National Park and Headwaters Forest Reserve[1]

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Abstract

Managers of second-growth forests at Redwood National Park and the Bureau of Land Management's Headwaters Forest Reserve encourage the development of late seral forest characteristics using mechanical thinning, where competing vegetation is removed to promote growth of residual trees. Yet the ability to quantify and reliably predict outcomes of treatments such as these is hindered by the long time scales at which forests respond to thinning. Here we present analyses of tree growth at Redwood National Park (RNP) and Headwaters Forest Reserve (HDWT) from sites that have had > 5 years to respond to thinning treatments.

Compared to untreated stands, thinned stands had lower stem density (trees ha^{-1}) and basal area (m^2 ha^{-1}), primarily due to removal of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). Individual tree growth (basal area increment, BAI, m^2 yr^{-1}) was related to tree size (basal area, m^2) and treatment history, with the highest growth rates observed in large trees. Both redwood (*Sequoia sempervirens* (D. Don) Endl.) and Douglas-fir appeared to have a small, but detectable, positive growth response to thinning treatments. Early results suggest a large degree of variation among sites, with possible systematic differences in growth responses between RNP and HDWT. Future work will focus on identifying site-level differences (site quality, local competition, slope, aspect, stand age, distance from the ocean) to improve our understanding of the growth response.

Introduction

Accelerating structural development of second-growth forests is one of the primary challenges to conserving coastal redwood (*Sequoia sempervirens* (D. Don) Endl.) ecosystems at Redwood National Park (RNP) and the Bureau of Land Management's Headwaters Forest Reserve (HDWT). The need for action is clear; well over half of the coastal redwood forests at the RNP and HDWT have had a history of logging within the past 60 years, resulting in second-growth stand structures that typically impede the recovery of old-growth characteristics (O'Hara et al. 2010, Teraoka and Keyes 2011). Second-growth stands are commonly comprised of dense, even-aged Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and redwood stump sprouts, with simple canopy structure and little understory development. Under these conditions the relatively shade-intolerant Douglas-fir is expected to exclude redwood from the upper canopy until large gaps are formed, a process that may take centuries (Thornburgh et al. 2000). Moreover, many of these young second-growth stands are believed to provide poor habitat for old-growth dependent wildlife species and be vulnerable to disturbance in the form of drought, disease, and fire.

The primary management tool for forest restoration in this system is mechanical thinning, where competing vegetation is removed to improve the growth of residual trees. However, understory

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thinning prescriptions used in earlier treatments may not always encourage redwood dominance (Teraoka and Keyes 2011). Additionally, earlier thinning prescriptions have often created spatially uniform tree spacing (unlike old-growth forests; Dagley 2008, van Mantgem and Stuart 2012). Gaining better information on the effectiveness of thinning operations is difficult, as decades are needed to directly observe the results of these treatments.

Both prospective (projecting forest conditions forward in time) and retrospective (looking backwards in time) analyses are needed to determine how well current thinning treatments are meeting management objectives. Prospective analyses estimate potential long-term responses to thinning using models of forest tree growth (Dixon 2002, van Mantgem and Das 2014). Retrospective analyses consider the effects of past treatments to empirically determine if thinning has actually enhanced growth of residual trees. Long-term information concerning individual tree growth responses to thinning treatments is relatively limited.

Here, we present analyses of tree growth at RNP and HDWT from sites that have had > 5 years to respond to thinning treatments. We also document the differences in thinning responses among stands. If thinning responses are highly variable, it suggests that managers may want to tailor thinning prescriptions to individual sites, precluding simple “one-size-fits-all” prescriptions.

Methods

Study Sites

The region containing RNP and HDWT features a Mediterranean climate, with mild, rainy winters and cool, dry summers (Sawyer et al. 2000). Annual mean temperatures are approximately 15 °C (59 °F), with annual precipitation of about 170 cm (67 inches), mostly occurring as winter rain. Summer fog is common near the coast, moderating the dry summer conditions. Soils are primarily derived from sandstone, mudstone and schist. Historically, fire has shaped coastal redwood forests (Lorimer et al. 2009), but has been largely excluded over the past 100 years (Ramage et al. 2010). The general condition of forest vegetation in RNP and HDWT is a mix of second-growth forest and areas of unmanaged old-growth with no history of logging. Several second-growth stands have been thinned, with thinning prescriptions calling for removal of Douglas-fir, with redwood exempted from removal.

We surveyed second-growth forests that have undergone experimental thinning, where stands have had > 5 years to respond to thinning treatments (30 plots; 23 thinned, 7 unthinned). At RNP several sites meet this criteria, locally known as the “Whiskey 40” and “Holter Ridge” sites. The “Whiskey 40” is a 16 ha area of second-growth forest embedded within old-growth forest. The site was logged in 1963 and features extremely dense stands, consisting of Douglas-fir, redwood sprouts and species seeded in the site following logging (Sitka spruce (*Picea sitchensis* (Bong.) Carrière) and Port-Orford cedar (*Chamaecyparis lawsoniana* (A. Murray) Parl.). A 14 ha area was thinned in 1995, removing all trees ≤ 11.4 cm (4.5 inches) diameter at breast height (DBH; 1.37 m or 4.5 ft) and exotic conifers of all sizes (single-entry treatment) (Teraoka and Keyes 2011). The remaining 2 ha area at the “Whiskey 40” was left unthinned. The nearby “Holter Ridge” site was originally logged in the early 1950s (Chittick and Keyes 2007). In the fall and winter of 1978 and 1979 approximately 80 ha of second-growth forests at Holter Ridge were experimentally thinned, with treatment intensity intended to create 10 m spacing between stems. The “A972” stand was originally harvested in 1968, followed by broadcast burning and aerial seeding. The “A972” was subjected to a range of experimental thinning treatments in 2007, with overstory thinning at 55 percent or 20 percent basal area removal, or understory thinning with 55 percent or 20 percent basal area removal. At HDWT sites were initially logged in the early 1990s. We do not have records on post-harvest treatments. Thinning treatments were executed in 2004. The HDWT thinning treatments were designed to create 4.3 m (14 ft) spacing among residual stems, with coast redwood exempted from cutting. We used untreated second-growth stands in or adjacent to the “Whiskey 40”, “Holter Ridge”, and “A972” sites as comparison “control” sites to assess thinning effects.

Field Measurements

In 2008 and 2009 study plots (average plot area = 0.16 ha [0.4 ac]) were established in thinned and unthinned second-growth stands at RNP and HDWT. In all plots, we tagged all individual trees, recording species identity, clump affiliation if a stem was part of a stump sprout (to separate genetic from environment effects between redwood sprout clumps), stem diameter measured at the tag (generally DBH), tree height, live crown base height, crown class, canopy condition and noted obvious injuries (bear damage is common in this area). We conducted these surveys again in 2013, 2014 and 2015, searching for any new trees (recruits) that are above the plot minimum stem diameter (20 cm [7.9 inches] DBH) and documenting mortalities of existing trees. Radial growth (mm yr^{-1}) and basal area increment (BAI, m^2 yr^{-1}) for individual trees at each plot was determined from repeated measurements of stem diameter.

Errors in repeated diameter measurements may arise due to inaccurate measurements, bark sloughage, and/or stem moisture. Errors in growth were defined from existing data in mixed conifer forests of the Sierra Nevada of California, as data specific to coast redwood stands are absent. Likely errors in growth measurements were identified as from a single 1 ha plot measured twice in a single year in a mixed conifer forests (primarily white fir, *Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.) in Sequoia National Park. Absolute discrepancies (AD) in the diameter measurements were modeled as a linear function of DBH, best fitted as AD = 0.0035 × DBH + 0.2032. For the lower growth limit, we removed negative growth measurements where the second diameter measurement was more than four times smaller than the potential measurement error given the AD relationship. To identify the upper limit of growth, we fit a gamma distribution to 243,679 annual growth ring measurements from common species collected in mixed conifer forests of the Sierra Nevada. From the gamma distribution, only approximately 1 percent of annual ring widths were > 6 mm yr^{-1} (0.2 inches yr^{-1}) which was used as our upper growth limit. Observations of radial growth beyond the upper and lower limits were likely due to measurement or data recording errors and were removed from analysis. Using these criteria, we removed 135 of 893 redwood (15 percent of our sample) and 85 of 1452 (6 percent of our sample) Douglas-fir from analyses of growth.

Analyses

We calculated differences in forest structure (stand density, trees ha^{-1}; basal area, m^2 ha^{-1}) and growth (BAI) in thinned and unthinned second-growth stands. We also compared average stem size distributions in thinned and unthinned second-growth stands, using the Gini coefficient.

We used linear mixed models (LMMs) (Gelman and Hill 2007) to determine differences in BAI among individual trees, while accounting for plot-level differences. We considered only species where we had sufficient redwood and Douglas-fir data. For each species our model estimated radial growth for individual trees as BAI_{ij} of tree *i* in plot *j* as:

$$BAI_{ij} = \beta_0 + a_j + (\beta_{BA} \cdot BA_{ij}) + (\beta_{Treatment} \cdot Treatment_{ij}),$$
$$a_j \sim N(0, \sigma^2)$$

where BA is the individual tree stem basal area (m^2) at the first measurement, and Treatment is an indicator variable for stand treatment history (control or thinned). The BAI values were log transformed to better fit model assumptions. The variable a_j represents plot-level variations in the regression intercept. We considered varying-intercepts with varying-slope models, but these formulations did not improve model performance. We evaluated improvements to this model, including interaction terms using the Akaike Information Criterion adjusted for sample size (*AICc*) (Burnham and Anderson 2002), with differences in *AICc* (Δ*AICc*) > 4 used as evidence of substantial model dissimilarity. We calculated averaged estimates for models with similar amounts of support (Δ*AICc* ≤ 4) (Grueber et al. 2011). The proportion of variation explained using the individual-level variables only (marginal R^2) and the combined individual- and plot-level variables (conditional R^2) of the fitted models were calculated following Nakagawa and Schielzeth (2013). Analyses were conducted using the R statistical language (R Development Core Team 2015) with the 'lme4' (Bates et al. 2015) and 'MuMIn' (Barton 2015) packages.

Results

Relative to the untreated 'control' stands, thinned stands had 43 percent lower stand density (trees ha^{-1}) and 38 percent lower basal area (m^2 ha^{-1}) compared to the untreated control stands. These differences were maintained by the second census, where thinned stands had 37 percent lower stand density and 29 percent lower basal area compared to the 'control' stands. Average basal area growth for redwood between the first and second censuses in the 'control' plots was 2.0 m^2 ha^{-1}(8.7 ft^2 ac^{-1}) (SE = 0.4) while in the thinned plots redwood basal area growth was 4.8 m^2 ha^{-1} (20.9 ft^2 ac^{-1}) (SE = 0.9). Over the same interval, basal area growth of Douglas-fir was 2.2 m^2 ha^{-1} (9.6 ft^2 ac^{-1}) (SE = 0.7) in the 'control' plots and 4.1 m^2 ha^{-1} (17.9 ft^2 ac^{-1}) (SE = 0.5) in the thinned plots. Inequality measures suggest similar relative representation of small trees in the thinned and unthinned stands at the first census (Thinned average Gini = 0.18, Control average Gini = 0.19), which remained essentially unchanged by the second census (Thinned average Gini = 0.19, Control average Gini = 0.20) (fig. 1).

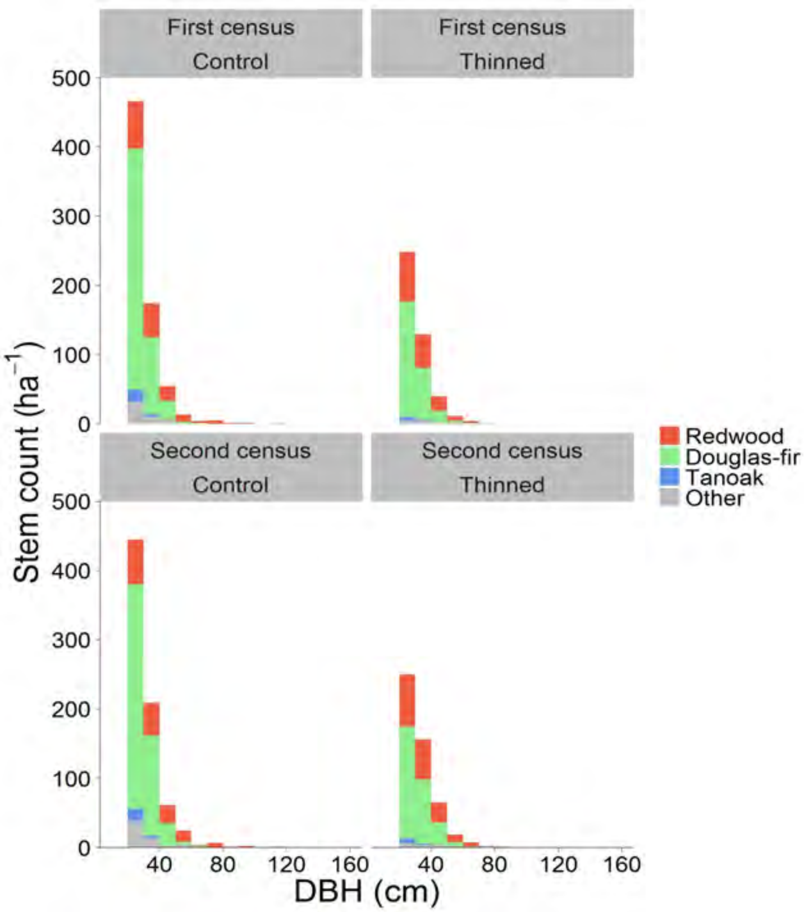

Figure 1—Stem diameter size class distributions in unthinned ('control') and thinned plots during the first (2008 or 2009) and the second census (2014 or 2015).

The LMM results showed that the BAI varied by tree size (BA) and history of thinning, with varying average responses between redwood and Douglas-fir (table 1 and table 2). For both species BAI was primarily related to individual tree size (BA), with larger trees showing higher BAI. The LMM model averaged parameter estimates suggested that Douglas-fir BAI increases faster with increasing BA compared to redwood.

Table 1—Model selection for LMMs of growth for common conifers, including terms for tree size (BA, stem basal area) and history of thinning (Treatment = unthinned or past thinning treatment) using *AICc*; evidence for substantial model dissimilarity was Δ*AICc* > 4)

Species	Model predictors	*AICc*	Δ*AICc*
Redwood	BA * Treatment	846.2	1.65
	BA + Treatment	844.5	0
	BA	849.6	5.06
	Treatment	1092.6	248.1
Douglas-fir	BA * Treatment	277.2	0
	BA + Treatment	282.9	5.67
	BA	287.5	10.25
	Treatment	1211.6	934.36

Table 2—LMM parameter estimates for individual-level effects for growth of common conifers (we used average parameter estimates for redwood, where two models had similar levels of support by *AICc*; BA refers to individual stem basal area (m^2); Treatment refers to stand treatment history, with unthinned control stands used as the reference condition)

Species	Individual-level effect	Estimate	Std. error	95 % CI
Redwood	BA	2.97	0.19	2.60 to 3.34
	$\text{Treatment}_{\text{Thin}}$	0.26	0.10	0.07 to 0.45
	$\text{BA*Treatment}_{\text{Thin}}$	0.22	0.35	-0.47 to 0.90
Douglas-fir	BA	7.73	0.31	7.07 to 8.36
	$\text{Treatment}_{\text{Thin}}$	0.30	0.09	0.12 to 0.47
	$\text{BA*Treatment}_{\text{Thin}}$	-1.09	0.39	-1.88 to -0.33

Thinning treatments were associated with slightly higher BAI, which was potentially more pronounced in large trees for redwood (i.e., a weak but positive BA*Treatment interaction term), but in small trees for Douglas-fir (i.e., a negative BA*Treatment interaction term). Thinning treatments appeared to have similar effects on both redwood and Douglas-fir (CIs for thinning treatments overlapped between species), with redwood perhaps showing slightly stronger response to thinning treatments than Douglas-fir (fig. 2). We found similar effects of thinning when considering radial increment predicted by stem diameter and treatment. The individual-level effects of the model explained a relatively large amount of variation in tree mortality for both redwood and Douglas-fir (marginal $R^2 > 0.31$), though the inclusion of the plot-level effect (plot identity) improved model performance (conditional $R^2 > 0.45$). The models that included thinning effects only explained a small, but non-zero, amount of variance (marginal $R^2 > 0.02$, conditional $R^2 > 0.07$). Estimated random effects suggest high model intercepts for plots in HDWT for both redwood and Douglas-fir, potentially indicating higher growth rates in HDWT relative to RNP.

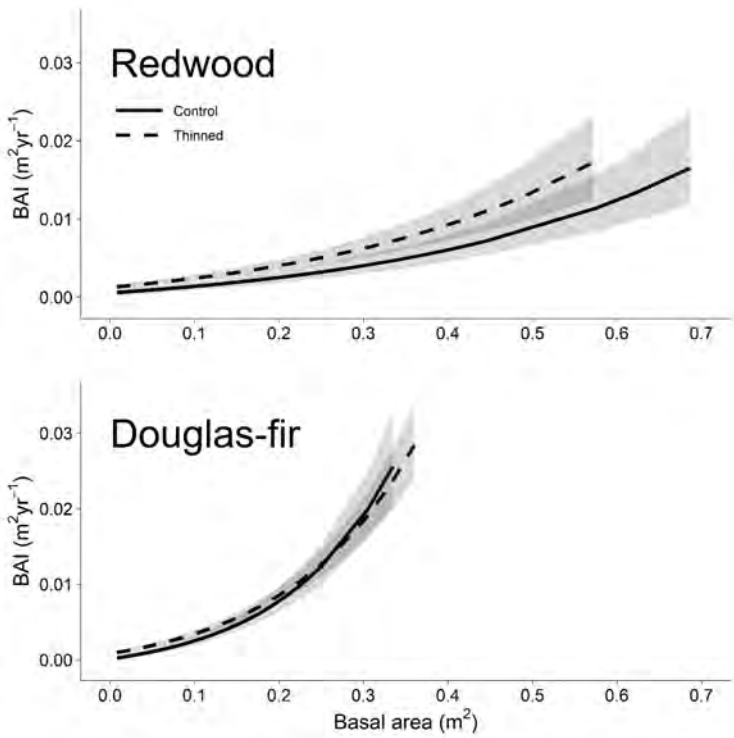

Figure 2—Predicted individual tree growth in thinned and unthinned plots. The heavy lines represent modeled average trends and the shaded band represents 95 percent bootstrapped confidence intervals from uncertainty in the individual-level parameter estimates.

Discussion

Our results suggest that redwood growth continues to be favorable > 5 years following thinning treatments at RNP and HDWT. Residual Douglas-fir also responded positively to thinning treatments, but the stand structures created by these treatments (retention of redwood) likely allowed for enhanced redwood basal area accumulation at the stand level. A more rigorous examination of plot-level thinning effects will require experimental data, rather than the post-hoc results presented here. However, these observations are in agreement with earlier assessments of coastal redwood forest stand development (Teraoka and Keyes 2011), which found that unthinned areas will be very slow to recover redwood dominance. As stand development continues, how long these treatments will remain effective at encouraging redwood growth is not known. Second-entry thinning treatments may be possible in some areas, but funding and the removal of old logging roads as part of an overall restoration treatment following first-entry thinning may limit these operations.

Growth at the individual tree level provides an indication of the range of response that Douglas-fir and redwood may have to thinning treatments. Here, we found thinning to have small, but measurable, effects on basal area growth for redwood and Douglas-fir. Between these species, the growth potential appears to be greater for Douglas-fir compared to redwood in our stands. This confirms earlier findings (using data from some of the same plots considered here) that on a per-individual basis, under current second-growth conditions at our sites Douglas-fir appears to be a better competitor relative to redwood, but that redwood may be better able to exploit growing conditions created by aggressive thinning treatments (e.g., 40 percent stand basal area reductions) (van Mantgem and Das 2014). Next steps include examining height and volume growth, which will be essential to gain a broader perspective of treatment effects.

Our present assessment of thinning effects on individual tree growth would be improved with spatially-explicit data. Individual-level tree growth is strongly influenced by local conditions, which may or may not reflect plot-level averages. But these data are relatively time-consuming to collect and a critical assessment is needed if spatially-explicit models (e.g., van Mantgem and Das 2014) offer a substantial improvement in projecting stand conditions relative to traditional non-spatially explicit models of forest growth (e.g., FVS, CRYPTOS). Comparing estimates from spatial and non-spatial models will help determine if inclusion of this additional information (e.g., clumped, random, or uniform spacing; interior or gap-edge location) strongly influences predictions of residual tree growth.

The relatively large magnitude of plot-level differences identified from the linear mixed models of tree growth suggest that growth responses to thinning may vary substantially among sites. Identifying the environmental factors (e.g., slope, aspect, stand age, distance from the ocean) that contribute to variation in thinning responses will be an important future step. Thinning responses will likely also be strongly correlated to site quality (Berrill 2008), which could also be included in future studies. Collecting data across different thinning intensities, patterns, and treatment frequencies simulated for different site types will demonstrate how site conditions and thinning treatments interact.

Restoring young forests is a key component of coastal redwood forest conservation, not only to accelerate the development of old forest structure, but may also sufficiently reduce competitive pressures among remaining trees so that they may be more resistant (more likely to survive) when faced with environmental stressors, such as drought. Though it is still unclear if thinning treatments will confer resilience to disturbance in coastal redwood forests, observations from other forest types are promising (D'Amato et al. 2013, Fulé et al. 2012). This presumed benefit from restoration thinning may become an increasingly important consideration in an era of climate change.

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Ecosystem Responses to Variable-Density Thinning for Forest Restoration in Mill Creek[1]

Lathrop P. Leonard,[2] John-Pascal Berrill,[3] and Christa M. Dagley[3]

Background

Variable-density thinning (VDT) has promise as a forest restoration tool that accelerates development of old-growth redwood (*Sequoia sempervirens* (D.Don) Endl.) forest characteristics (O'Hara et al. 2010) but can lead to bear damage in north coastal California (Hosack and Fulgham 1998, Perry et al. 2016). Three novel VDT prescriptions (O'Hara et al. 2012) were tested across an extensive area at the Mill Creek addition of Del Norte Coast Redwoods State Park, near Crescent City, in Del Norte County, California. This area is primarily composed of young, crowded forests dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) regenerating after a history of industrial forest management. These forests were once dominated by large, widely-spaced redwood and Douglas-fir, and are located in a watershed that plays an important role in protecting old-growth forest located downstream in Jedediah Smith Redwoods State Park. The even-aged stands prioritized for VDT had been regenerated between 1982 and 1992, and prior to treatment had around 1300 stems ha^{-1} averaging 15 cm diameter at breast height (DBH; 1.37 m) of which 2/3 were Douglas-fir.

Each stand was assigned one of four experimental prescriptions replicated five times throughout the ownership and monitored in large sample plots. Three plots were established in each stand soon after treatment (within 1 year of the treatment date). All plots were then re-measured 4 years after establishment. We summarized tree- and stand data from the 60 monitoring plots in 20 stands receiving one of four treatments: low-density thin to 6.1 m spacing (LDT), high-density thin to 4.9 m spacing (HDT), localized release (LR), and no-thin control (C). Specifically, we calculated averages for each monitoring plot, and compared these averages among the four treatments. Our objectives were to compare the effectiveness of VDT treatments at promoting redwood dominance, redwood tree growth, and stand structural complexity. We also compared incidences of bear damage and depth of slash following each thinning treatment and over the same time period in unthinned control stands.

Structural (Tree-Size) Diversity

After treatment, trees of all species combined were tallest and largest on average (10.2 m height (HT); 17.5 cm DBH) after HDT. Trees in LR stands (9.55 m HT; 16.8 cm DBH) were taller but slightly smaller in DBH than the LDT stands (9.26 m HT; 16.9 cm DBH). Trees in unthinned control stands were smallest on average (6.87 m HT; 14.8 cm DBH). Structural diversity in terms of tree-size variability after thinning was greatest after LR, where standard deviation of tree DBH in each plot averaged 6.32 cm. LDT and HDT had equivalent, intermediate levels of tree-size variability (s.d. 5.86 and 5.90 cm, respectively), and controls the lowest average s.d. of tree DBH (4.90 cm).

Species Composition

Proportion of stand basal area (BA) in each species gave tree species composition after treatment. Redwood represented 29 percent of stand BA in unthinned control stands, 37 percent after HDT, 44

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percent after LR, and 46 percent of stand BA after LDT. Douglas-fir represented 66 percent of stand BA in unthinned control stands, 60 percent after HDT, 50 percent after LR, and 48 percent of stand BA after LDT.

Tree Diameter Growth

From the 8366 tree records for all tree species in all plots, 1696 tree records were excluded from growth (DBH and BA increment) calculations. Excluded were trees with damaged tops, bear damage, and/or missing or estimated DBH data. Among 2282 tree records for redwood trees without damage, the average redwood tree DBH increment over 4 years after treatment ranged from 0.42 cm yr^{-1} in unthinned controls up to 0.73 cm yr^{-1} and ranked LR > HDT ≈ LDT > Control. Tree BA increment over 4 years after treatment ranged from 13.1 cm^2 yr^{-1} in unthinned controls up to 25.4 cm^2 yr^{-1} and ranked LR > HD > LD > Control. These undamaged redwood trees were slightly larger on average, immediately following LR and HDT treatments, than the undamaged redwood trees giving dbh increment data in LDT and Control treatments, suggesting that BA increments would give a better indication of differences in post-treatment growth than dbh increment. Tree BA growth of the 50 largest redwood/ha and 50 largest Douglas-fir/ha, presumably a big part of the future restored old-growth overstory, ranked LR > LD > HD > C for redwood and HDT > LDT > C ≈ LR for Douglas-fir where localized release favored redwood while restricting Douglas-fir growth.

Bear Damage

We noticed virtually no bear damage while overseeing the thinning and assume all bear damage happened after treatment. Most damage was recorded in the first measurement, when monitoring plots were installed, within 1 year of treatment (not immediately after). Additional damage was noted during the second assessment 4 years later. Among 3230 redwood trees in all plots sampling all treatments, 24 percent to 26.5 percent were damaged after thinning treatments whereas only 8.2 percent were damaged in unthinned control plots over the same period. More Douglas-fir were damaged after LDT (20 percent) than HDT (13.8 percent) and LR (12.8 percent), and very few were damaged in unthinned stands (1.1 percent). Bears mainly damaged redwood and Douglas-fir; damage was only noted for 19 other conifer trees (assortment of five species) and one tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh) in the sample of 8366 trees in 60 plots across all treatments.

Slash (Fuel Bed) Depth

Treatment-wide averages calculated for 4,686 measurements of fuel depth in 58 monitoring plots indicated that LR generated the deepest fuel beds (averaging 0.98 m). LDT had an intermediate fuel depth (0.85 m) and HDT the shallowest fuel beds (0.72 m), but that variability in fuels among measurement locations were equally variable under each treatment. Mortality within unthinned control plots was contributing some fuel load (0.12 m) measured after 4 years of monitoring. Fuel beds were decreasing in depth by 24 percent to 37 percent over the 4 years since treatment, with HDT exhibiting the greatest decrease; possibly due to more smaller/fewer large trees cut decaying and breaking down more quickly.

Recommendations for Adaptive Management

These results inform adaptive management, guiding ongoing restoration of thousands of hectares of crowded young forest at Mill Creek. Outcomes from each VDT treatment differed in terms of species composition, structural diversity, tree growth, incidences of bear damage, and fuel load from cut wood and debris, across the wide range of sites and stands included in this landscape-scale manipulative experiment at Mill Creek. LDT enhanced redwood tree growth and resulted in the greatest shift in species composition in favor of redwood. Tree-size development and variability were both enhanced by the LR treatment. Assuming such changes are desirable, LR appeared to garner most benefit overall by enhancing redwood dominance, accelerating redwood growth, and promoting structural heterogeneity in terms of tree-size variability (fig. 1). LR was most efficient to implement (0.79 ha/person/day) > HDT (0.56 ha/person/day) > LDT (0.53 ha/person/day), demonstrating that promoting complex stand conditions can be accomplished at a cost competitive with more traditional forest management PCT treatments that resemble HDT.

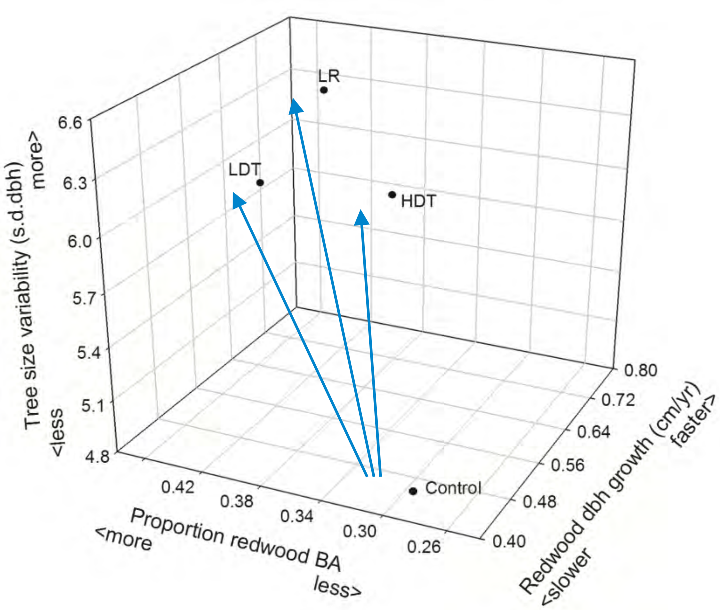

Figure 1—Stand attributes after variable density thinning compared with unthinned control stands. Averages based on data from 60 plots in 20 stands receiving one of four treatments: low-density thin to 6.1 m spacing (LDT), high-density thin to 4.9 m spacing (HDT), localized release (LR), and no-thin control stands at Mill Creek.

Variable slash accumulation averaging up to a meter depth, including thick patches and areas without slash, can be expected. The slash will settle and decay quickly, but could be treated in specific locations such as fuel breaks and strategically-placed defensible spaces. In anticipation of bear damage and loss of trees after any thinning operation, we could design thinning prescriptions where additional trees are retained to compensate for loss or damage (e.g., 25 percent extra redwood and 20 percent Douglas-fir after heavier thinning).

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Physiology and Growth of Redwood and Douglas-fir Planted After Variable Density Retention Outside Redwood's Range[1]

Lucy Kerhoulas,[2] Nicholas Kerhoulas,[3] Wade Polda,[4] and John-Pascal Berrill[2]

Abstract

Reforestation following timber harvests is an important topic throughout the coast redwood (*Sequoia sempervirens* (D. Don) Endl.) range. Furthermore, as drought-induced mortality spreads across many of California's forests, it is important to understand how physiology and stand structure influence reforestation success. Finally, as climate throughout the West is projected to become hotter and drier, it is important to investigate seedling regeneration under hotter and drier conditions, particularly for species such as coast redwood that are generally restricted to mesic habitats.

To study the influences of climate and stand structure on regeneration success, we monitored physiology and aboveground growth of coast redwood and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) seedlings during the 2015 growing season following a 2014 variable density retention harvest at the L.W. Schatz Demonstration Tree Farm in Maple Creek, California. We hypothesized that because redwood does not naturally occur this far inland, it would be more water stressed than Douglas-fir and would, as a result, grow less. We also hypothesized that seedlings planted in the moderately thinned treatment would be the least water stressed and therefore realize the most growth due to reduced competition for water and light, increased precipitation throughfall, and minimally increased evaporative loss of soil water.

We found that water stress and aboveground growth were significantly lower in redwood than in Douglas-fir. These findings suggest that greater stomatal regulation to conserve water reduces CO_2 uptake and growth in redwood when compared to Douglas-fir. Alternatively, and not mutually exclusively, redwood, a species renowned for its lignotubers and sprouting ability, may allocate more carbon belowground during seedling establishment compared to Douglas-fir. Greater belowground carbon allocation would explain our findings of decreased water stress, resulting from increased fine root production and/or mycorrhizal associations, and aboveground growth in redwood compared to Douglas-fir. We also found that seedlings of both species in our heavily thinned treatment were the least water stressed and had the highest growth compared to seedlings planted in the moderately thinned and control (not thinned) treatments.

We recommend further research on stomatal conductance and carbon allocation patterns in these two species to identify the mechanism(s) driving the decreased water stress and aboveground growth observed in this study. We also recommend heavy thinning treatments to achieve minimal water stress and maximum growth in establishing seedlings.

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SESSION 6 – Wildlife, Native Plants and Habitat

The Response of Swamp Harebell (*Campanula californica*) to Timber Harvest: a Case Study[1]

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Abstract

A perennial herb of the Campanulaceae (bellflower) family, swamp harebell (*Campanula californica*) is endemic to the north and central coast of California. The California Department of Fish and Wildlife (CDFW) considers the species to be moderately threatened, and is concerned with the severity of impacts from land use activities, and the effectiveness of mitigation measures proposed under California Environmental Quality Act compliance. A timber harvesting plan (THP) submitted during 2000 reported the species distributed in several clusters in the proposed Sonoma County logging area. During review, CDFW and the landowner agreed to evaluate the impacts of some standard timber harvest practices. Following a repeated-measures design with sampling the year prior to harvest and post-harvest years 1, 3, and 5, we enumerated swamp harebell plants in five 30 cm x 30 cm subplots systematically placed within twenty 3 m x 1 m cover-class quadrants. We situated the cover-class quadrants to assess four immediate on-site impacts: road reopening/hauling, timber falling and skidding, reducing canopy, and minimal direct impact. The number of plants on roadways declined substantially in the first year post harvest, and remained low in subsequent years. Likewise, the number of plants declined between pre-harvest and post-harvest year 1 where trees were felled and skidded, but some recovery was apparent by the final year. Reduced canopy plots did not show substantive response attributable to harvest. Of the un-impacted sites, one declined continually and substantially (to 2 percent of its original count) over study period, while the others also generally declined. Drought conditions during the monitoring period likely impacted swamp harebell numbers more than many impacts of timber harvest, other than those resulting from road construction and use.

Keywords: *Campanula californica*, drought, roads, swamp harebell, timber harvest impacts

Introduction

Timber harvest practices alter canopy, microclimate (especially temperature and humidity regimes), hydrology, and soil conditions. The associated changes in landscapes and habitats effect different plant species in different ways. Early-successional species or hardy generalists may benefit from logging-related changes, while specialists or late-successional species may be negatively impacted. Some species may experience a “mixed-bag” of effects from harvest practices. For instance, a plant species may benefit from increased solar input resulting from canopy reduction, while being negatively impacted by drying or increased competition from non-native species introduced through ground disturbance or erosion control efforts. The effects of timber harvest practices have been well studied for few plant species. For an exception, see Renner et al. 2011. Impacts to many forest-dwelling plant species remain undocumented. For example, we know of no rigorous assessment of the response of swamp harebell (*Campanula californica*) to forestry practices.

Swamp harebell is a rhizomatous perennial herb in the bellflower family (Campanulaceae). The California Native Plant Society (CNPS) (Anonymous 2016a) and the California Department of Fish and Wildlife (CDFW) (Anonymous 2016b) categorize the species as rare, threatened or endangered in

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California and elsewhere, fairly endangered in California (1B.2) and assign the degree of threat as moderate due to either 20 to 80 percent occurrences being threatened or a moderate immediacy of threat. Currently, neither the state nor federal governments list it under their respective endangered species acts. The range of *C. californica* is limited to coastal areas in Sonoma, Marin and Mendocino counties, and CNPS (Anonymous 2016a) considers it extirpated from Santa Cruz County. Swamp harebell has slender clamoring stems 10 to 30 cm long with stiff recurved hairs and thin, opposite ovate leaves. Pale blue, bell-shaped flowers may appear from June through October. Habitats for *C. californica* include bogs and fens, closed-cone coniferous forests, coastal prairie, meadows and seeps, marshes and swamps (freshwater), North Coast coniferous forest/ mesic, at elevations less than 500 m. The CNPS (Anonymous 2016) asserts threats to be competition, grazing, development, marsh habitat loss, logging, road maintenance, and trampling.

Due to limited distribution and number of plants as well as lack of knowledge regarding its response to disturbance, this small and shallowly-rooted herbaceous plant may require some form of protection from disturbance during land management activities. Based on limited observations, some forestry professionals suggest *C. californica* benefits from disturbance caused by logging practices. These claims have not been evaluated rigorously. In the absence of supporting information, the common practice for resource professionals is to avoid or minimize impacts to *C. californica* found in harvest areas. For some projects, avoidance is not always possible.

During 2000, Registered Professional Forester (RPF) Nicholas Kent submitted timber harvesting plan (THP) 1-00-321 SON for Mr. D.M. Richardson, a non-industrial private landowner. The THP reported the discovery of swamp harebell in several clusters over part of the plan area. During the review process for the plan, the landowner and forester agreed to monitor the impacts of the timber harvest on *C. californica* populations. The CDFW took the lead role in designing and implementing this effort. Because of the study's sites treatments were not randomly assigned and are not completely independent due to the THP's limited geographic area, single time-frame, and proximity of the clusters, as well as small sample size within treatments, and the limited silviculture and harvest intensity, we consider this monitoring effort a case study.

Methods

The THP was located near Horseshoe Cove in northwestern Sonoma County, approximately 0.25 km to 1.6 km from the Pacific Ocean, and 5.6 km south of the town of Stewart's Point in Sonoma County. The plan called for Selection and Alternative Prescription silviculture harvest in a coast redwood (*Sequoia sempervirens* (D.Don) Endl.), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), and Bishop Pine (*Pinus muricata* D. Don) forest mosaic. The RPF surveyed the THP area for swamp harebell in July of 2002, and provided a map to CDFW. In June of 2003, prior to harvesting operations, we examined the THP area and identified 20 clusters of swamp harebell to be monitored. The clusters of *C. californica* we selected ranged from 0.3 to 0.6 km from the ocean, 60 to 120 m elevation, and were expected to experience a range of timber operations. We chose to evaluate the impacts of the plan as proposed and did not specify operational constraints be applied to any of the *C. californica* clusters; however, two clusters where within non-operational locations.

Based on THP tree marking and flagging, we assigned clusters to three different common timber harvesting related activities based on their potential effects on swamp harebell (table 1). These treatments were: road reconstruction and use, canopy (shade) reduction, and tree falling and log skidding. Clusters where timber operations were not anticipated immediately on site based on the absence of flagging or trees with harvest marks were identified as "controls". All clusters selected for study were occupied by healthy colonies of *C. californica* during study layout. The road sites were on an existing, overgrown seasonal road that was intended to be reopened. Impacts likely on the road sites included surface grading, soil compaction, change in local surface hydrology, canopy reduction, and excavation for crossing installation. All upslope sites evidenced historical timber operations disturbance in the form of stumps and skid trails.

Table 1—Site treatment assignments and condition notes based on pre-harvest assessment of probable impacts

Road	Canopy reduction	Full operations	Controls
Rd1 - On existing seasonal road. Soil compaction from road use.	S10 - centered in swale above barely developed class III watercourse. Canopy reduction, no ground disturbance.	S3 - On historic skid trail. A tree will be dropped and skidded over the site. Canopy reduction, soil disturbance and possible compaction.	S1 - On historic skid trail; flagged as sensitive for *Lilium maritimum* population and avoided.
Rd2 - On existing seasonal road. Soil compaction from road use.	S11 - Canopy reduction, no ground disturbance.	S4 - On historic skid Site skidded over, some canopy reduction, soil disturbance and possible compaction.	S2 - On historic skid trail, area to be flagged as sensitive and avoided.
Rd3 - On existing seasonal road. Soil compaction from road use.	S12 - Canopy reduction, no ground disturbance.	S6 – Edge of historic skid trail. Site will be skidded over. Canopy reduction, soil disturbance and possible compaction.	S5 - In a swale, area to be avoided.
Rd4 - On existing seasonal road. Road widened. Grading, soil compaction from road use.	S13 – Canopy reduction to south, no ground disturbance.	S7 - On historic skid trail. A tree will be dropped and skidded over the site. Canopy reduction, soil disturbance and possible compaction.	S14 - Area within WLPZ, to be avoided.
Rd5 - On existing seasonal road in class II WLPZ. Possible excavation and widening for crossing installation. Grading, soil compaction from road use.		S8 - A tree will be dropped and skidded over the site. Canopy reduction, soil disturbance and possible compaction. Slash added to site post-harvest.	S15 - Area to be avoided.
		S9 - Site in skid trail related ditch. Site to be skidded over, soil disturbance- ditch not to be maintained- hydrography change and slash added to site post-harvest.	

During the first week of July, 2003, and prior to any timber operations, while the plants were in flower, we recorded metadata, characterized each site, and enumerated swamp harebell. Because our study design was based on permanent plots and repeated measures, each site was benchmarked at two nearby leave trees and the distance and direction to plot center recorded to enable plot relocation in subsequent years. In addition to a benchmark description, information included Site ID; GPS location; location description; local slope (recorded from 2 m upslope or up-road to 2 m downslope or down-road at mid-plot); aspect (compass); approximate elevation (contour map); canopy cover (i.e., shade, determined with a Solar Pathfinder® using the template for August, one reading at mid-plot); associated species; and qualitative characterization of tree canopy (e.g., open/filtered), soil characteristics, site moisture (saturated, mesic, or xeric), habitat, and topography. Swamp harebell at each site was characterized by cover class (coded 0 for none, 1 for < 5 percent, 2 for 5 to < 10 percent, 3 for 10 to < 15 percent) in a 1 x 3 m plot (hereafter "cover plot") oriented perpendicular to the road alignment (road plots), across draws where present, and along contour where other surface micro-topography was slight. Within each cover plot, we systematically placed five 30 x 30 cm subplots (fig. 1) in which flowering and vegetative stems were enumerated. Because the number of flowering stems was very few, this analysis sums the counts of flowering and non-flowering plants on all five subplots at each site.

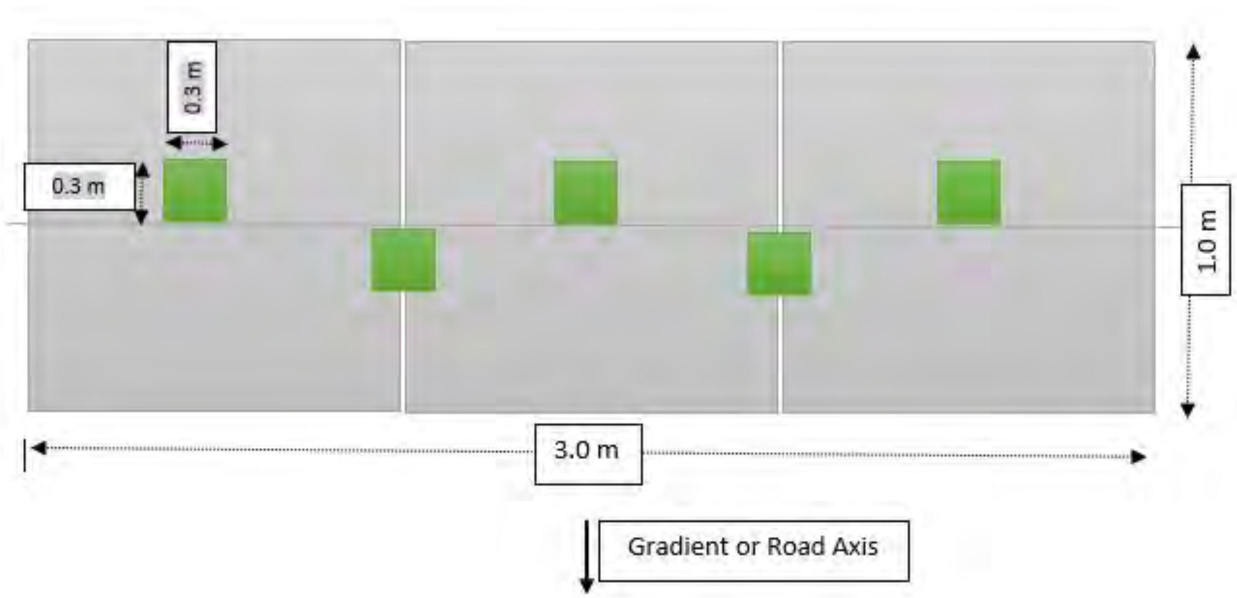


Figure 1—Layout of cover plot (1 x 3 m) and count subplots (30 x 30 cm).

Timber was harvested in 2003 immediately following the initial, pre-harvest assessment. The assessment was then repeated during early July for post-harvest years 1 (2004), 3 (2006), and 5 (2008).

Results

The solar radiation that was shaded by tree canopy was largely unaffected by the timber harvest (table 2). With the exception of one site (S8), timber operations reduced the shade among all the sites by less than 20 percent as measured the first year post-harvest (fig. 2). Within treatment categories, all the shade reduction sites experienced reduced shade in the first year post-harvest as expected, but two (S10, S13) had completely recovered by the third year post harvest. One site (S8) experiencing full operations had substantially reduced shade from pre-harvest to year 1 post-harvest, and it did not recover during the study. All other full operations sites showed stable canopy during the study period. Shade at the road sites was unaffected from pre-harvest to year 1 post-harvest and three remained stable while the forth (R2) dropped substantially during the last year. There was no apparent trend in shade canopy for the control sites.

Table 2—Mean (SD, range) percentage of potential solar radiation during August shaded by tree canopy

Assessment Year	Control (n = 5)	Road (n = 5)	Reduced canopy (n = 4)	Full operations (n = 6)
2003	80 (10, 64-89)	85 (13. 62-96)	77 (14, 63-96)	91 (5, 85-99)
2004	78 (14, 56-94)	84 (15, 60-98)	68 (16, 56-90)	82 (13, 55-90)
2006	84 (11, 67-95)	83 (12, 66-95)	79 (8, 68-87)	86 (8, 72-93)
2008	78 (14, 54-90)	82 (18, 50-94)	80 (4, 76-84)	83 (9, 67-91)

With the exception of one site (S8), timber operations reduced the shade among the sites less than 20 percent as measured during the first year post-harvest (fig. 2).

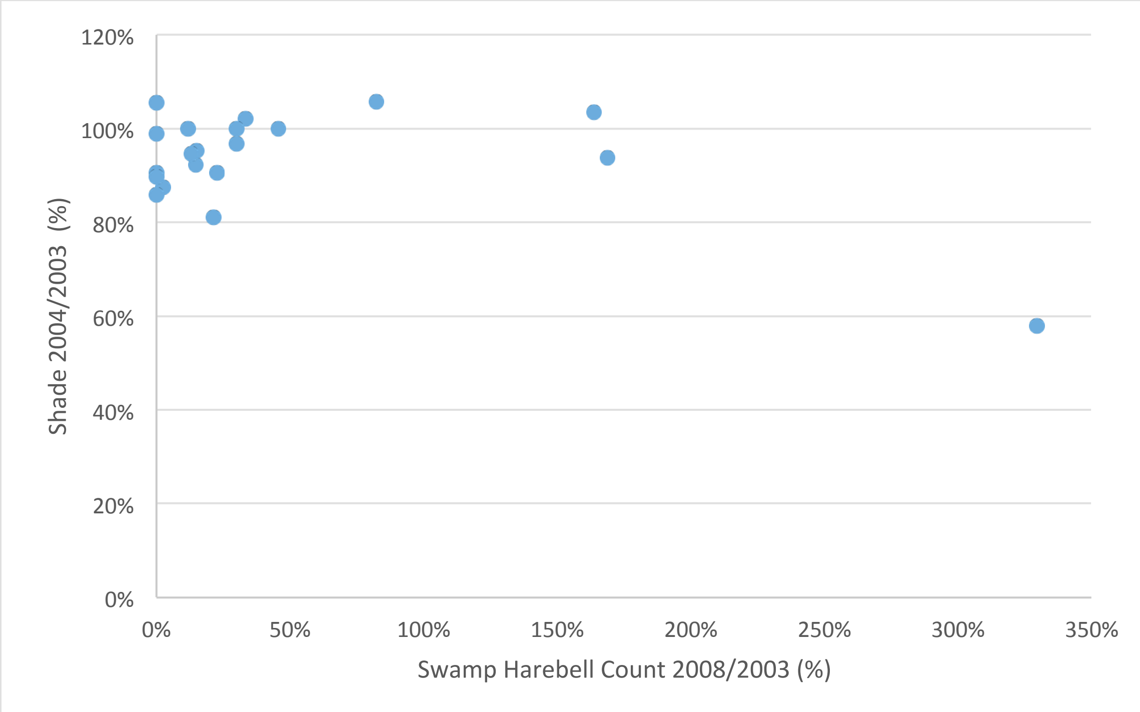


Figure 2—Percent population change from pre-harvest (2003) to year 3 post-harvest (2008) vs. change in shade from pre-harvest (2003) too post-harvest year 1 (2004).

Cover class of swamp harebell did not change from the pre-harvest to post-harvest year 1 in the shade reduction sites (fig. 3a). By the third year, all shade-reduction sites had less than 5 percent cover. The cover class was variable at sites that experienced full operations (fig. 3b). The cover class of one (S4) declined from cover class 2 pre-harvest to cover class year 1 post-harvest, and remained rank 1 for the remainder of the study. The cover class of one site (S9) was unchanged initially, but then fell to zero through the study. The greatest cover class assignment (3) was achieved during year 2 post-harvest, rising from a cover class code of 1, and then declining to a class of 2 during the final year. One site (S7) rose to a cover class 2 during the last year after three previous cover class 1 assignments. One road site (Rd 4) dropped from a cover class of 2 to 1 apparently in response to harvest, an assignment it maintained for the remainder of the study (fig. 3c). All other road sites were assigned a cover class of 1 during the entire study. Control sites (fig. 3d) showed more variability relative to the other sites, but all declined, and one (S2) was extirpated the final year.

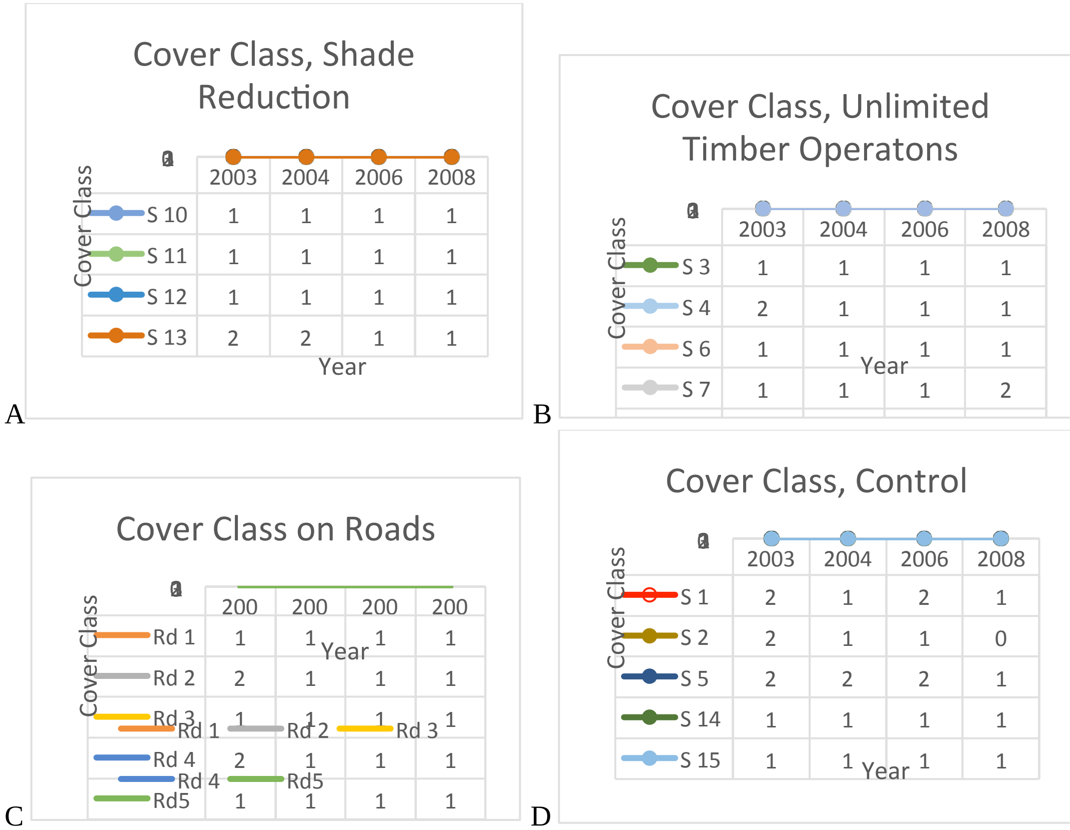
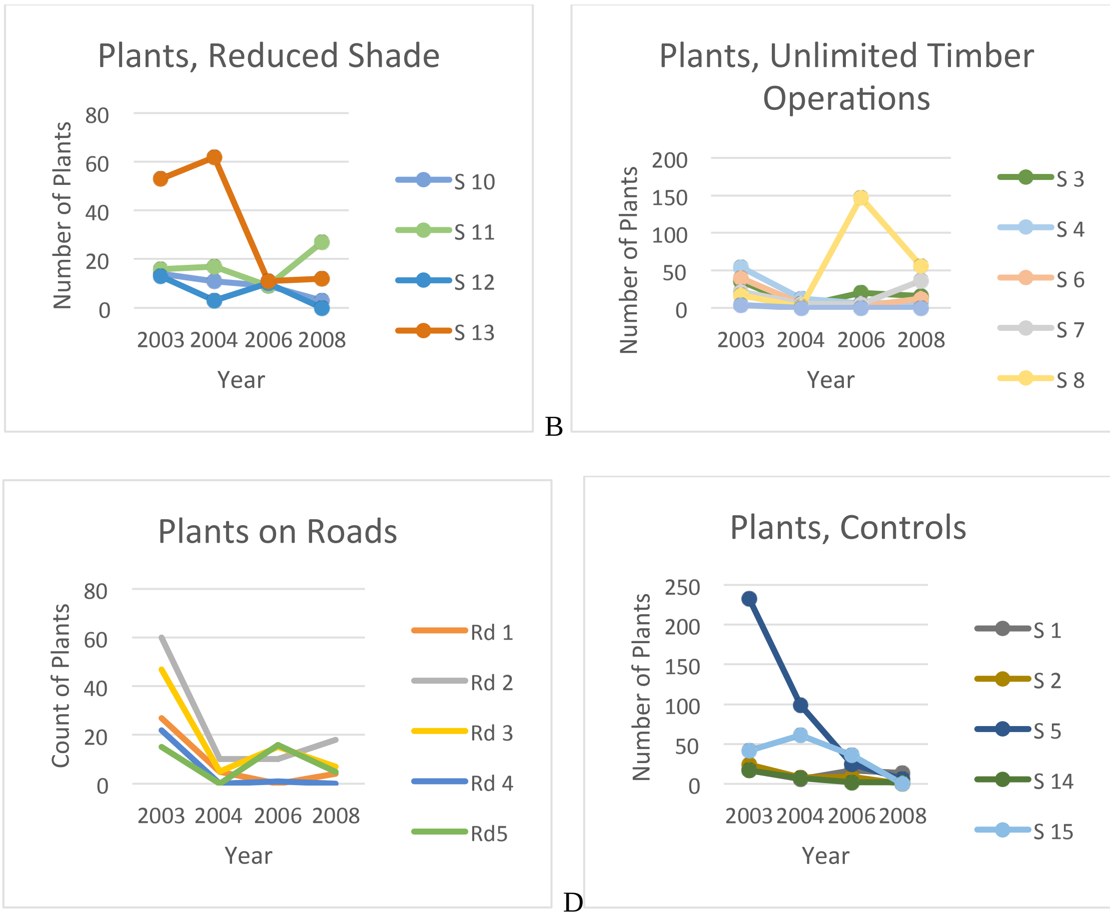


Figure 3—Cover class over time by treatment type: A shade reduction, B full operations, C road construction, D controls.

The number of plants at shade reduction sites remained stable from the pre-harvest to the first post-harvest year (fig. 4a), but one site (S13) suffered an 80 percent decline from the first to the third post-harvest year. One site (S12) was extirpated by the final year. All sites experiencing full operation (fig. 4b) declined from the pre-harvest to the first post-harvest year, and one (S9) was extirpated. However, *C. californica* numbers at one site (S8) increased more than 8-fold between the pre-harvest and the third post-harvest. Interestingly, this site also had the greatest operations-induced shade reduction (fig. 2). With the exception of the extirpated site, trends at full operation sites suggest limited recovery at all sites by the final year compared to the pre-harvest year. Swamp harebell counts at three road sites (fig. 4c) declined from pre-harvest to year 1 post-harvest by greater than 80 percent, and they were extirpated at one (Rd 4). While the species remained present at four of five of the road sites over the study period, recovery to pre-harvest numbers was not apparent. Counts declined from pre-harvest to year 1 post-harvest at three of four control sites (fig. 4-d), more than 55 percent at one (S15). Trends appear generally negative at the control sites until the end of the study, with two control sites (S2, S15) being extirpated.



A B C D

Figure 4—Plant count over time by treatment type: A shade reduction, B full operations, C road construction, D controls.

Discussion

Our study plan called for evaluating swamp harebell populations using both cover class and direct stem counts. This decision reflected the rhizomatous and clamoring growth form of the species. That is, where the species grows densely and robustly, its growth form might make direct individual stem counts so difficult that cover class becomes a more pertinent measure. However, where it grows less robustly, a direct count is not difficult. A combination of the measures, under appropriate conditions, might reveal subtle changes in vigor among sites or years. However, we found cover class to be a poor variable to assess response of harebell population size in a site. Despite being small classes (5 percent), resolution may have been inadequate. We recorded only 4 classes, one of which (class 3) was assigned at only one site and only one year. Further, we possibly mis-classed cover. To assess this possibility, we compared sub-plot counts to cover class codes (fig. 5). While there is correlation, the overlap of counts among classes is substantial. Overlap might reasonably reflect differential effects of plant vigor on the two variables, e.g., cover per plant may be greater in good rather than poor growing conditions. We cannot determine the cause of the overlap, but believe its presence underscores the importance of training and quality control to achieve repeatable and accurate measures of cover in long-term monitoring projects.

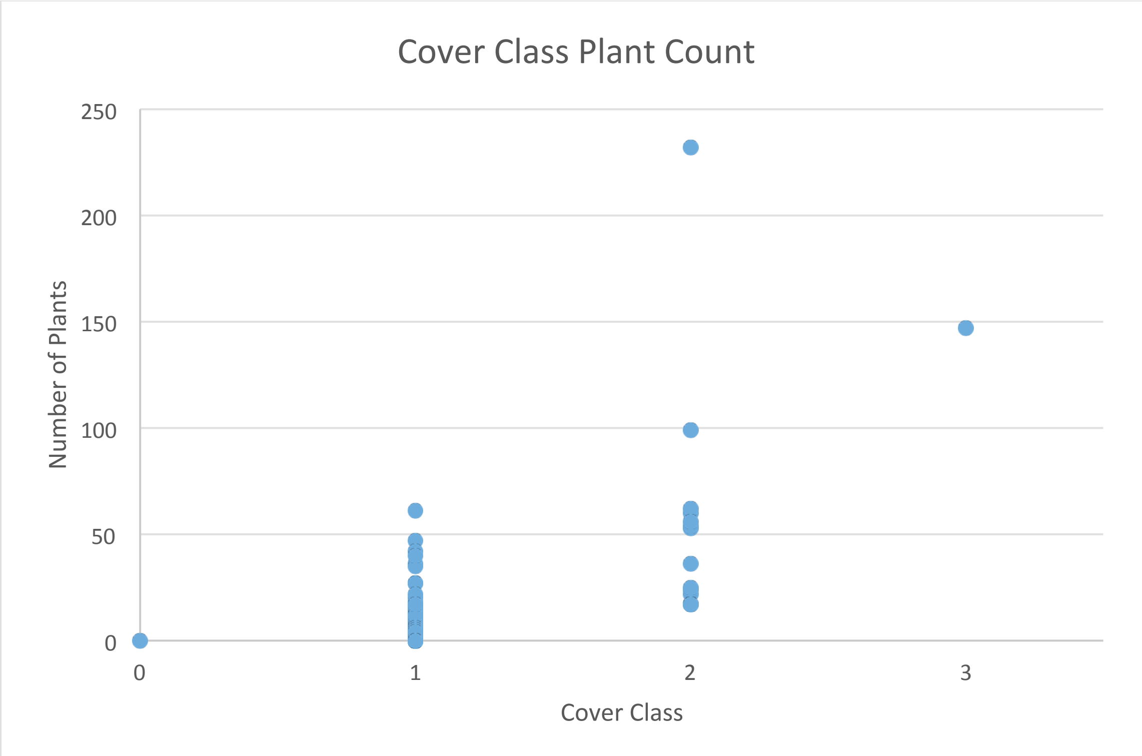


Figure 5—Plant count vs. cover class for all years, all sites.

Changes in shade did not affect the number of harebell plants. The one site with the greatest harvest-related decline in shade had the largest response in the number of plants, but this response was noted at a site where not only the shade was reduced, but trees were also felled and skidded on it. The methods used to measure shade reduction may have been insensitive to this mostly light harvest. The solar pathfinder measures solar energy blocked by vegetation between the instrument and the sun arc. Much of sun arc will remain obstructed by vegetation at low sun angles when the timber harvest intensity is light. The design with a single point measure of shade at each site may have also rendered it insensitive to the low harvest intensity. The lack of apparent relationship between canopy reduction and plant count might reflect the THPs proximity to the ocean where the marine influence on fog and temperature may ameliorate the possible desiccating effects of reduced canopy. Lastly, the species is adapted to mesic canopy openings in the coastal forest and woodland habitat (Sholars and Golec 2007), which are transitory with stand density and age or from disturbances such as fire and tree fall.

Wild pig (*Sus scrofa*) rooting impacted several sites. At the first post-harvest assessment, pigs had rooted through Rd 5 and vicinity. Although its cover class remained 1 throughout the study (fig. 3c), the count of plants dropped to 0 concurrent with the rooting, but returned to pre-harvest numbers in the third year post-harvest (fig. 4c). Pigs also rooted through a control site (S1) prior to data collection in post-harvest years 1 and 5 (2004 and 2008). Concurrently, cover class declined from the prior sampling period from 2 to 1 (fig. 3d) in both years; however, the response in the number of *C. californica* plants (fig. 4d) was not notable. The data suggests that at the intensity of pig rooting experienced, impacts to swamp harebell was immediate, but not a persistent effect at the site scale.

Only three of 20 sites had more swamp harebell plants at the end of the study than during the pre-harvest year. Five sites were extirpated and all but one of the remaining sites were occupied by less than half the pre-harvest count. The apparent strength of this timber operation impact signal is somewhat weakened by count declines observed in all the sites we assigned as “controls”, two of which were extirpated (S2, S15). Also, two of three sites (S7, S8) with increase counts of *C. californica* were in the full timber operations sites. The apparent incongruous response of harebell numbers among treatments may in part be due to the lack of independence of the sites. Other variables such as the species short life span (Sholars and Golec 2007) may have led to variable response among the clusters.

The subjective nature of our soil and soil moisture data prevents their use in assessing impacts. However, our observations on the study sites, as well as the matrix between suggests that being able to quantify soil moisture, may well enable better impact assignment and mitigation. Examples from our study sites that soil moisture is a critical habitat parameter for swamp harebell includes the strong and consistent decline in numbers on the road sites due most likely to soil compaction, consistent with Sholars and Golec's (2007) swamp harebell timber harvest risk assessment. In comparison, the data sheet for full operations site S8 that had high cover and numbers noted comparatively moist and lush herbaceous flora and attributed it in part to harvest-generated slash in the form of spread branches retaining runoff on-site for extended periods. Further, although not quantified in any way, we observed *C. californica* clusters during post-harvest years in the matrix, often in locations of greater apparent soil moisture. Because we cannot confirm swamp harebell presence at these matrix sites prior to harvest, we cannot say if they are examples of release of on-site propagules or examples of colonization.

We only collected information on associated species within the cover plots during 1 year, and then only presence. Thus, information on changes in species composition, either through on-site release or the intentional introduction for erosion control or accidental introduction from equipment and personnel, is not discernable from our data.

Swamp harebell is generally considered a hydrophilic plant (Anonymous 2016a, Baldwin et al. 2012, Sholars and Golec 2007), and Lichvar et al. (2016) assign it a wetland status of "obligate". Thus, the species is sensitive to changes in soil moisture and hydrology. However, timber harvest impacts associated with changes in soil moisture observed from this study is further complicated by the impact of the generally unusual and drying weather over the study period (fig. 6). The Palmer Drought Severity Index (PDSI) categorizes relative drought or wetness in a way that measures both the current moisture anomaly from average adjusted with that for prior time periods – thus drought or moist period's depth and duration. The PDSI calculates a single value based on precipitation and soil moisture as measures of supply and potential evapotranspiration and soil deficit as measures of demand, and groups these values into classes that range from extreme through severe, moderate, and near normal drought or moist. The PDSI helps explain the overall decline in *C. californica* numbers (fig. 2) even at the control sites that were not subjected to on-site timber impacts (fig. 5d). Using July as the month of assessment, PDSI values reveals the pre-project data was collected during near normal but droughty conditions preceded by two years of severe to moderate drought (Anonymous 2016c). Thus, swamp harebell plants may have already been under stress at the start of the study. Two years were relatively moist, but may have not been prolonged enough to enable swamp harebell to recover from the prior dry years. All other monitoring years were under extremes of drought/moist spectrum, the final monitoring period was "extreme drought." Furthermore, the wide swings in annual climate conditions may have lead our calendar-driven inventory schedule to be more variable than a phenologicaly-driven inventory.

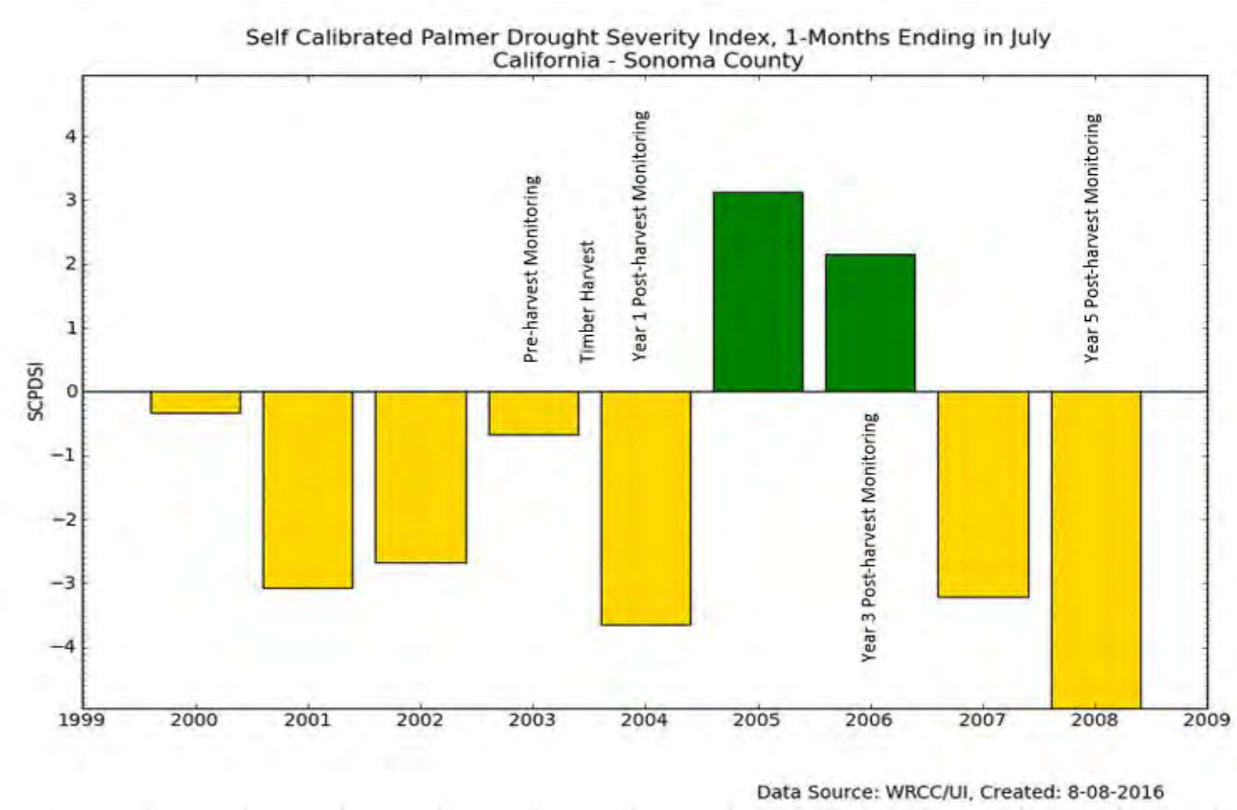


Figure 6—Palmer Drought Severity Index for Sonoma County from 2000 (3 years prior to study initiation) through 2008 (final year of study).

Despite a few notable exceptions, our results suggest a decline in *C. californica* numbers possibly as a response to timber operations when measured at the site scale. The species presence in this second-growth forest pre-harvest was not in itself an indication of its viability and resilience to past timber management. We have no information on earlier period swamp harebell plant numbers on which we could assess long-term trends and resilience to prior management. Many variables—natural and timber harvest related, in themselves or in combination—likely contributed to swamp harebell's post-harvest decline. For example, increases in solar insulation with partial canopy alteration and increase in seasonal moisture should have positive impacts. Otherwise, crushing or uprooting plants, changes in hydrology leading to soil drying, slash deposition heavy enough to obstruct light to the herbaceous layer, introduction and proliferation and of invasive plants, soil compaction, and herbicide application are expected to have negative impacts (Sholars and Golec 2007). To ensure the persistence of swamp harebell in managed timber lands, a better knowledge of its habitat, life history, and response to disturbance across the species range and under a range of climatic conditions is needed. Such knowledge will enable timber mangers to more effectively manage timberlands to promote swamp harebell viability.

Acknowledgments

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Two California Lineages of *Oxalis oregana*: Genetic Evidence for a Pleistocene Separation into Northern and Southern Glacial Refugia[1]

Chris Brinegar[2]

Abstract

In the Pacific Northwest, there are discontinuities in the lineages of several plant and animal species in the northern California/Oregon region that are thought to have their origins in the separation of populations into refugia during the Pleistocene glacial periods. Redwood sorrel (*Oxalis oregana* Nutt.), a common understory species of the California redwood forests and other Pacific Northwest temperate rainforests, was found to have two distinct genetic lineages in California based on sequence analysis of two chloroplast intergenic loci (*psbJ-petA* and *trnQ-5'rps16*) and the nuclear rDNA internal transcribed spacer (ITS) region. A "southern" lineage was detected in five populations from Big Sur to southern Humboldt County, and a "northern" lineage was dominant in two populations in northern Humboldt County and Del Norte County. The southern individuals had mixed sequence chloroplast haplotypes (presumably due to locus duplication and divergence or from chimeric tissue) while the vast majority of northern individuals had single sequence haplotypes. The northern and southern ITS variants were markedly divergent from each other, indicating a long period of separation between the lineages. Hybridization is occurring, as evidenced by an individual in a northern population that possesses a hybrid ITS genotype. The data suggest that these two groups were derived from an ancestral form that separated into two glacial refugia: a northern refugium within, or north of, the Klamath-Siskiyou ecoregion and a southern refugium in the California coastal forests.

Keywords: chloroplast DNA, glacial refugia, internal transcribed spacer, Klamath-Siskiyou ecoregion, *Oxalis oregana*, phylogeography, redwood sorrel

Introduction

Redwood sorrel (*Oxalis oregana* Nutt.) is a perennial herb that ranges from northern California to southern British Columbia in coastal temperate forests. In California it is strongly associated with *Sequoia sempervirens* (D. Don) Endl. and is often a dominant understory species of the redwood forest. Like the coast redwood, redwood sorrel is capable of both sexual and vegetative reproduction. The requirement for moist and shaded habitat makes redwood sorrel especially susceptible to the predicted climatic changes that will occur in much of the redwood region this century (Hayhoe et al. 2004). Therefore, it is important to document the genetic diversity and population structure of redwood sorrel in California as a climatic "indicator species" of the redwood forest.

The ranges of species with widespread Pacific Northwest coastal distributions were likely altered significantly by fluctuating climate during the glacial/interglacial periods of the Pleistocene epoch (2.58 to 0.012 mya). Genetic discontinuities have been observed in the north-south distributions of many Cascade and Sierran species, including Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.). (Furnier and Adams 1986), western sword fern (*Polystichum munitum* (Kaufl.) Presl) and red alder (*Alnus rubra* Bong.) (Strenge 1994), tailed frog (*Ascaphus truei*) (Nielson et al. 2001), dusky shrew (*Sorex monticolus)* (Demboski and Cook 2001) and Pacific giant salamander (*Dicamptodon tenebrosus*) (Steele and Storfer 2006). This phylogeographic pattern has been attributed to the contraction of ancestral populations into glacial refugia followed by southern and/or northern post-glacial recolonization (Brunsfeld et al. 2001, Soltis et al. 1997). Such studies yield insights into past

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responses to climate change and might also help predict how species will respond to the current warming trend.

In this study, seven populations of *O. oregana* from Big Sur to the Oregon border were analyzed at two chloroplast loci and one nuclear locus for DNA sequence variation. The discovery of two geographically distinct genetic lineages of redwood sorrel in California not only expands our knowledge of the genetics of redwood forest flora, but also provides strong evidence for a Pleistocene glacial refugium in coastal California for the southern lineage.

Methods

Sampling and DNA Extraction

Sampling was conducted between May 25 to June 2, 2012 in seven redwood state parks and reserves in California, ranging from Monterey County to Del Norte County (fig. 1). In each population single leaves from 10 individuals spaced a minimum of 50 m apart were collected and dried on silica gel desiccant. Genomic DNA was extracted from dried tissue using the method of Xin et al. (2003) and the Plant Genomic DNA Mini-prep Kit (Bay Gene, Burlingame, CA).

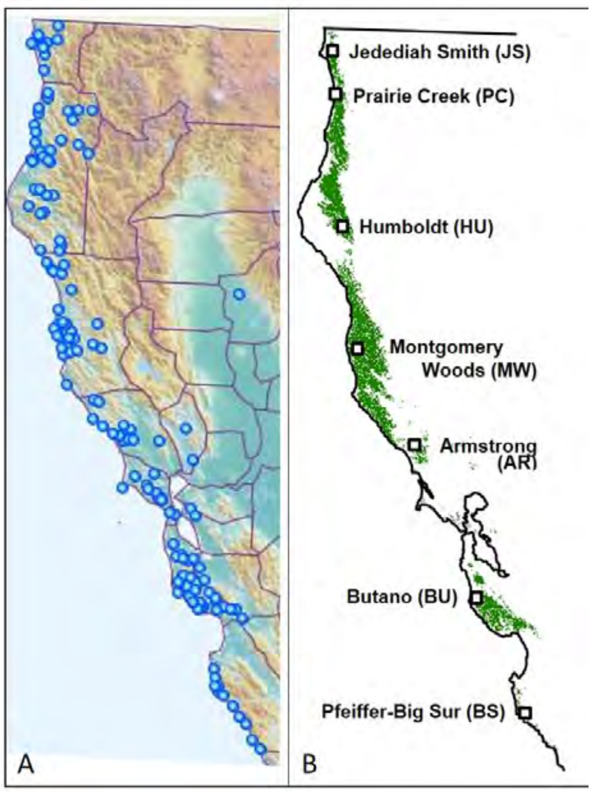

Figure 1—A. *Oxalis oregana* distribution in California (Calflora 2016). B. Coast redwood forest distribution with sites of state parks and reserves where *O. oregana* populations were sampled.

Amplification Conditions

The chloroplast intergenic sequences *psbJ-petA* and *trnQ-5'rps16* were amplified using the primers of Shaw et al. (2007). The internal transcribed spacer (ITS) region of nuclear ribosomal DNA (including ITS-1, 5.8S rDNA and ITS-2) was amplified using the ITSA and ITSB primers of Blattner (1999). All PCR reactions (20 µL) contained 1X PCR Gold Buffer (Applied Biosystems, Foster City, CA), 2.5 mM $MgCl_2$, 0.1 percent BSA, 0.2 mM each dNTP, 0.5 µM each forward and reverse primer, 1 unit AmpliTaq Gold DNA polymerase (Applied Biosystems) and 1 µL DNA extract.

Amplification of the *psbJ-petA* locus was performed with an 8 min polymerase activation at 94 °C followed by 30 cycles of 94 °C (30 sec), 56 °C (30 sec) and 72 °C (1 min), then a final extension at 72 °C (6 min). The *trnQ-5'rps16* locus PCR conditions were the same except 35 cycles were used. For the ITS locus, denaturation was at 94 °C (45 sec) and primer annealing was at 55 °C (1 min), otherwise conditions were the same as with the *psbJ-petA* amplification. PCR products were checked by electrophoresis in 2 percent agarose gels.

DNA Sequence Analysis

PCR products were purified for sequencing using PCR Clean-up Kit spin columns (Bay Gene) or DNA Clean and Concentrator-5 spin columns (Zymo Research, Orange, CA). Sequencing was performed by the University of Maine DNA Sequencing Facility (Orono, ME) using the psbJ, trnQ$^{(UUG)}$ and ITSA PCR primers. Sequences were aligned using ClustalX v.2.0 (Larkin et al. 2007). Areas of duplicate sequences in the "mixed se quence" haplotypes were analyzed manually and confirmed with Mixed Sequence Reader (Chang et al. 2012). Heterozygous genotypes at the ITS locus were identified by double peaks at variable sites in the sequencing electropherograms. Bootstrapped neighbor-joining trees of ITS sequence variants were constructed in MEGA5 (Tamura et al. 2011) using IUPAC ambiguity codes for the heterozygous genotypes.

Results

Chloroplast DNA Haplotypes

psbJ-petA Locus

PCR amplification of the *psbJ-petA* locus resulted in a product of approximately 700 bp. The alignment length was 616 bp of which 548 to 557 bp comprised the intergenic sequence (the length variation due to an indel). Three variable sites (two substitutions and a 9 bp indel) were detected in six haplotypes (table 1).

Table 1—Haplotypes of the chloroplast *psbJ-petA* locus in *O. oregana*

	Variable site[b]		
Haplotype[a]	168	275	276-284
A	G	T	ATCGAAACT
B	T	T	ATCGAAACT
C	T	T	ACCGAAACT
D i	G	C	deletion
ii	T	T	deletion
E i	G	T	ATCGAAACT
ii	T	T	deletion
F i	T	T	ATCGAAACT
ii	T	T	deletion

[a] Haplotypes D-F are "mixed sequence" haplotypes, each with two sequences indicated by i and ii.
[b] Haplotype A Genbank accession number is KX906973. Haplotype B-F polymorphisms are annotated as "variations" in the Haplotype A Genbank feature table.

Haplotypes A-C were single sequences as expected from haploid chloroplast loci. However, individuals with the other three haplotypes (D-F) had "mixed sequences" due to the presence of two closely related templates (indicated by i and ii in the table) in their PCR products. There were double electropherogram peaks at site 168 (in Haplotypes D and E) and site 275 (in Haplotype D). A sudden double sequence began at site 276 in the electropherograms of Haplotypes E and F and continued to the end, the cause due to the deletion of the nine bases at positions 276 to 284 in one of the two templates.

Changes in PCR annealing temperatures, the use of touchdown PCR protocols, the addition of PCR enhancers (DMSO and betaine), and the resequencing of all samples amplified from reisolated DNA gave the same results. In addition to the reproducible nature of the single and mixed sequence haplotypes, there was also a distinct geographic pattern to their distribution (see next section).

It should be noted that the Haplotype A sequence is the same as the Haplotype Ei sequence, and the Haplotype B sequence is the same as the Haplotype Fi sequence. Also, the second sequences (ii) of all three of the mixed haplotypes are the same.

trnQ-5'rps16 Locus

For the *trnQ-5'rps16* locus, PCR amplification resulted in a product of approximately 850 bp. The alignment length was 798 bp of which the intergenic sequence was 758 bp. Four *trnQ-5'rps16* haplotypes were detected due to base substitutions at three variable sites (table 2).

Table 2—Haplotypes of the chloroplast *trnQ-5'rps16* locus in *O. oregana*

Haplotype[a]	Variable site[b]		
	195	250	264
1	T	T	A
2 i	T	T	A
ii	T	T	T
3 i	T	G	A
ii	T	T	T
4 i	G	T	T
ii	T	T	T

[a] Haplotypes 2-4 are "mixed sequence" haplotypes, each with two sequences indicated by i and ii.
[b] Haplotype 1 Genbank accession number is KX906974. Haplotype 2-4 polymorphisms are annotated as "variations" in the Haplotype 1 Genbank feature table.

Only Haplotype 1 had a single sequence. The others (Haplotypes 2-4) were mixed sequence haplotypes, each with two closely related templates (i and ii) in their PCR products. Haplotype 1 is the same as the Haplotype 2i sequence, and the second sequences (ii) of all three of the mixed haplotypes are the same.

Results were reproducible regardless of PCR conditions or DNA preparation as with the *psbJ-petA* locus. Similarly, there was a distinct geographic distribution to the single vs. mixed sequence haplotypes. All individuals that had a single sequence haplotype at one chloroplast locus also had a single sequence haplotype at the other locus. The same was true for individuals with mixed sequence haplotypes.

Chloroplast DNA Haplotype Distributions

The distributions of chloroplast haplotypes for each locus are shown in fig. 2. In the pie charts, mixed sequence haplotypes are color-coded in orange and red shades while the single sequence haplotypes are displayed in blue shades.

Both the *psbJ-petA* (fig. 2A) and *trnQ-5'rps16* (fig. 2B) loci clearly show a sharp demarcation between populations having the single vs. mixed sequence haplotypes. All populations south of, and including, Humboldt Redwoods State Park (HU) in southern Humboldt County have mixed sequence haplotypes. For the *psbJ-petA* locus, Haplotypes E and F comprise 54 percent and 44 percent of the haplotypes in these populations; Haplotype D (2 percent) was found in only one HU individual. At the *trnQ-5'rps16* locus, Haplotype 2 was found in 90 percent of individuals in the five southern populations, with Haplotypes 3 and 4 contributing 8 percent and 2 percent, respectively.

Ninety percent of individuals in the two northernmost populations (PC and JS) had single sequence haplotypes at both loci. Haplotype A was dominant at the *psbJ-petA* locus (80 percent) and Haplotype 1 was dominant at the *trnQ-5'rsp16* locus (90 percent). Two individuals in Jedediah Smith Redwoods State Park (JS) had mixed sequence haplotypes at both loci (Haplotypes E and 2).

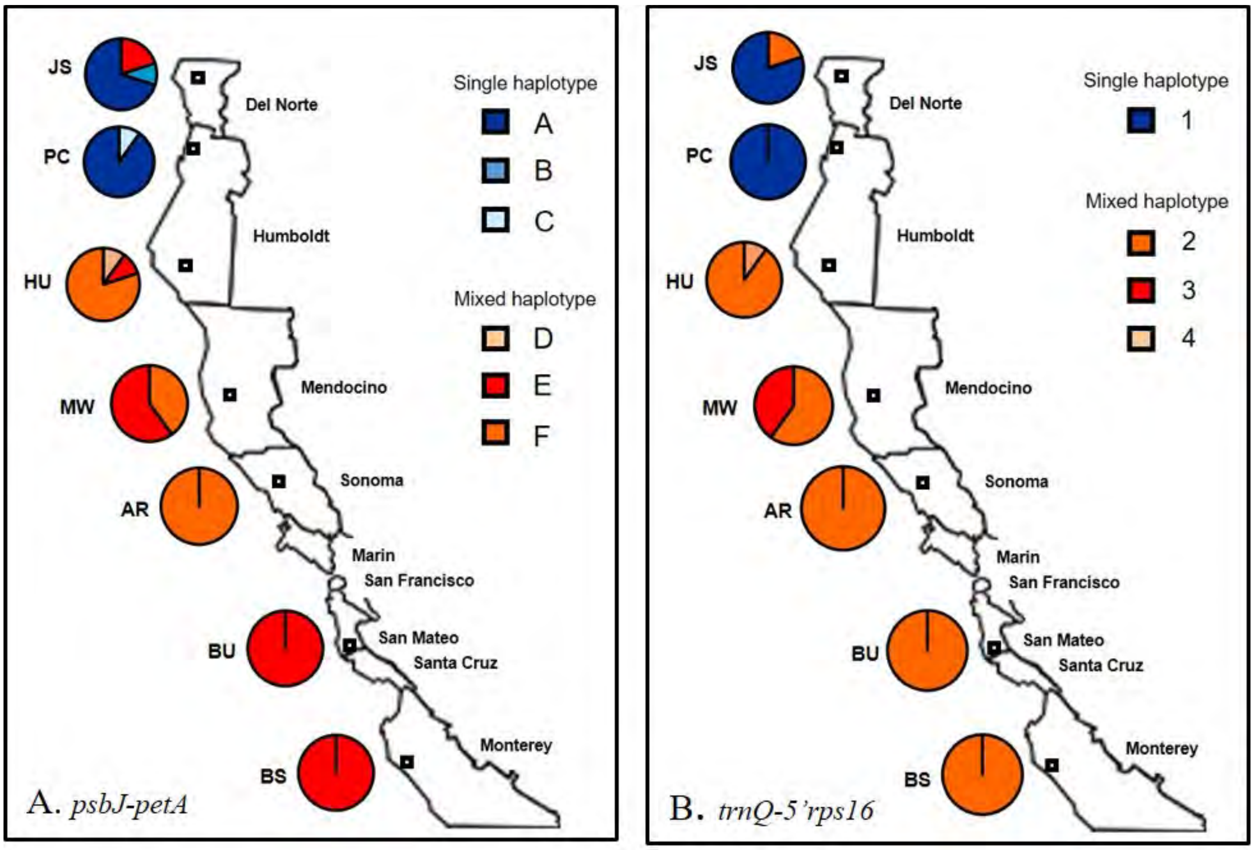

Figure 2—Distribution of chloroplast DNA haplotypes from loci *psbJ-petA* (A) and *trnQ-5'rps16* (B) in seven populations of *Oxalis oregana*. Two-letter abbreviations of sampled populations are at left. County names are at right. Haplotype descriptions are provided in tables 1 and 2.

ITS Sequence Variant Distributions

PCR amplification of the nuclear rDNA ITS region resulted in a product of approximately 850 bp. The total alignment length was 665 bp. The ITS-1 and ITS-2 alignment lengths were 223 bp and 221 bp, respectively.

There were 10 variable regions in the ITS-1 region and six in the ITS-2 region (table 3). Six closely related variants (S_1-S_6) were found almost exclusively in the five southernmost populations (fig. 3). A more divergent variant (N) was only found in the two northernmost populations. Four variants were homozygotes with only one allele (S_1, S_2, S_4 and N) while the other three (S_3, S_5 and S_6) were heterozygous, each having alleles that differed at only one variable site (as indicated by the G/A, C/T and C/T designations at sites 136, 469 and 591, respectively). A single individual in the JS population had a heterozygous N/S_1 genotype (not shown in table 3, but marked by an asterisk in fig. 3).

Table 3—Internal transcribed spacer (ITS) sequences of the southern (S) and northern (N) *O. oregana* variants

ITS variant	ITS-1 Variable site[a] 49	77	93	99	114	126	136	168	223	226	ITS-2 394	422	469	572	591	594
S_1	G	C	G	G	T	–	G	G	T	A	T	C	C	A	C	T
S_2	A	·	·	·	·	·	·	·	·	·	·	·	·	·	·	·
S_3	·	·	·	·	·	·	G/A	·	·	·	·	·	·	·	·	·
S_4	·	·	·	·	·	·	·	·	·	·	·	·	T	·	·	·
S_5	·	·	·	·	·	·	·	·	·	·	·	·	C/T	·	·	·
S_6	·	·	·	·	·	·	·	·	·	·	·	·	·	·	C/T	·
N	·	T	–	T	G	A	·	T	G	C	C	T	·	G	·	C

[a] Dots indicate the same base as the S_1 variant. The N variant Genbank accession number is KX906971. The S_1 Genbank accession number is KX906972. The S_2-S_6 polymorphisms are annotated as "variations" of the S_1 sequence in the Genbank feature table.

ITS Phylogeny

A neighbor-joining tree was constructed (fig. 4) using the southern and northern *O. oregana* ITS variants from table 3 (using IUPAC ambiguity codes for the three southern heterozygous variants) and three Genbank *Oxalis* ITS sequences: an *O. oregana* accession from southwestern Washington (JN836782), *O. acetosella* (JN836783) and *O. rosea* (JN836784), the latter serving as an outgroup.

The northern *O. oregana* variant (N) is 100 percent homologous with the Genbank *O. oregana* accession from Washington. Along with *O. acetosella* (common wood sorrel), these two form a cluster that is a sister clade to the southern *O. oregana* variants.

Discussion

Genetic analysis of one nuclear locus (ITS) and two chloroplast loci (*psbJ-petA* and *trnQ-5'rps16*) show conclusively that there is an abrupt genetic break in *O. oregana* in Humboldt County. Only mixed sequence chloroplast haplotypes and the southern ITS variants were found from Humboldt Redwoods State Park (HU) in southern Humboldt County to Pfeiffer-Big Sur Redwoods State Park (BS) in Monterey County. The single sequence chloroplast haplotypes and the northern ITS variant were found exclusively in Prairie Creek (PC) and Jedediah Smith (JS) Redwoods State Parks in northern Humboldt County and Del Norte County. The presence of two individuals with mixed sequence haplotypes in the JS population suggests a northward migration of the southern group, and the detection of a north/south ITS hybrid genotype in one of those individuals indicates that the two lineages are capable of mating.

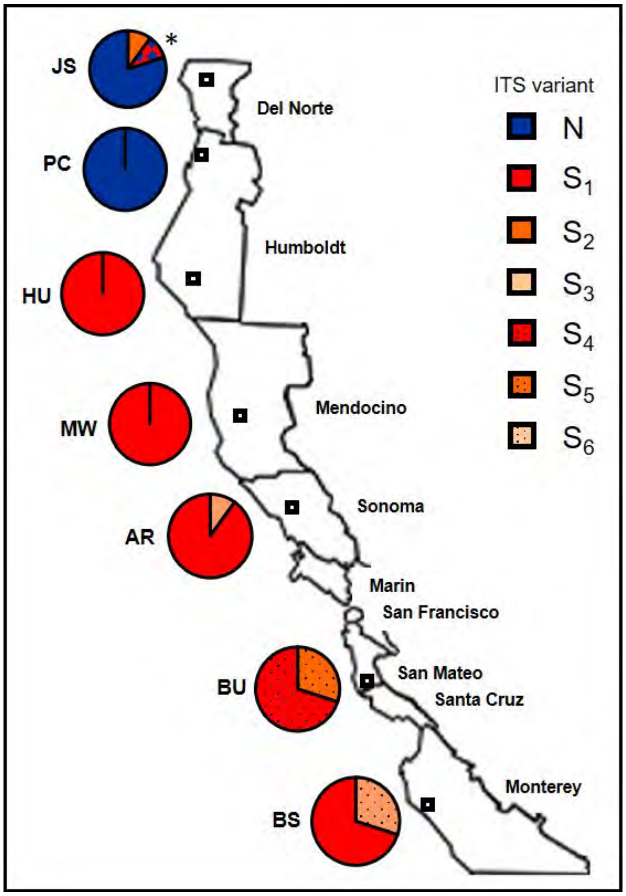

Figure 3—Distribution of nuclear ITS genotypes in seven populations of *Oxalis oregana*. The asterisk marks one individual in the JS population that had a north/south hybrid genotype (N/S_1). Two-letter abbreviations of sampled populations are at left. County names are at right.

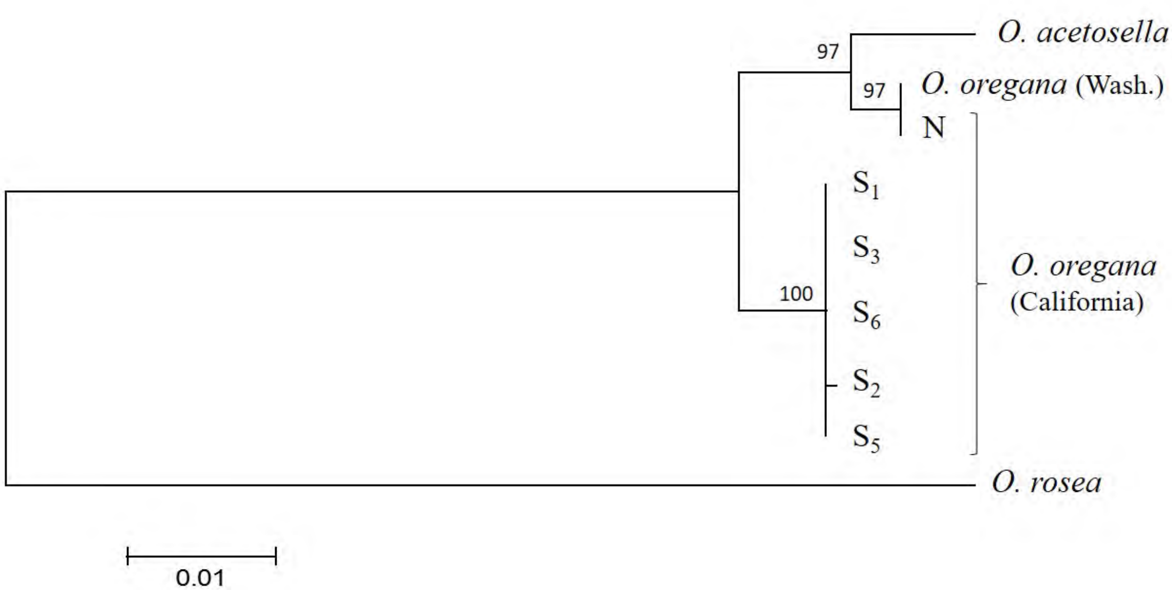

Figure 4—Neighbor-joining tree of northern and southern *Oxalis oregana* ITS variants with related *Oxalis* species. Numbers at nodes are bootstrap percentages.

The most obvious physical barrier separating these two genetically distinct groups is the northwest to southeast oriented mountain range that forms the southwestern border of the Klamath-Siskiyou ecoregion and divides the Redwood Creek watershed from the Klamath River watershed (fig. 5). Of the two northern *O. oregana* populations, JS is well within the Klamath-Siskiyou ecoregion while the PC collection site is just within the Redwood Creek watershed but only 8 km (5 mi) from the Klamath River at a coastal location where the mountains dividing the two watersheds are at their lowest elevation of less than 300 m (1000 ft). This area could be a migration pathway between watersheds for *O. oregana*.

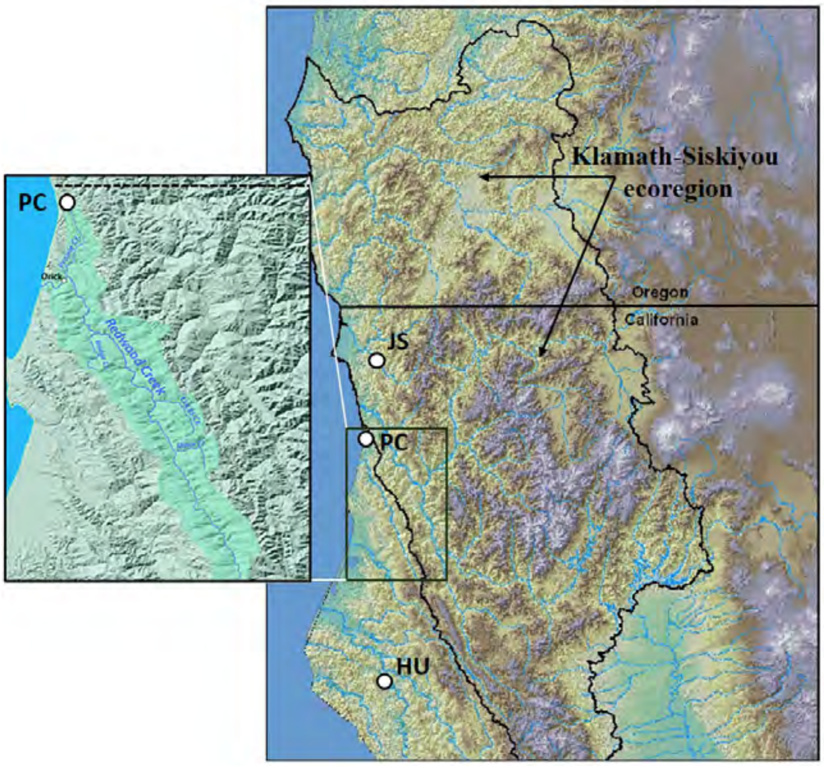

Figure 5—*Oxalis oregana* collection sites HU, PC and JS relative to the Klamath-Siskiyou ecoregion (Klamath-Siskiyou Wildlands Center 2016). Inset: Redwood Creek watershed (light green) and the PC collection site. Dashed line is the border between Humboldt and Del Norte counties.

The Klamath-Siskiyou ecoregion has been suggested as one of many possible Pacific Northwest glacial refugia (Roberts and Hamann 2015, Smith and Sawyer 1988, Whitaker 1961). Other potential refugia include Vancouver Island and the Olympic Peninsula as well as other coastal sites (Brunsfeld et al. 2001, Soltis et al. 1997).

In a review of chloroplast DNA-based phylogeographies of six Pacific Northwest plant species, Soltis et al. (1997) identified "northern" clade populations ranging from Alaska to central/southern Oregon and "southern" clade populations ranging from central Oregon to northern California. The authors offered two explanations for the observed data: the "north-south recolonization" hypothesis and the "leading edge" hypothesis. The first hypothesis requires that populations are separated into two distinct northern and southern refugia which diverge during their isolation. Post-glacial expansion might reunite these populations, but there will still be a marked north-south genetic discontinuity. The leading edge hypothesis allows for only one southern refugium. Rapid post-glacial northern dispersal and survival of limited genotypes (through drift and bottlenecks) at the leading edge of migration will ultimately result in less diverse, or even fixed, populations in the northern regions. In their analysis of several plant and animal studies, Brunsfeld et al. (2001) found the data to be more consistent with the north-south recolonization hypothesis.

The very different genetic compositions of the two groups of California redwood sorrel identified in this study support the conclusion that the lineages are derived from separate northern and southern glacial refugia which have met in northern California after post-glacial recolonization. The sharp geographic transition from one lineage to the other and the absence of northern cpDNA haplotypes and ITS genotypes in the southern group are strong arguments against the leading edge hypothesis as an explanation for the observed phylogeographic distribution.

It is possible that the southern redwood sorrel lineage was isolated in coastal California in the Pleistocene coast redwood ecosystem that at times extended as far south as Santa Barbara (Sawyer et al. 2001). The greater genetic variation found in the southern lineage suggests that its refugial population was larger than that of the northern lineage. The location of the northern lineage's refugium is much less certain. The northern ITS variant (N) from California and an accession from Washington have identical sequences (as do their *psbJ-petA* loci). This is consistent with a rapid post-glacial migration from a bottlenecked northern lineage and suggests two possible migration scenarios: 1) the northern lineage's refugium was in the Klamath-Siskiyou ecoregion with a subsequent migration northward during the Holocene, or 2) the refugium was in a more northern location followed by southward migration to its current southern limit in Humboldt County. A third possibility, though less likely from geographic considerations, is that the "northern" lineage was derived from a small, bottlenecked population at some other well isolated refugium in California, i.e., a second "southern" group that migrated northward through the Klamath-Siskiyou region.

The fact that the ITS region sequence of the northern variant is more closely related to common wood sorrel (*O. acetosella*) than to the southern variants indicates a very long separation between the two lineages. Löve (1968) once proposed that *O. oregana* be reclassified as a subspecies of *O. acetosella*, but results from morphological and hybridization studies countered her argument (see Packham 1978). However, the ITS data show the close evolutionary relationship between these two species, confirming the work of Gardner et al. (2012).

The mixed sequence haplotypes detected in the chloroplast loci of the southern lineage have two possible origins, one being that each locus underwent duplication and divergence. These types of mutations have been reported in many plant species (Xiong et al. 2009) and could result in mixed sequence haplotypes as long as the PCR priming sites are conserved. For this to occur in two widely separated loci such as (*psbJ-petA* and *trnQ-5'rps16*) is improbable. However, I have evidence (data not shown) of mixed haplotypes occurring only in the southern populations at two other chloroplast loci, as well. These combined four loci span the entire Large Single Copy (LSC) region of the chloroplast genome. There are no documented examples of duplication and divergence of the entire LSC region.

The other explanation for mixed sequence haplotypes is chimerism – the existence of two different variants of chloroplasts in different leaf cells of the same individual. This could originate by the mutation of chloroplast DNA in a leaf meristematic cell. Division of that cell (and of wild-type meristematic cells) would result in chimeric leaf tissue and lead to the amplification of diverged loci from two different chloroplast sources in the same individual. Chimerism, in the form of leaf variegation, is common in *Oxalis* species and occurs to a small degree in some natural populations of *O. oregana*. For entire populations to become chimeric would require vegetative reproduction since chimerism cannot be propagated sexually. Vegetative propagation would be an advantage in the more southern extent of the *O. oregana* range where summer temperatures are higher and precipitation is far less than the northern part of the range. In *S. sempervirens*, the diversity of a chloroplast microsatellite locus is significantly reduced in the species' southern populations (Brinegar 2011) which could be an indication of greater rates of vegetative reproduction. A more detailed genetic analysis will be required to determine whether the southern lineage of *O. oregana* is indeed chimeric.

Over the past century coastal fog has decreased approximately 33 percent in northern California (Johnstone and Dawson 2010) and summer temperatures are predicted to rise which will put more stress on the flora of the southern redwood forests and cause a northward recession as coastal woodlands and savanna expand (Hayhoe et al. 2004). If, in fact, the southern lineage of *O. oregana* is chimeric and propagates vegetatively, it might be somewhat buffered against these changes.

It is perhaps premature to consider the northern and southern lineages of *O. oregana* as subspecies, especially without conclusive evidence of phenotypic differences to support such a reclassification (as argued by Patten 2015). However, this study has shown that these lineages fit the other requirements of subspecies status: geographic and genetic distinctiveness, and the ability of the two groups to hybridize. Further research is needed to determine whether such a taxonomic change is warranted.

Acknowledgments

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Western Sword Fern Avoids the Extreme Drought of 2012-2014[1]

Emily E. Burns,[2] Peter Cowan,[2] Wendy Baxter,[3] Deborah Zierten,[2] and Jarmilla Pittermann[4]

The California drought of 2012 to 2014 was the most severe drought on record for the last century and likely millennium. Warm temperatures with below-average precipitation compounded over the three-year period, creating significant and sustained aridity over the course of three growing seasons throughout the coast redwood ecosystem. The citizen science project, Fern Watch, tracked the morphological response of *Polystichum munitum* (Western sword fern) in the coast redwood forest to the recent drought across a plot network spanning 10 sites and a latitudinal gradient of more than 800 km. Annually, tagged *P. munitum* individuals in the study were monitored for changes in crown size.

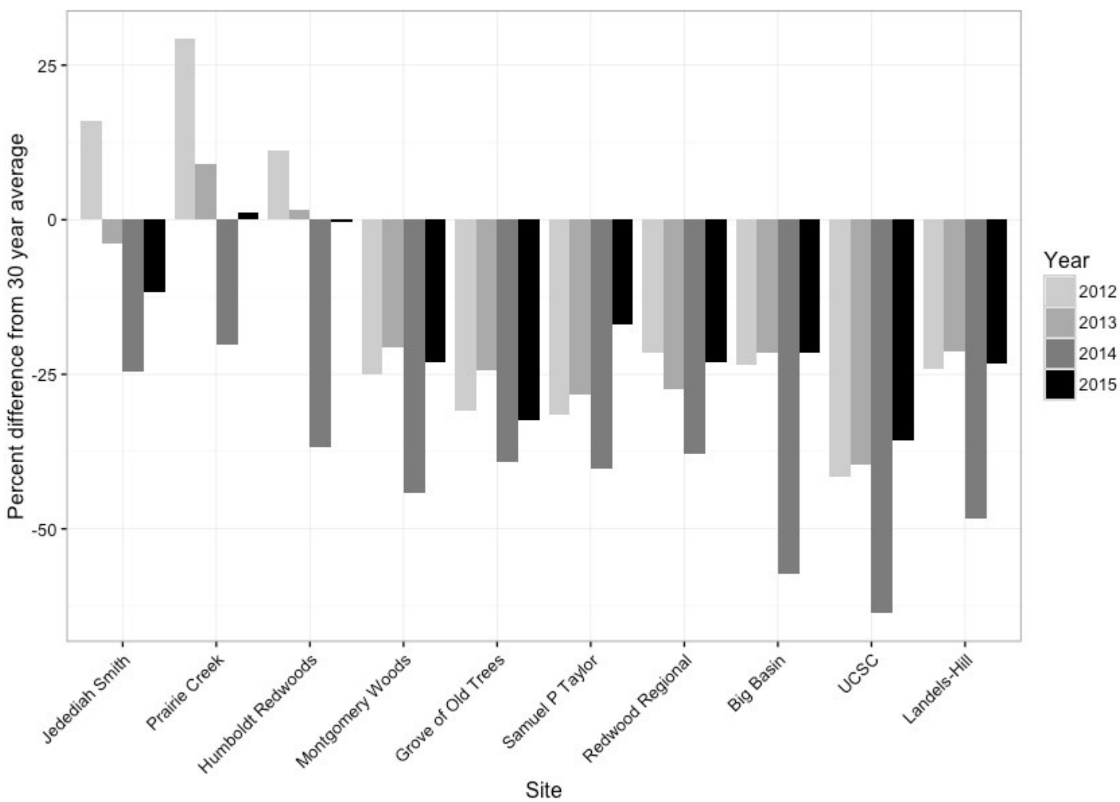

Figure 1 – The percent difference in total annual precipitation from the 30-year historic precipitation average by year and site. Sites are listed from north (left) to south (right) along the latitudinal gradient of the coast redwood ecosystem.

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[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.

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When compared to the 30-year climatic averages, the 2012-2015 drought manifested differently across the ecosystem (fig. 1). The three northernmost sites received >10% above-average total annual precipitation in 2012, while all sites to the south had significantly below-average annual precipitation with the most severe decrease in annual precipitation of 35% at the University of California, Santa Cruz Natural Reserve. In 2013, total annual precipitation showed a similar trend to 2012, with two of the three northern sites still receiving above-average precipitation (Prairie Creek Redwoods State Park and Humboldt Redwoods State Park) and the remaining sites receiving below-average precipitation. In 2014, the lowest total annual precipitation was recorded for each site, with three of the four most southern sites receiving less than 50% of historic total annual precipitation. In 2015, precipitation increased across the ecosystem, though 12 of the 14 sites still received below-average total annual precipitation by as much as 30%.

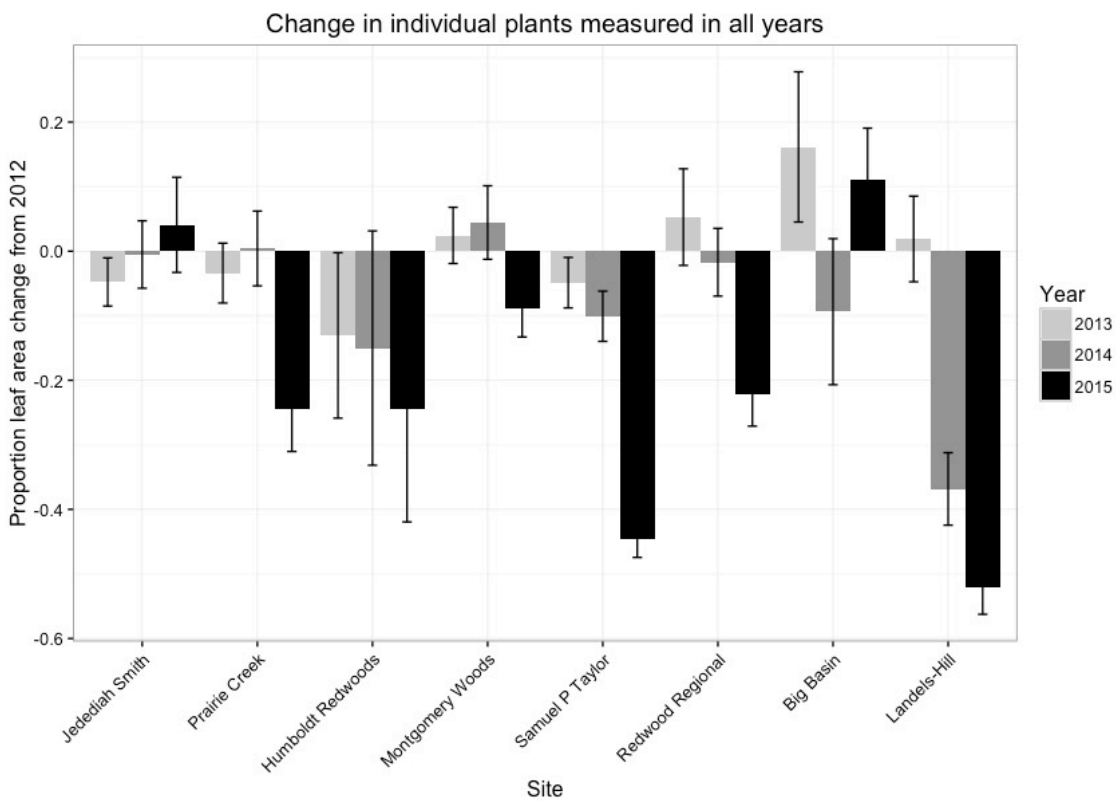

Figure 2 – Trends in mean (± standard error) total leaf area of *Polystichum munitum* across Fern Watch sites.

Results showed that *P. munitum* throughout the ecosystem range avoided the drought by reducing total crown leaf area by approximately one third, though leaf area increased over the drought period at the northernmost site (fig. 2). Individual ferns in several of the northern and wettest coast redwood forests had the highest average leaf area when the study began in 2012 and drought-induced reductions in leaf area caused these northern ferns to shrink and become more similar to *P. munitum* individuals in southern forests by 2015. Slightly higher precipitation levels in 2015 did not cause *P. munitum* to increase leaf area in the final year of the study. This may be because the winter rains occurred significantly earlier than the onset of new leaf growth, causing the 2015 spring growing season to be as dry as in previous years of the drought. Generally, the observed reduction in leaf area for *P. munitum* was caused by the production of shorter fronds and a reduction in the number of fronds per crown during the drought. While the lifespan of individual fronds was not tracked over the study, the reduction in crown leaf number could be the result of earlier frond senescence in the fall (shorter leaf lifespan), the production of fewer new fronds in the spring, or a combination of both.

Survivorship of *P. munitum* during the recent drought suggests that this hardy perennial fern species is capable of withstanding extreme drought events, providing insight into why this is the most common herbaceous species in the coast redwood ecosystem.

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Humboldt Marten Denning Ecology in a Managed Redwood-Dominated Forest Landscape[1]

Desiree A. Early,[2] Keith A. Hamm,[2] Lowell V. Diller,[2] Keith M. Slauson,[3] and William J. Zielinski[3]

Abstract

The Humboldt marten (*Martes caurina humboldtensis*) historically occurred in California's coastal redwood (*Sequoia sempervirens* (D. Don) Endl.)-dominated forests from northern Sonoma County, California to the Oregon border. The subspecies was thought to be extirpated due to over-trapping and loss of habitat until a small, remnant population was rediscovered in 1996 on the Six Rivers National Forest (Slauson, personal communication). Surveys conducted from 2004 to 2011on managed forests owned by Green Diamond Resource Company to the west of the remnant population yielded marten detections at several stations in the Pecwan and Bear Creek watersheds. To better understand the composition, movements, fates, and habitat use of marten on these managed lands, a collaborative effort between Green Diamond, the U.S. Department of Agriculture Forest Service Pacific Southwest Research Station, the California Department of Fish and Wildlife, and the Yurok Tribe was initiated in 2012. Between October 2012 and August 2016, 33 individual marten were captured (18 male, 15 female), and 24 (13 male, 11 female) were radio-marked. More than 2,000 telemetry locations have been obtained to date, resulting in 115 rest sites. After documenting reproduction in 2014, all adult female marten were monitored throughout each denning season (2014 to 2016) to determine denning phenology, kit production, site fidelity, and the characteristics and spacing of denning structures.

Sixteen female marten were monitored, resulting in 12 reproductive efforts and 34 confirmed den sites. Eleven female marten successfully weaned a minimum of 17 kits. Two reproductive efforts failed because the adults died prior to kit independence, and one reproductive effort was assumed to have failed due to lack of observation of the female with kits during August. Females less than 2 years of age did not attempt reproduction. The majority of confirmed dens (74 percent) were located in cavities of live trees or snags. The den trees/snags contained complex structural features (i.e., complex crowns, large limbs, broken tops, basal hollows, multiple cavities, and others) and were larger diameter trees than those within the surrounding stand. The location of den structures ranged from 6.1 to 610 m (20 to 2,000 ft) from the nearest manmade edge (road or recent harvest unit) and were located in a variety of stand ages. Fifty percent of reproductive females monitored for at least two breeding seasons reused a den structure from a previous season, and 60 percent reused a den structure within the same season. This study is ongoing and has the promise of providing important insights on how managed forests in the redwood region can provide denning habitat for coastal marten.

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Stream Amphibians as Metrics of Ecosystem Stress: a Case Study from California's Redwoods Revisited[1]

Hartwell H. Welsh, Jr.,[2] Adam K. Cummings,[2] and Garth R. Hodgson[2]

Abstract

Highway construction of the Redwood National Park bypass resulted in a storm-driven accidental infusion of exposed sediments into pristine streams in Prairie Creek Redwoods State Park, California in October 1989. We evaluated impacts of this ecosystem stress on three amphibians, larval tailed frogs (*Ascaphus truei*), coastal giant salamanders (*Dicamptodon tenebrosus*), and southern torrent salamanders (*Rhyacotriton variegatus*), by comparing densities by mesohabitat type in five sediment-impacted with five unimpacted streams. Impacted streams had six-fold higher pool bowl sediment loads and significantly lower densities of giant and torrent salamanders in most mesohabitats in impacted streams. Larval tailed frog densities were lower in the impacted stream set just in faster water, the only habitats where they occurred (riffles and step runs). In the winter of 1995, a large storm caused a second influx of sediments into all 10 streams. This created a gradient of disturbance, allowing us to examine the effects of repeated disturbances on this aquatic amphibian community. During the intervening 6 years, pool bowl sediment loads had increased 14-fold in the previously unimpacted and 3-fold in the previously impacted streams. Larval tailed frogs and torrent salamanders had declined further in both sets, and now no significant differences in overall densities were found; however, densities did vary by mesohabitat type. In contrast, giant salamander densities increased in both sets, but less so in the impacted streams, with numbers now greater in the faster mesohabitats of both sets. Of the three amphibians, giant salamanders were the most resistant to these extreme erosion events, whereas the other two species were accurate quantifiers of disturbance intensity, indicating their value as metrics of resilience in the redwood ecosystem.

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Remote Camera Monitoring and a Mark – Recapture Study of the Wandering Salamander in a Redwood Forest Canopy[1]

Jim Campbell-Spickler[2] and Stephen C. Sillett[3]

Abstract

Crowns of old redwoods (*Sequoia sempervirens* (D. Don) Endl.) are teaming with life. Storm damage followed by recovery via trunk reiteration increases the structural complexity of redwood crowns over time. Bark and wood surfaces within complex redwood crowns accumulate debris and become covered with epiphytes. Arboreal soils develop beneath mats of the leather-leaf fern, *Polypodium scouleri*, and within pockets of decaying wood colonized by the evergreen huckleberry, *Vaccinium ovatum,* promoting water storage and allowing desiccation-sensitive creatures to flourish high above the ground. These habitats provide year-round refugia for insects, mollusks, and vertebrates like the wandering salamander (*Aneides vagrans*).

In 2013, we installed 48 crack-boards, which were designed to provide shelter for salamanders, within crowns of three trees at the Redwood Experimental Forest (REF), Klamath, California. Since then we conducted 38 visits and marked 59 salamanders, 21 of which have been recaptured at least once. Twelve recaptured salamanders were found in locations differing from their original point of capture. Of the 59 marked individuals, 27 were fitted with passive integrated transponder (PIT) tags. Using a handheld reader, tagged individuals can be located within arboreal habitats without disturbance. Detecting movement between ground and crown locations is also possible via tag-reading stations installed around tree bases.

Using power from a tree-based solar panel array and battery bank, a network of motion-triggered cameras continuously monitors use of selected habitats within two redwoods. To date, our remote monitoring yielded 70 salamander sightings and observations of many other arboreal organisms, including a frog, three rodent species, a bird, and a wide variety of invertebrates. Video data are sent via Wi-Fi to an off-site recording device that can be accessed remotely via computer. Within-crown microclimate data, including soil moisture at six locations, are also collected and available for download via the same Wi-Fi network system supporting the cameras. Internet service is available at the off-site location, which makes online live-streaming of video and microclimate data possible.

The mark-recapture study will be continued through 2018 and expanded to include PIT tagging salamanders captured on the ground beneath study trees. We will use paired stationary dataloggers and tag-reading stations installed both near ground level and within crowns to quantify movement of tagged salamanders between arboreal and terrestrial habitats. Video monitoring of arboreal habitats will also continue. Many of our video detections have been unexpected, revealing arboreal behaviors of species that until recently were unknown to occur in redwood canopies.

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[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.
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Investigating the Distributional Limits of the Coastal Tailed Frog (*Ascaphus truei*) Near its Southern Range Terminus[1]

Robert B. Douglas,[2] David W. Ulrich,[3] Christopher A. Morris,[2] and Matthew O. Goldsworthy[4]

Abstract

Documenting species distribution patterns and habitat associations is a necessary prerequisite for developing conservation measures, prioritizing areas for habitat restoration, and establishing baseline conditions for long-term monitoring programs. The coastal tailed frog (*Ascaphus truei*) ranges from coastal British Columbia to northwestern California and is one of several species of co-occurring amphibians that breed in cold headwater streams. Several accounts of tailed frog distribution suggest that the species' range has been unchanged during the age of industrial timber harvesting. However, explicit knowledge regarding its historic distribution is incomplete as there are little published data to support contemporary delineations of its southern range boundaries in Mendocino County, California.

To address this knowledge gap, we initiated tailed frog distribution surveys on 90,157 ha of coast redwood (*Sequoia sempervirens* (D. Don) Endl.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests managed for commercial timber production. From 2003 to 2011, 400 stream reaches were surveyed in 59 planning watersheds, including several in northern Sonoma County. Larval tailed frogs were detected in 82 reaches covering 16 planning watersheds, all located in Mendocino County. Based on these surveys, the southern and eastern limits of tailed frog distribution were Schooner Gulch (7 km southeast of Point Arena) and Elk Creek (11.4 km east of the Pacific Ocean), respectively. Because much of this region is privately owned, additional surveys in nearby streams and watersheds may be necessary to establish more definitive range boundaries.

Although distribution surveys were the primary focus of this project, we also evaluated environmental factors potentially influencing tailed frog presence/absence in a model selection framework based on the published literature. A total of 12 variables were measured, either in the field or using a GIS, to characterize habitat at survey locations. Preliminary results indicate that factors such as stream embeddedness, water temperature, and distance from the coast may not only be important predictors of tailed frog presence at a given location, but may also interact to ultimately limit their distribution within this region. We discuss these results further in light of data quality, analysis methods, and tailed frog biology.

*Key*words: *Ascaphus*, distribution, range limit, redwood, temperature

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Tree Size, Growth, and Anatomical Factors Associated with Bear Damage in Young Coast Redwood[1]

John-Pascal Berrill,[2] David W. Perry,[2] Larry W. Breshears,[2] and Garrett E. Gradillas[2]

Precommercial thinning is an important tool for coast redwood (*Sequoia sempervirens* (D Don) Endl.) forest management but is often followed by black bear (*Ursus americanus*) damage in northern parts of redwood's natural range (Fritz 1951; Giusti 1988, 1990; Hosack and Fulgham 1998). The bears scrape off bark and feed on the sugar-rich phloem of coast redwood and coast Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirb.) Franco) (Kimball et al. 1998). A prior study at the same study area within the Mill Creek watershed in Del Norte Coast Redwoods State Park, near Crescent City, in Del Norte County, California, showed that frequency of damage was higher among larger trees in these conifer-dominated mixed even-aged stands, and that redwood was more likely to be damaged than Douglas-fir, especially near roads. Precommercial thinning (PCT) incited damage to redwood, and PCT to lower residual densities incited more damage in Douglas-fir. Unthinned control stands were least damaged. Increment cores collected from pairs of damaged and undamaged redwood trees confirmed that damage occurred after thinning and revealed that,at the time of bear damage, trees sustaining damage had been growing faster than undamaged trees of similar size (Perry et al. 2016). These findings support mitigation strategies such as lighter thinning, leaving higher densities of redwood in anticipation of higher damage rates, and leaving unthinned buffers adjacent to roads and other paths travelled by bears.

We examined relationships between phloem thickness, recent annual radial growth, and redwood tree diameter at breast height (DBH; 1.37 m) on increment cores collected in the stands studied by Perry et al. (2016) at Mill Creek. Breast height increment cores were collected in eight stands: three controls, three low-density (heavily thinned) stands, and two high-density (lightly thinned) stands. In each stand, core samples were taken from three damaged trees and three undamaged neighboring trees of similar diameter and height. In order to ensure damaged tree and neighboring tree experienced similar site and stand conditions, the neighboring tree had to be located < 3.66 m away from the damaged tree (Perry et al. 2016).

At the lab, all core samples were dried at 40 °C for 24 hours, then glued to medium density fiber board. The samples were then sanded down sequentially starting with coarse sandpaper and ending with an extra fine 1600 grit. The samples were scanned and imported into WinDENDRO (Regent Instruments Inc.). Thickness measurements were collected for phloem on undamaged trees, and the last (most recent) annual growth ring and the last 5 years of radial growth immediately preceding the year of damage. Variability in phloem thickness prompted us to gather more data by measuring and coring an additional 27 undamaged trees in six of the sample stands.

We used linear regression analysis with SPSS (IBM Software) to study relationships between DBH at the time of damage, radial growth leading up to the time of damage (1 year or 5 years of growth prior to damage), and phloem thickness measured on undamaged trees (table 1). Data were transformed to reduce skewness in data distributions. Model selection was based on AIC (Burnham and Anderson 2002).

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Table 1—Summary data for redwood tree size (DBH) and anatomical variables: (DBH increment over most recent growing season (1-yr), average DBH increment over most recent 5 years (5-yr), and radial thickness of phloem layer at DBH, Mill Creek, Del Norte County, California)

Variable	n	Mean	Std. dev.	Min.	Max.
DBH (cm)	47	29.81	8.22	15.75	51.74
1-yr DBH increment (mm yr^{-1})	47	3.09	1.36	0.48	5.79
5-yr DBH increment (mm yr^{-1})	47	3.52	1.64	0.61	7.25
Phloem thickness (mm)	47	3.83	0.90	2.01	5.38

The phloem layer was thicker in larger redwood trees, and slightly thicker in redwoods exhibiting more rapid diameter growth over the growing season preceding sampling. The 1-yr DBH increment was a better predictor of phloem thickness than the 5-yr DBH increment. The most parsimonious model for phloem thickness included only tree size (DBH) as a predictor variable. The slightly better-fitting model with equivalent AIC score included tree size and recent DBH growth (1-yr DBH increment) (table 2). The low variance inflation factors (VIF = 1.028; where 1 = no relationship, 10 = important collinearly) indicated that these two predictor variables were indeed independent. Modeled estimates indicated that phloem thickness was influenced by growth rates more among smaller trees in our sample (fig. 1).

Table 2—Regression models for radial thickness of redwood phloem layer at breast height as a function of tree size (DBH) and DBH increment over most recent growing season (1-yr), Mill Creek, Del Norte County, California (dependent variable: squared phloem thickness, mm^2)

Variable	Estimate	Std. error	Pr > |t|
Intercept	-16.6432	11.08	0.1401
Ln DBH (cm)	7.8832	3.29	0.0210
Ln 1-yr DBH increment (mm yr^{-1})	4.1687	2.64	0.1217
$R^2_{adj.} = 0.144$ $AIC = 174.97$			
Intercept	-13.8969	11.12	0.2178
Ln Dbh (cm)	8.7429	3.30	0.0111
$R^2_{adj.} = 0.116$ $AIC = 175.55$			

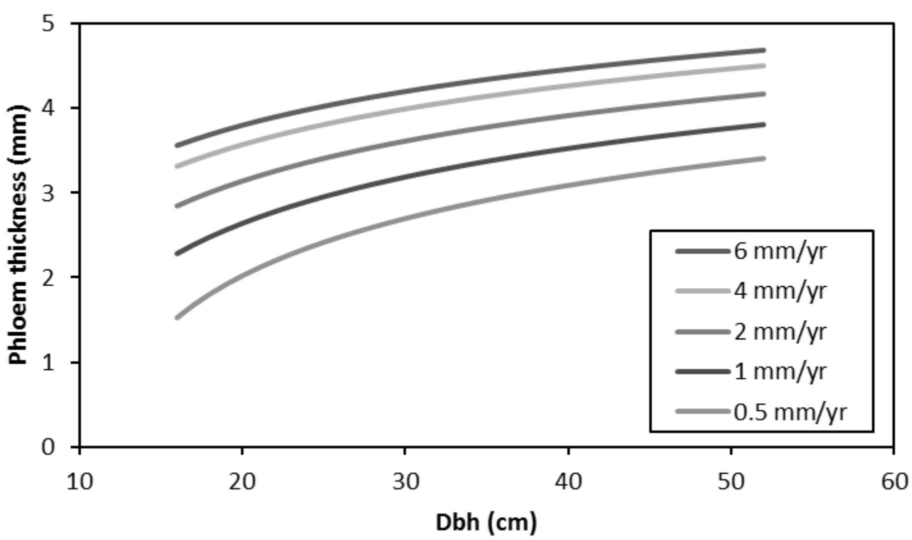

Figure 1—Modeled estimates of phloem thickness at DBH, Mill Creek, Del Norte County, California.

We recommend further study into the relationship between anatomical attributes such as phloem thickness and their relationship to forest management and the probability of bear damage in a variety of stand ages and structures.

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Black Bear Damage to Northwestern Conifers in California: a Review[1]

Kenneth O. Fulgham[2] and Dennis Hosack[3]

A total of 789 black bear damaged trees were investigate over a multi-year period on 14 different study sites chosen on lands of four participating timber companies. The sites ranged from 30 to 50 years of age. Four different conifer species were found to have black bear damage: coastal redwood (*Sequoia sempervirens* (D. Don) Endl.), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), Sitka spruce (*Picea sitchensis* (Bong.) Carr.), and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.). Numerous variables were measured on each black bear damaged tree (diameter at breast height [DBH; 1.37 m], distance to nearest neighbor, height to beginning of damage, age of damage, and tree species). Very important variables were: tree species, DBH, percent girdle, distance to road, and basal area.

Results are present in fig. 1 and tables 1 and 2. Black bear damaged trees varied widely in DBH from approximately 10.2 cm (4 inches) up to approximately 104.1 cm (41 inches), with the overall average DBH of about 45.7 cm (18 inches). Black bear damaged coastal redwood trees were found to be growing alone in about 33 percent of the population and 67 percent of the population growing as multiple stump sprouts. The black bear damage that occurred on any one tree varied from complete girdling to a small patch of bark removed near the base of the tree. Twice as many black bear-damaged redwood trees occurred in the 76 to 100 percent girdled-circumference group than occurred in the 1 to 25 percent girdled-circumference group. Mean trees damaged by black bears ranged from approximately 4.2 trees/ha (1.7 trees/ac) to 72.5 trees/ha (29.3 trees/ac), with the average approximately 19.0 trees/ha (7.7 trees/ac). Average annual increment of black bear damage ranged from 0.3 trees/ha (0.1 trees/ac) to 23.5 trees/ha (9.5 trees/ac). The mean annual increment of black bear damage was 6.0 trees/ha (2.4 trees/ac).

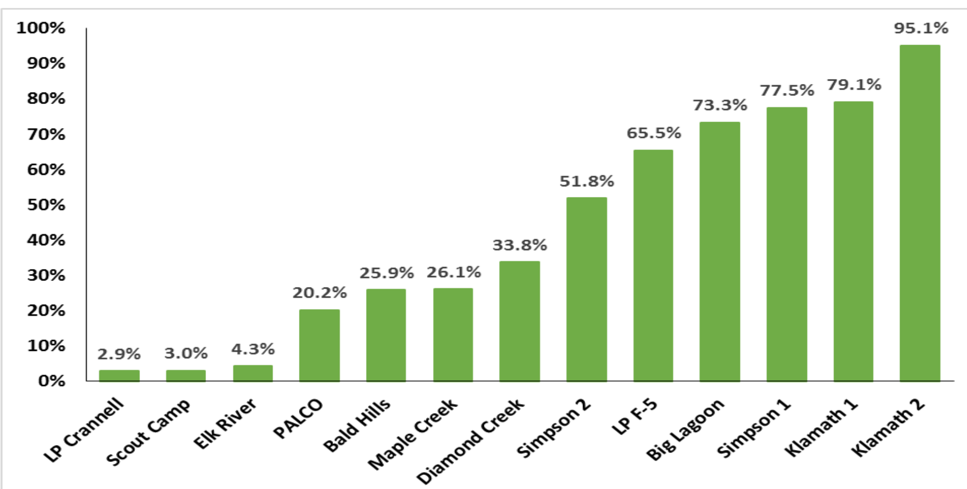

Figure 1—Percent of black bear-damaged redwoods by location.

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Table 1—Number of black bear damaged trees per hectare or per acre on 14 sites

Site	Stand age	Damaged trees	Mean number damaged trees/ha	Mean number damaged trees/ac
Crannell	35/50	14	4.2	10.38
Scout Camp	17	13	5.1	12.60
ARCO	50	153	7.9	19.52
Elk River	23	27	9.7	23.97
Big Lagoon	38	224	14.8	36.57
Bald Hills	40	65	17.1	42.26
Simpson 2	35	91	21	51.89
Simpson 1	30	141	26.7	65.98
Diamond Creek	17	81	36.8	90.93
PALCO	30	101	38.5	95.14
Maple Creek	35	100	40.2	99.34
Klamath River 1	17	86	40.8	100.82
Klamath River 2	23	86	42	103.78
LP F-5	23	151	72.5	179.15
Mean	29	95	26.5	65.48

Black bear damage to regenerating redwoods conclusions from 1993 McIntire-Stennis Report #94[4] by Fulgham and Hosack are:

- Mean annual bear damage to regenerating redwoods of six trees/ha/year (13.4 trees/ac/year) may represent a serious economic problem.
- Bear damage at rotation age indicated that bear damaged redwoods was highly variable, but redwoods comprised almost all damaged trees.
- Yield loss due to bear damage projected to rotations age (~50 years) was highly variable and averaged over 21 percent with the highest over 54 percent.
- Precommercial thinning was not determined as no replicated paired plots were studied.
- Personal communication during the period of this study indicated that most of the black bear damage to redwoods occurred in Humboldt and Del Norte counties with some observed damage.
- Mean density of 26.5 bear-damaged trees per hectare (65.48/trees/ac) suggests serious impact on the regenerating stands studied (1998 to 1991).
- On eight of 13 sites, redwood was damaged in significantly greater proportions than it was available in the stands studied.
- Damaged redwoods had a larger mean DBH than the average redwood tree found in the stands studied.
- There were more redwoods in the 76 to 100 percent girdling category than in the other three percentile categories in the stands studied.
- More bear-damaged redwoods were found along roads and trails than were statistically suspected in the stands studied.

[4] MS #94 can be found at: http://hdl.handle.net/10211.3/176892.

Only redwood trees that had received 100 percent girdled-circumference were considered for an estimate of economic loss. Total volume lost on all transects, average volume lost per unit area, and total volume lost per site were calculated. Therefore, the total economic loss per site was calculated and is presented.

Table 2—Loss estimation based on average prices for logs 43.2 cm (17 inches) or larger in city of Arcata Contract-2016

Study Site		BdFt-No Bear	BdFt-Bear	BdFt-Loss	Value Lost		
Big Lagoon		76,726	56,068	20,658	$ 21,825.18		
Simpson 1		103,907	67,336	36,571	$ 38,637.26	**$ 22,381.38**	**Average loss per acre for all the stands shown**
Bald Hills		106,354	101,832	4,522	$ 4,777.49		
Simpson 2		108,454	106,626	1,828	$ 1,931.28		
PALCO		206,999	181,745	25,254	$ 26,680.85	**21,184**	**BdFt volume loss per acre averaged over the**
LP Crannell		149,375	147,027	2,348	$ 2,480.66		**13 Stands**
LP F-5		41,390	30,073	11,317	$ 11,956.41		
Maple Creek		163,114	127,086	36,028	$ 38,063.58	**16%**	**This is the volume lost due to bear damage.**
Klamath 1		94,048	42,971	51,077	$ 53,962.85		
Klamath 2		82,706	59,799	22,907	$ 24,201.25		
Elk River		263,389	229,239	34,150	$ 36,079.48		
Scout Camp		274,508	272,754	1,754	$ 1,853.10		
Diamond Creek		53,078	26,094	26,984	$ 28,508.60		
Sampled Stands	**#**	1,724,048	1,448,650	275,398	**$ 290,957.99**	**This represents the value lost on 13 acres of studied**	
						timberland (one acre from each sampled stand)	

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The Political Ecology of Forest Health in the Redwood Region[1]

Chris Lee,[2] Yana Valachovic,[3] and Dan Stark[3]

Abstract

Imported forest pests have changed North American forests and caused staggering monetary losses in the centuries since the country was founded. Since most problem-causing non-native pests are innocuous in their home ranges, where they have coevolved with their host trees, experts cannot predict which pathogens or insects will have lethal effect on other continents. Many non-native pests are unknown to science until they cause problems in their new homes. One common response to the threat of non-native insects and diseases in our forests is to appeal to science to develop technical means for management or eradication, yet common sense tells us that it would be more cost-effective and ecologically efficient to prevent pest introductions in the first place. The discipline of political ecology explores the ways in which many environmental issues that are usually presented as scientific or technical problems are actually policy issues that have been redirected into scientific discussion in order to avoid acknowledging the need for hard political choices. The political ecology of forest pest management is very relevant to 21st-century forestry in the redwood (*Sequoia sempervirens* (D. Don) Endl.) region, where we have no way of knowing whether the next pest will be the one to target redwood or another native California tree species. These questions are especially important to consider and to educate policymakers about in California, where the iconic coast and Sierra (*Sequoiadendron giganteum* (Lindl.) Buchholz) redwoods have limited distributions that may make them vulnerable to future pest invasions.

Introduction

The process of globalization begun 5 centuries ago continues to intensify, as communication, physical travel, cultures, and markets integrate themselves more tightly. In the United States, political debate plays out over the proper role and scale of American engagement, but in one way or another involvement with overseas friends and enemies shapes nearly all American lives at the most personal levels. This involvement shapes the physical environment as well. Not only do Americans extract resources for export, but they also bring in living organisms from across the globe, and some of these organisms have the potential to remake both physical landscapes and biotic communities. No kingdom of life, from the smallest prokaryote to the tallest redwood tree, is entirely immune to this reshaping.

Non-native invasive species (NNIS) cause devastating economic losses to individuals and communities, and devastating ecological losses to forests, every year. Pimentel et al. (2005) estimated that the costs of NNIS in the United States alone amounted to nearly $120 billion per year and that 42 percent of species listed as threatened or endangered were imperiled primarily because of non-native, invasive competitors or predators. Losses attributed to NNIS include impacts to wildlife dependent on specific plant species; increases in wildfire hazard as trees die and become part of the fuelbed; hazards to human life and infrastructure from falling trees; loss of tree species that are culturally and spiritually important to specific communities; aesthetic degradation; loss of amenity values provided in urban and individual home settings; and nuisance impacts to water supplies, roadways, and houses.

The first-world practice of unintentionally bringing exotic organisms ashore has changed the ecology of world forests. Some of these changes have happened more or less quickly and disruptively, some more slowly and subtly. *Phytophthora cinnamomi*, an oomycete pathogen

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probably moved from southeast Asia, has reduced entire forests of jarrah (*Eucalyptus* spp., with tree, shrub, and grass associates) to barrens (Weste 2003, Weste and Marks 1987)—an example of a disruptive change. On the more subtle end of the scale, the pine-infecting decay fungus *Heterobasidion irregular*, was probably introduced by United States troops to Italy during World War II, since when it has been slowly expanding in range and appears to be outcompeting the native pathogen *Heterobasidion annosum*, a process that requires genetic analysis to trace fully (Gonthier et al. 2007).

Understanding the biological processes of these invasions, and their impacts, has required scientific expertise. The task of dealing with the impacts has inspired numerous scientist-led coalitions for pest management in invaded forests (some examples: Save the Ash Tree Coalition, California Oak Mortality Task Force, O'ahu Invasive Species Committee, Dieback Working Group, Continental Dialogue on Non-Native Forest Insects and Diseases, National Invasive Species Council—not to mention those dealing with non-forest or non-plant pests). Although these groups have not been comprised of scientists exclusively, but have also included plant care professionals, politicians, concerned citizens, and landscape managers, by and large they have looked to those with extensive scientific training to lead their efforts. Problems involving forest pests are among the natural resources management issues for which university extension programs and management agency personnel increasingly invoke the need for "science-based information" to satisfy a need for objectivity among competing public and private claimants.

Although technical expertise is clearly required to identify NNIS, to understand their biology, and to develop management technologies for tree and forest protection once NNIS are present on the landscape, there are numerous steps along the pathway of forest pest management that depend not on scientific understanding, but rather on political decision-making. Appealing to the authority of science can often be a potent way for decision-makers to mask the political nature of decisions that affect the environment, or to hand off responsibility for making those decisions. Exposing the politics behind these decisions and quantifying their environmental effects is the domain of the hybrid academic discipline called political ecology (Robbins 2004). This paper brings the political ecology project to pest management in the coast (*Sequoia sempervirens* (D. Don) Endl.) and Sierra (*Sequoiadendron giganteum* (Lindl.) Buchholz) redwood regions by seeking to demystify the role of science in invasive species and forest health protection and pointing out some of the ways in which science-based decision-making, to be effective, should be supported by moral awareness, place-based argument, and political action. Without intending to fear-monger, we use examples from California and beyond to argue that we must recognize the interplay between politics and science as it underlies forest pest management if we want to prevent future invasions of NNIS that could extirpate iconic tree species in the redwood region—perhaps even redwood itself.

Pest Exclusion

It helps to look at management of NNIS in two distinct phases: pre-invasion and post-invasion. We can call pre-invasion management practices "exclusion" and post-invasion practices "management." Management involves a myriad of possible actions, including eradication, prophylaxis, containment, slowing the spread, resistance breeding, and others. Exclusion, on the other hand, involves just a few practices, such as legislation, inspections, and pest destruction. Although common sense tells us that the most effective means of NNIS management consists of preventing them from arriving and establishing in the first place, we don't have to depend on common sense: we have numerous convincing examples of the costs of failing to do this. Chestnut blight, caused by *Cryphonectria parasitica*, arrived in the United States in 1904 and within 50 years had completely wiped out mature chestnut (*Castanea dentata* (Marsh.) Borkh.) stands throughout the eastern United States (Anagnostakis 1987). Other devastating non-native insect and pathogen pests in United States forests include Dutch elm disease (caused by *Ophiostoma ulmi*), sudden oak death (SOD; caused by *Phytophthora ramorum*), emerald ash borer (*Agrilus planipennis*), white pine blister rust (*Cronartium* *ribicola*), hemlock wooly adelgid (*Adelges tsugae*), and Port-Orford-cedar root disease (caused by *Phytophthora lateralis*). Although individual management actions can spot eradicate, slow the spread, or contain NNIS after arrival, costs and manpower are usually astronomical and often prohibitive (Moser et al. 2009)—especially when compared to the costs of heading non-native pests off at the port.

Despite the advantages of a proactive, exclusionary stance toward NNIS, the American system of pest exclusion is anemic. A strong system of exclusion would enable countries to engage in what economists disparagingly call "protectionist" activities, e.g., to reject materials strongly suspected of or shown to be harboring NNIS. This would require a framework of laws that enable flexibility in import inspection and rejection, including outright forbidding of particular types of goods considered extremely high-risk for NNIS, such as container-grown plants in growing media or wood packaging materials. Moreover, a strong exclusionary system would prioritize the inspection process by providing adequate facilities and personnel to oversee the process. The United States, like many other modern countries, has elected to do very little of this. In general, the American plant inspection and exclusion system is plagued by a paucity of plant protection staff, an insufficient number of pest inspections performed, low pay for inspection personnel, and insufficient equipment and facilities for plant species holding and identification (Reaser and Waugh 2007). Simberloff (2006) mentions this underbudgeting as a key reason for certain U.S. Department of Agriculture, Animal and Plant Health Inspection Service (USDA APHIS) functions having been transferred to the Department of Homeland Security, even though this did not solve the problem. McCullough et al. (2006), citing National Research Council (2002), point out that only 2 percent of high-risk cargo targeted for inspection each year can be inspected by APHIS personnel.

Beginning in the 20th century, and particularly during the past 20 years, the United States corporate capitalist class has collaborated in the creation of a system of global "free" trade that valorizes porous borders and penalizes protectionism in any form. Many political economists call this laissez-faire capitalistic economic philosophy "neoliberalism"; although the term covers a wide range of possible economic, political, and philosophical developments and should be used with caution, its general ideas as widely understood today underlie such economic innovations as free-trade zones, fiscal austerity, World Bank and International Monetary Fund-imposed structural adjustments, and extensive market-oriented deregulation (Boas and Gans-Morse 2009, Harvey 2005). In terms of pest exclusion, the most damaging such innovation has been the gradual eclipsing of American authority to regulate potentially harmful imports through quarantines and inspections by the ascendancy of the principle of non-interference with trade. In 1912, the introduction of white pine blister rust stimulated the U.S. government to formulate the Plant Quarantine Act, which established federal control over inspections and sanitary measures related to the movement of plants and animals into and out of the country and between states (Aukema et al. 2010, Weber 1930). This act was modified and held in force as "Quarantine 37" for most of the 20th century, giving the government wide latitude over rejection of potentially harmful imported materials, but the formation of the World Trade Organization (WTO) in 1990 began its *de facto* dismantling.

Article 5, no. 4 of the Agreement on the Application of Sanitary and Phytosanitary Measures by the WTO, negotiated in Uruguay and ratified in 1994, subjects all environmental protection considerations to the principle of non-interference with trade. Members have the "right" to develop phytosanitary measures, but only so long as they are consistent with the agreement, which generally states that, should the need for phytosanitary measures arise within any two member states, the members must bilaterally agree on the scope and substance of the measures. There are exceptions to the articles for urgent situations, but these exceptions cannot be extended indefinitely. Moreover, to implement such an exception, members cannot justify the exception politically or morally, i.e., with an appeal to a precautionary principle for environmental protection. Such an appeal is interpreted by the WTO as a "disguised restriction" on trade. Rather, members must marshal scientific evidence supported by a formal risk assessment in order to implement emergency phytosanitary measures. The onus of proof that such measures are necessary falls upon the receiving, not the exporting, country. It is interesting to note that the agreement is built on a political priority (free trade), but the political nature of this preference is couched as an appeal to an apolitical authority (science) (World Trade Organization 1994).

One of the original assumptions of Quarantine 37 was that imported stock would only be used to establish domestic propagation operations, not for direct resale. But under the 1994 Agreement, this practice is a disguised restriction on trade. Therefore, the United States cannot require that only seed be imported for development of ornamental plant stock or that all ornamental plant stock be propagated and grown within United States borders. WTO rules also prevent the United States from keeping a "black list" of plant genera that are known to be associated with dangerous pests or groups of pests, although some WTO member countries, such as Australia, have been willing to court WTO disapproval by maintaining such lists (Keller et al. 2007). Recently, APHIS has developed a gray list of plants that are provisionally banned pending further study (Liebhold et al. 2012), but in general, the United States has not been willing to maintain any black lists.

As the above example shows, governmental adherence to the WTO-controlled phytosanitary system varies from country to country, even among WTO members. Using examples from the United Kingdom and Australia, Keller et al. (2007) show that phytosanitary strictness and laxness even varies within the same country according to commodity. In Australia, for example, domestic apple producers have been successful at restricting apple imports from New Zealand because the bacterial disease fire blight is found in New Zealand apple orchards but not in Australia, whereas Australian beef producers were not ultimately successful in securing wide-ranging restrictions on beef imports from the United States even when concern about mad cow disease (present in the United States) was very high in Australia. According to Maye et al. (2012), these cases exemplify the asymmetry of global economic power relations, showing that phytosanitary measures may be based upon objective science, but tend to stray from it depending on how the regulating country construes the economic risks and opportunities that hinge on these measures—a balancing act between neoliberal economic development and biosecurity. Indeed, this pairing of cases is an extreme example. Because mad cow disease presented a potent human health threat, the contrast between Australia's weak position on excluding beef imports from the dominant world power and its strong position regarding excluding apple imports (not a human health threat) from a complacent regional partner amply demonstrate the ascendancy of geopolitical considerations over domestic biosecurity.

Insufficiency of Science

Scientific expertise is clearly necessary for the formulation of effective phytosanitary measures, but it is also clearly insufficient. Although the formal risk assessment procedure mandated by the WTO may initially seem reasonable as an adjudicator of whether stringent exclusion measures are necessary, these assessments often take too long (sometimes years) to make them meaningful. Pest risk assessments can give an idea of a pest's likelihood of establishment in a non-native environment, and sophisticated modeling can often predict post-establishment NNIS spread with surprising precision, but these assessments do not, as a rule, balance competing economic interests or account for so-called "externalized" environmental costs such as the cascading ecological effects of the extirpation of a keystone plant species, aesthetic losses, or spiritual losses (Brasier 2008, Perrings et al. 2005).

Moreover, many non-native forest pests are unknown to science until they have already become established and are causing damage in their new homes (Brasier 2008, Roy et al. 2014). This was the case, for example, for two newly introduced and extremely destructive forest insects from eastern Asia, polyphagous shot hole borer and Kuroshio shot hole borer (unnamed species of *Euwallacea*), which were unknown before entering southern California, where they have decimated extensive stands of riparian hardwood trees (Boland 2016). It was at first assumed that both insects were the tea shot hole borer, *Euwallacea fornicatus*, until extensive genetic study well into the insect outbreak determined that they are actually related, separate species. Many pest species may not be harmful in their home ranges, where they have co-evolved with their host trees and cause only minor, inconspicuous damage. This is the case with *P. lateralis*, an oomycete pathogen responsible for extensive mortality of Port-Orford-cedar (*Chamaecyparis lawsoniana* (A. Murr.) Parl.) in California and Oregon. This pathogen was first found in the United States in 1923, but was only traced to its probable center of origin in 2010, when it was discovered in Taiwanese forest soils with a close relative of Port-Orford-cedar (Brasier et al. 2010). In each of these cases and many others, a fully developed scientific understanding of the pest, its damage potential, and its pathways of introduction did not emerge until well after the pest had become established in the United States. In the case of the southern California shot hole borers, this knowledge is still developing, and there is nothing to suggest that these boring beetles will not spread northward through the state. These examples demonstrate the reactive nature of current plant pest detection systems: when the burden of examining pests is postponed until they escape, establish, and cause a problem, the receiving country's natural environment becomes, over and over again, a *de facto* laboratory for large-scale, uncontrolled experiments in pest pathogenicity, aggressiveness, and/or virulence.

All phytosanitary measures are based on acceptable risk, since no government can spend the money or time required to exclude or inspect every traded article. It is clear from the above examples that basing pest exclusion measures on scientific considerations without adducing additional political or moral concerns as part of biosecurity regimes poses a very high level of risk. As an additional example, consider the case of Native American tribes in northwestern coastal California, whose ancestral lands are being invaded by *Phytophthora ramorum*. To these tribes, the acorns of tanoak (*Notholithocarpus* d*ensiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oho), the primary susceptible host tree, have provided a principal food source for thousands of years (Bowcutt 2013), and tanoak along with other plants that host this pathogen have great spiritual importance—unlike traditional natural resource extraction markets, which recognize tanoak as a species that competes with valuable timber trees (Alexander and Lee 2010b). Although this spiritual and foodway importance is widely if informally recognized, this recognition has not enabled the tribes to garner additional money or efforts to fight the pathogen. The California and United States governments primarily provide environmental protection by regulating formalized intra-and interstate commercial activities such as timber harvesting and agriculture. No California agency is tasked with proactive intervention, especially based on spiritual or quality of life concerns, to alleviate forest degradation across administrative boundaries. It may be that most world governments that participate in the current free-trade system are comfortable with this kind of risk, in which vulnerable constituencies who locate value elsewhere than in the market suffer damage because of economically- and scientifically-based decisions made elsewhere. But such risks do not only pertain to the exclusion phase of pest management; the insufficiency of science as a guide to action extends also to post-invasion biosecurity measures.

Political Complexities in Post-invasion Management

This paper has concentrated on the politics of pre-invasion pest management (pest exclusion), but we should briefly mention the many non-scientific factors that determine the success or failure of post-invasion pest management efforts. These include the following:

1. *Funding*. Nothing articulates environmental priorities, and lays bare their essentially political nature, quite so loudly as a government budget. In California alone, numerous trade and public safety priorities trump forest health protection. Whether this is good or bad is immaterial; we simply point out that this priority ranking is essentially based on moral, economic, or social considerations, not on science.
2. *Time*. The grant funding and money appropriation mechanisms that underlie most pest management activities do not lend themselves either to the quick response that is necessary to eradicate most pests before they become established in an area or to the long-term, persistent efforts necessary to contain pest species after they become established in an area. Although much

of this is inspired by an ethos of good stewardship of public funds, it also displays to some extent the capriciousness of public opinion about biological emergencies. For example, in 2009-2010 alone, the California Department of Forestry and Fire Protection allocated $519 million to the base budget for fire operations, with an extra $182 million emergency "overflow" budget (Donald 2009); for SOD, a generous estimate over the decade-long research and management programs budgeted by both federal and state agencies up to that point—exclusive of USDA APHIS's considerable budgetary allocation—is 30 to 50 million dollars (Alexander and Lee 2010b). A stable and extensive funding base enables a focus on quick emergency response in the case of wildfire, and it has enabled the development of a vast wildfire response infrastructure of people and equipment. Both situations represent biological emergencies warranting extreme concern; both threaten human infrastructure and human life; but the differences in time scales and visibility between them assure that only one assumes a dominating position in the minds of most Californians.

3. *Territory*. Forest pest management success often depends on who controls legal, bureaucratic, and social territory. For example, the California Department of Food and Agriculture was unable to deploy mating disruption treatments to control the light brown apple moth, which feeds on many different kinds of trees, because a very vocal and politically influential public in the San Francisco Bay area interpreted these treatments as chemical pesticides (Garvey 2008)—an example of government's losing control of the social territory surrounding forest health and chemical use. As another example of the power of controlling discursive territory, SOD largely gained attention as a problem because it first appeared near the homes of affluent communities with expertise in leveraging their local, state, and national political representation (Alexander and Lee 2010b). This extends to geographic territory as well, where the success of pest control often depends on the willingness of landowners to engage with the problem. Politically-determined geographic boundaries often over- or under-regulate pests, and arbitrarily or historically drawn boundaries are usually not appropriately matched to the scales of NNIS invasion and spread (Thompson et al. 2016).

The Coast Redwood Region: a Case Study

The forests of California's north coast (and southwestern Oregon) suffered several major forest pathogen invasions in the 20th century, including *P. lateralis* in the 1950s (Zobel et al. 1985), *Fusarium circinatum* (cause of pitch canker) in the 1980s (Camilli et al. 2013), and *P. ramorum* in the 1990s (Rizzo et al. 2005). More recently, as mentioned earlier, several different species of bark and ambrosia beetles have invaded southern California forests and are poised to move northward throughout the ranges of their various hardwood hosts. The *Phytophthora* invasions to California have been linked to the trade in ornamental plants, while pitch canker arrived from the southeastern United States via an unknown route. In most of these cases, the invaders were unknown to science prior to the invasion, and large amounts of money and time have gone into biological study and adaptive management trials. The appearance of these invaders testifies to the risks presented to California's forests by their exposure to its large urban centers and intensive economic activities. In all cases, there has been fear that pathogen presence over a long enough period of time will result in the extirpation of the pathogen's primary host (for *P. lateralis*, Port-Orford-cedar; for *F. circinatum*, native stands of Monterey pine; for *P. ramorum,* tanoak and coast live oak [*Quercus agrifolia* Née]), although further study has revealed unexpected pockets of survival and the presence of resistance among some host populations.

The moist microclimates present within the coast redwood belt and extending to the ocean present a generally favorable environment for these non-native pathogens to survive and spread. Importantly, each pathogen infects host trees of secondary economic importance, and so the invasions have not yet stimulated the kind of widespread political action that would lead to changes in the established exclusion protocols or augmentation of infrastructure and personnel at either the national or state border inspection facilities. Political ecologists would call attention to (1) the ecological damage caused by the loss of rare species (e.g., Port-Orford-cedar) and keystone species (e.g., tanoak) as well as to (2) the ways in which the current prioritization of economic importance has marginalized the communities of people who depend on these tree species. As mentioned above, although tanoak is known primarily as a competitor to trees managed for timber values and is difficult to manage for timber values itself because of its particular wood properties, it is hard to overstate its importance as a food source and spiritual symbol to northwestern California Native American tribes (Alexander and Lee 2010a).

Even given the gradual diminishment over time of the timber industry's contributions to the California economy, the stately conifers of the redwood belt, especially coast redwood and Douglas-fir, (*Pseudotsuga menziesii* (Mirb.) Franco), are still a foundation resource in many ways: they contribute to economic, aesthetic, recreational, and spiritual values for millions of people. If a NNIS capable of killing these conifers *en masse* were to enter the state, one wonders what the political response would be, and whether this would be enough to catalyze a reconfiguration of the porous United States trading system.

Conclusion

This discussion has offered only the briefest of forays into the complexities of political ecological approaches as they apply to understanding the management of NNIS. As a relatively unified group of thinkers, political ecologists carry their own political affiliations, usually on their sleeves. They tend to be highly critical of neoliberal ideology and capitalism in general, and they try to draw attention to groups that are marginalized when governments use science as a cover for what are really political decisions. We do not have to subscribe fully to the politics behind this approach in order to recognize that it is calling attention to something important: appealing to the objective authority of science as a foundation for environmental decision-making can often be a red herring. In NNIS management, in particular, it is fairly clear that we need a stronger system of plant exclusion and inspection to keep harmful pests within their proper continental and regional borders, yet we continue to avoid dealing with this issue by keeping serious discussion in the technocratic-scientific realm.

We are by no means arguing that scientifically informed post-establishment management is unnecessary; on the contrary, insufficient money and effort is almost always allocated in that direction too. But the case is much more black and white in the case of exclusion than in that of post-invasion management. We believe that national governments need to work harder to find ways to accommodate both robust economic exchange and more thorough inspections/stronger national control of potential pest-carrying imports. At times, some protectionism and some restrictions on trade may be warranted in the service of the integrity of global forests. Drawing these lines is an essentially political activity: the rapid proliferation of a vocabulary of "science-based decisions" and "science-based risk assessments," along with continued dependence on quantitative science as a foundation for decisions and assessments rather than an adjunct to a more general prioritization of environmental principles, show that this is not generally recognized. Nevertheless, until forest managers and scientists do recognize it as such, they will be ill-equipped to participate in the process, or worse, they will be unwitting tools of the political establishment.

This issue should be of direct concern to those who live in regions like California that contain high numbers of endemic plant species. Because of their size and historical importance, coast and Sierra redwoods are rock stars of world forests. It would be a crushing blow to the state if these emblems of California's natural environment—or any of the myriad other compelling and rare tree species that call the state home—were extirpated by a hitherto unknown insect or pathogen hitchhiking from another continent. California is particularly vulnerable to such an invasion because of its major ports of trade and strong economy, as is shown by the problems we already have with invasive pests such as *P. ramorum*, *P. lateralis*, goldspotted oak borer (*Agrilus auroguttatus*), and the shot hole borers. Whatever their opinions may be about these issues (there is no doubt a healthy diversity), we call upon forest managers and concerned citizens throughout the region to intensify and enrich their conversations with state and national administrators and legislators. We think it wise to proactively move this conversation into the political arena by educating our key policy-makers now rather than to wait for another irreversible and devastating pest invasion.

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SESSION 7 – Policy, Economics and Community Forestry

Socioeconomics of the Redwood Region[1]

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Abstract

We compiled data from federal, state, and private academic databases to characterize the changing socioeconomics of the redwood region, which is part of the broader geography of the American West. The American West has turned economically away from commodity markets such as timber toward an economy dependent on knowledge and innovation, with job growth in service industries. We illustrate this shift by first comparing two distinct areas of the redwood region, the non-metropolitan northern counties (Del Norte, Mendocino, and Humboldt) and the metropolitan southern counties (Sonoma, San Mateo, and Santa Cruz), on variables including employment, income, and education. These two areas display dramatically different levels of financial and human capital, and represent two very different aspects of the American West. One illustrative distinction is that the northern counties have maintained a forest products industry, while the southern counties have turned almost entirely to other sectors. We then profile the role of the forest products sector within the regional economy, and how it has changed in terms of markets, wood sourcing, and infrastructure. We include trend data for mill capacity, wood prices, and export markets.

Keywords: forest economics, forest sociology

Introduction

The people of the redwood region are part of the broader geography of the "New" American West, which has shifted from commodity markets such as lumber, toward an economy dependent on "people's knowledge, skills, and innovation," with job growth in service industries such as health, professional, and technical services and jobs in finance, insurance, and real estate sectors (Gude et al. 2012, p. 420). Many areas of the West have experienced growth as a result of amenity migration, in which people migrate to a region because of its recreational opportunities and natural beauty (Gosnell and Abrams 2011). Amenity migration and shifting economic sectors are two components of what has been termed "rural restructuring," along with altered human-land relationships, from extractive or productive land uses to consumptive (aesthetic, recreational, and conservation) land uses (Nelson 2001).

In this paper, we investigate the social and economic characteristics of northern California's redwood region. We divide this region into two parts: the northern counties (Del Norte, Humboldt, and Mendocino), notable for their relatively intact forest products industry and low population density, and the metropolitan southern counties (Sonoma, San Mateo, and Santa Cruz), typified by extensive exurban development. While several other counties have redwood forest land, we only included data from counties with over 5 percent in redwood forest. Stewart (2007) noted the three northern counties contain the majority of redwood acres, but the southern counties have large populations in which redwood forests are valued as open space and for recreation. Our objective was to compile baseline and recent trend data regarding 1) county-level demographics, such as employment, income and poverty, and education; and 2) the role of forestry within the regional economy, including how forestry has changed in terms of infrastructure, wood markets, and wood sourcing. The data were gleaned from several sources, noted in the figures. Demographic data were

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compiled through the Headwaters Economics Economic Profile System, which collates data primarily from federal sources. Unless otherwise indicated, timber industry data were collected by the Bureau of Business and Economic Research (BBER) at University of Montana, which conducts periodic censuses of primary wood products manufacturers across the western United States, including the redwood region.

Demographics of the Redwood Region

The three southern counties (SC) are about six times larger by population (1.5 million) than the northern counties (NC) (250,554). Both the NC and SC grew in population from 2000 to 2014 (fig. 1). However, their rates of growth (between 1.6 percent and 7.2 percent) were lower than the growth of the United States and California, which grew by 11.6 percent and 12.4 percent, respectively, over this time period.

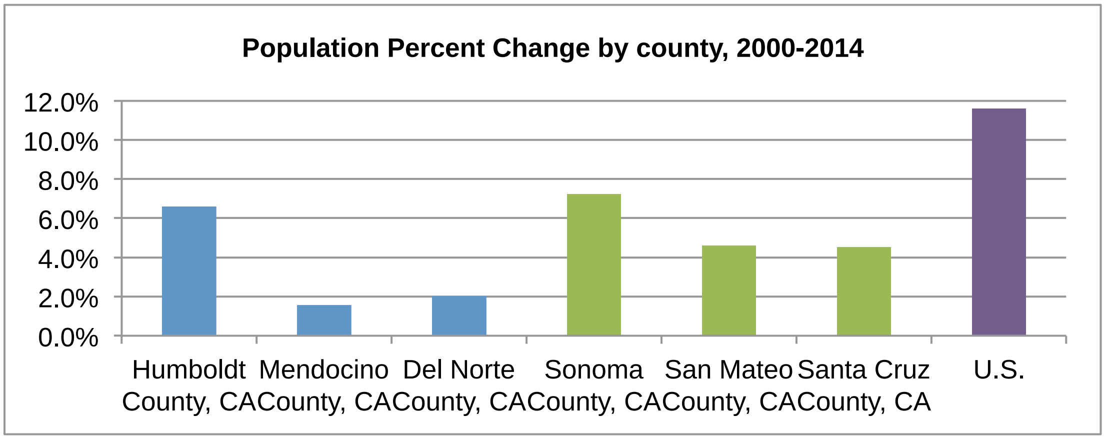


Figure 1—Population percent change from 2000 to 2014. Data compiled by Headwaters Economics; sources: U.S. Dept. of Commerce, Census Bureau (2015).

Economic Characteristics of the Redwood Region

The southern counties of the Redwood Region account for 88 percent of all employment in the region, and nearly all of the new growth is located in the southern counties—95 percent of new jobs since 2010 have located in the SC.

Virtually all of the new jobs are in services sectors, while non-services sectors have lost jobs during the last 15 years and make up a smaller share of the total employment base (fig. 2). The service sector consists of a wide mix of jobs, combining high-wage, high-skilled occupations (e.g., doctors, software developers) with low-wage, low-skilled occupations (e.g., restaurant workers, tour bus operators).[6] Non-services sectors consist of jobs in forestry, agriculture, construction, and manufacturing. Only including direct employment in forest industries, both regions have seen declines, though the NC still has significantly more people working in the timber industry (fig. 3). The timber industry, in this case, includes jobs associated with growing and harvesting timber, working in sawmills and paper mills, and wood products manufacturing.

[6] Despite the strong growth of employment in services, the term "services" is often misunderstood. The service sector typically provides services, such as banking and education, rather than creating tangible objects. However, some service sectors, such as utilities and architecture, are closely associated with goods-producing sectors.

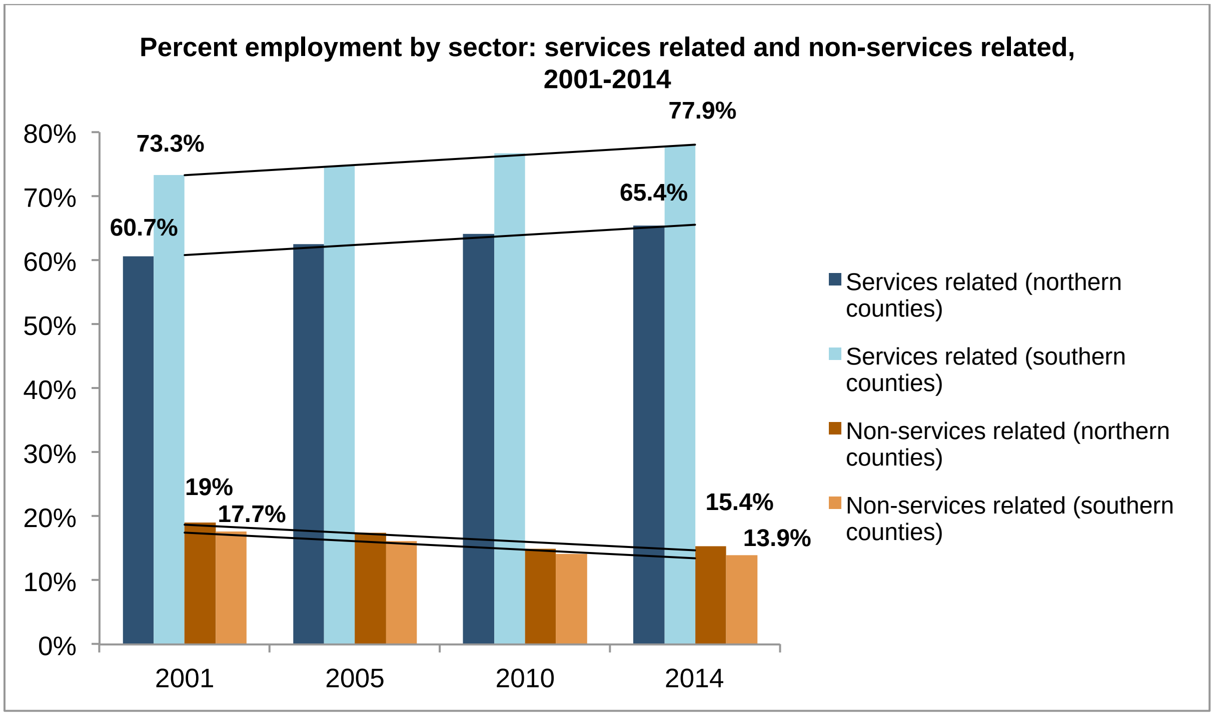


Figure 2—Percent employment by sector in the redwood region, 2001-2014. Data compiled by Headwaters Economics; sources: U.S. Dept. of Commerce. 2015. Bureau of Economic Analysis, Regional Economic Accounts, Washington, DC. Table CA30.

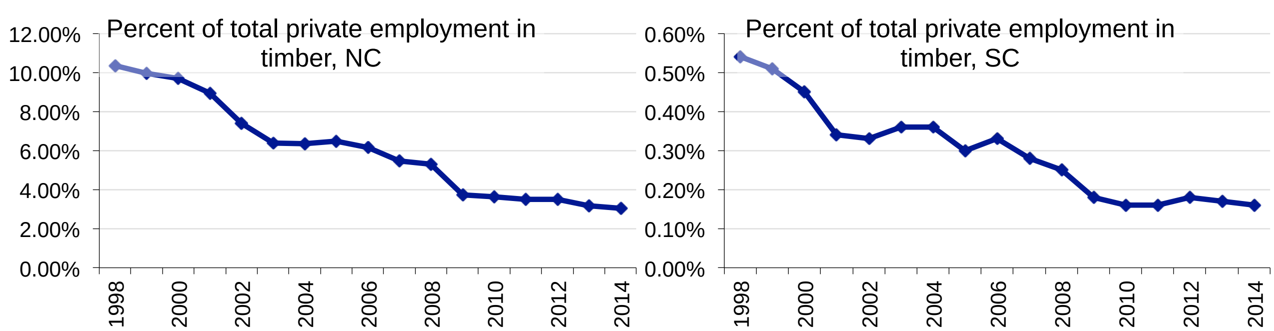
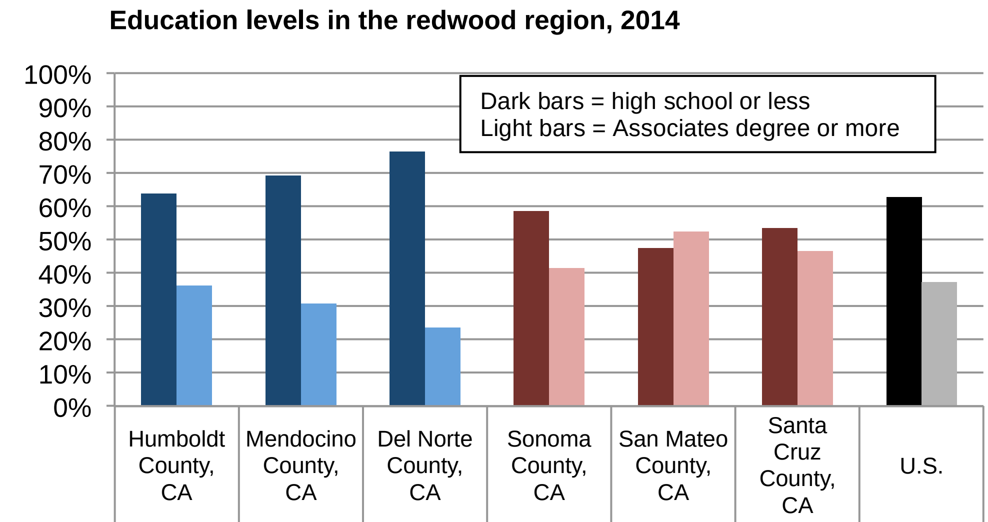


Figure 3—Percent of total private employment in the timber industry, 1998-2014. Data compiled by Headwaters Economics; sources: U.S. Dept. of Commerce. County Business Patterns, Washington, DC.[7]

On most socioeconomic indicators related to human and financial capital, the two parts of the redwood region displayed a bifurcated pattern, with higher levels of human and financial capital in the SC than the NC. This is evident in educational attainment (fig. 4). Residents of the NC had lower levels of education than residents of the SC, and the United States fell between the two.

[7] County Business Patterns data do not take into account the self-employed and are likely therefore underestimates of the total employment in the timber industry.



	Humboldt County, CA	Mendocino County, CA	Del Norte County, CA	Sonoma County, CA	San Mateo County, CA	Santa Cruz County, CA	U.S.
High school or less	63.8%	69.2%	76.5%	58.6%	47.5%	53.5%	62.8%
Associates degree or more	36.2%	30.8%	23.5%	41.4%	52.5%	46.5%	37.2%

Figure 4—Percent of adults by county with a high school degree or lower (dark bars), or Associates degree or higher (light bars). Data compiled by Headwaters Economics; sources: U.S. Dept. of Commerce. 2015. Census Bureau, American Community Survey Office, Washington, DC.

Per capita income and average earnings per job also demonstrated this bifurcation (fig. 5). However, this divergence occurred over time – real per capita income levels and average earnings per job were much closer in 1970, and over time have stagnated in the NC, while rising in the SC, particularly in the early 1990s (fig. 5).

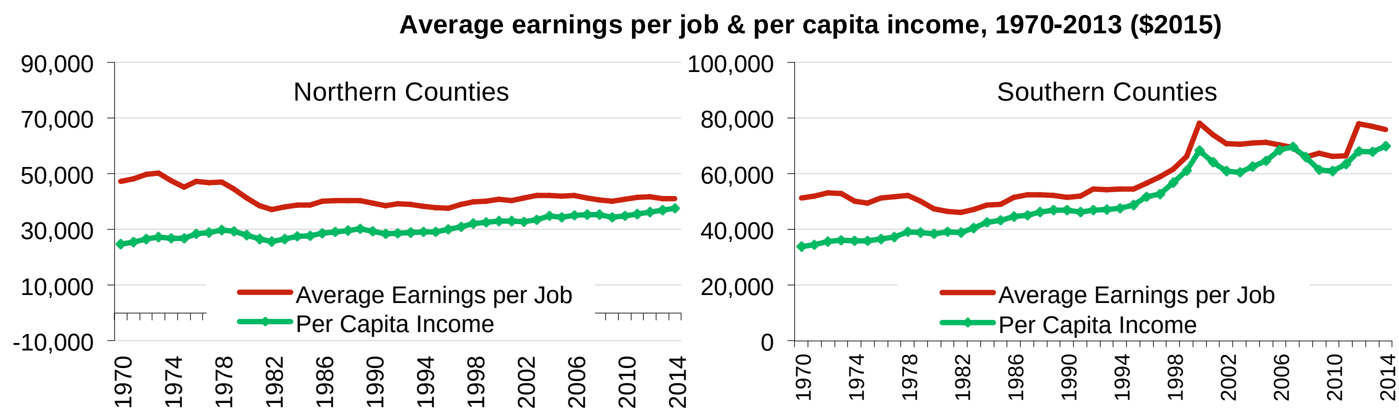


Figure 5—Average earnings per job (red) and per capita income (green) in $2015, from 1970 to 2013. Data compiled by Headwaters Economics, sources: U.S. Dept. of Commerce. 2015. Bureau of Economic Analysis, Regional Economic Accounts, Washington, DC.

This growing difference in earnings is likely due to the loss of high-paying manufacturing jobs in the NC, and the rise in high-paying white-collar jobs in the service sector economy of the SC.

One aspect of this distinction in socioeconomic levels was evident in the percent of non-labor income relative to total personal income (fig. 6). In both the SC and the NC, non-labor income as a percent of total income grew. In the SC, non-labor income grew by 9 percentage points between 1970 to 2014, from 27 percent to 36 percent; in the NC, non-labor income grew by 21 percentage points, from 28 percent in 1970 to 49 percent in 2014. The types of non-labor income differed, however. In the SC, a much higher proportion of non-labor income was from dividends, interest and rent (e.g., investments). In NC, a much higher proportion of non-labor income comes from transfer payments, which are generally hardship-related (e.g., Medicaid and welfare) or age-related (e.g., medicare and social security).

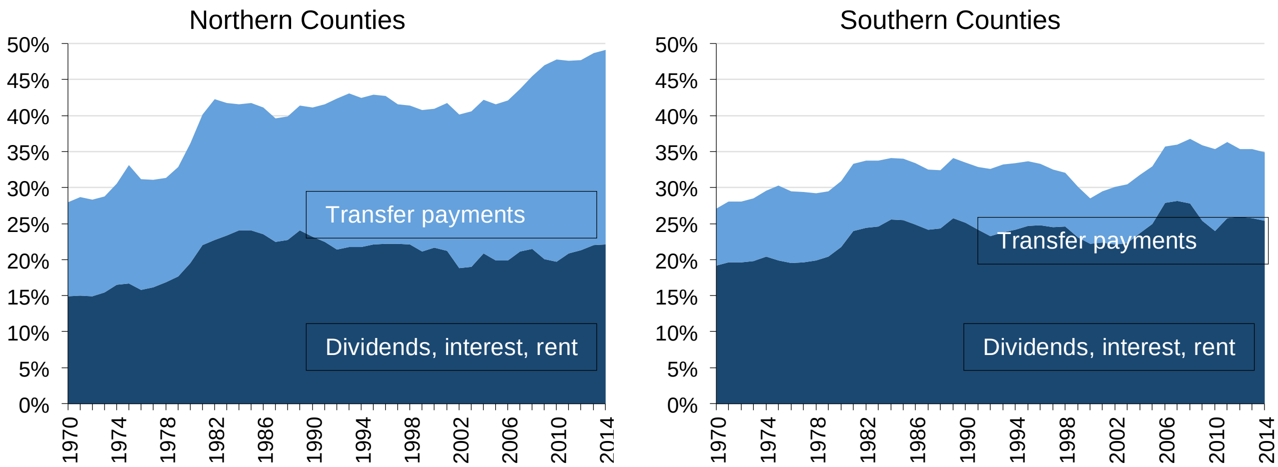


Figure 6—non-labor income as a percent of total personal income in the northern counties (NC) and southern counties (SC), from 1970-2014, including transfer payments (light blue) and dividends, interest and rent (dark blue).

Driving the uneven pattern of job and income growth between the NC and SC are dramatic structural changes in the United States economy in recent decades, affecting the economic opportunities for different types of counties. New jobs are being created in service sectors, the most important being a set of high-wage jobs in "innovation" sectors, including software, research and design, finance, and technology. High-wage service sector jobs create new wealth and support other sectors (e.g., they have multipliers that create additional jobs in related sectors) (Moretti 2012). Innovation jobs are locating in cities (and non-metropolitan areas connected to cities by airports that have access to finance, educated labor, and global markets). California's cities are competing successfully for these jobs and are driving the state's growth (Glaeser 2011).

Rural counties without easy access to markets or an educated labor force will not compete as successfully for these innovation jobs. They will remain more dependent on natural resources sectors. These sectors are volatile in price and production and subject to market and regulatory forces outside of California's full control, exposing rural communities to greater uncertainty over time. Manufacturing and timber jobs have also experienced significant productivity gains that have reduced the need for labor and stagnated wages in these sectors.[8]

Land Use and Ownership in the Redwood Region

In addition to being roughly three times larger than the southern counties, the northern counties also have a much higher proportion of forest land (71 percent vs. 33 percent for the SC), while the SC have more urban land, grassland, and shrubland (Miles 2016). Two metrics speak to the relative reliance on forests for their resource versus recreational value: the share of land in each region reserved from timber harvesting in the form of national parks, wilderness and national monuments; and the intensity of harvesting activities on those lands available to be harvested, termed timberland. Twenty percent of the acreage in the northern counties is reserved from harvesting activities, while 30

[8] U.S. Department of Labor, Bureau of Labor Statistics, "Productivity and Costs: Manufacturing and Mining Industries," 1987-2015. https://www.bls.gov/news.release/prin.toc.htm.

percent of the southern region is reserved (table 1). In addition, when comparing overall harvest volume in each region per hectare of timberland, the northern counties harvest nearly 30 percent more per hectare of timberland than do the southern counties (table 1). This difference indicates that landowners in the northern counties are more likely to actively manage their timberlands than the southern counties, where amenity values may be more important.

Table 1—Redwood Region total hectares, forestland hectares, and harvest volumes (BBER 2016, Miles 2016)

	Total (ha)	Forestland (ha)	Reserved (ha)	Timberland (ha)	Harvest (2012) (mbf)	Average harvest/ timberland (ha)
Northern counties	2,269,434	1,616,801	324,358	1,156,362	343,155	0.30
Southern counties	663,410	220,841	66,545	134,603	28,404	0.21

Counties in both the northern and southern redwood region are dominated by private land ownership (fig. 7). The one exception is Del Norte County, which has almost 70 percent of its total land ownership in public lands.

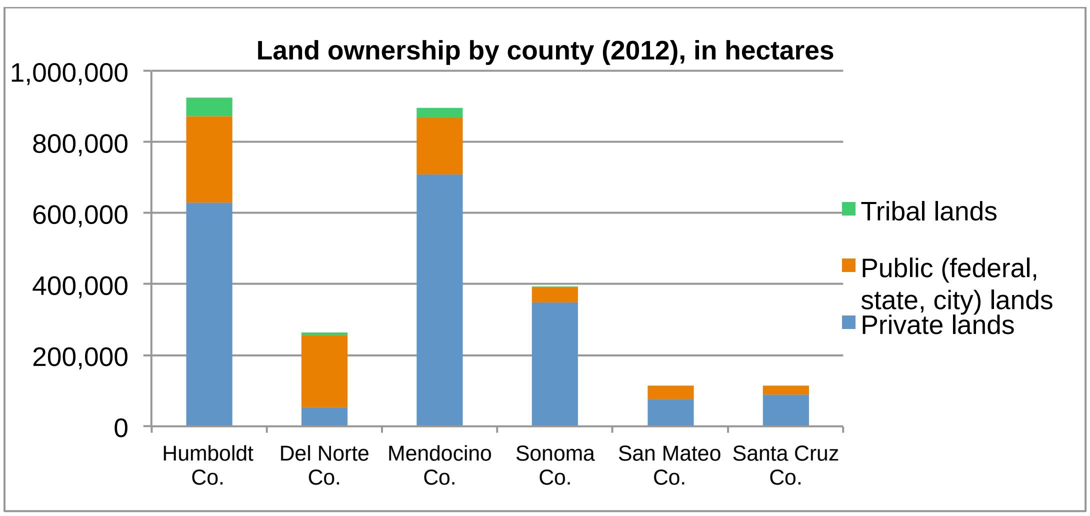


Figure 7—land ownership by county. Data compiled by Headwaters Economics; sources: U.S. Geological Survey, Gap Analysis Program. 2012. Protected Areas Database of the United States version 1.3.

Considering just the privately-owned lands of each county, the levels of urbanization and exurbanization are quite high. Urban and suburban development is defined as up to 1.7 acres per unit; exurban development is 1.7 to 40 acres per unit. When both of these forms of development are included, the counties of the SC have between 35 percent (Sonoma County) and 61 percent (Santa Cruz County) of the private land under development, compared to the NC, which has from 8 percent (in Mendocino and Humboldt counties) to 18 percent (Del Norte County) of the private land under development (fig. 8).

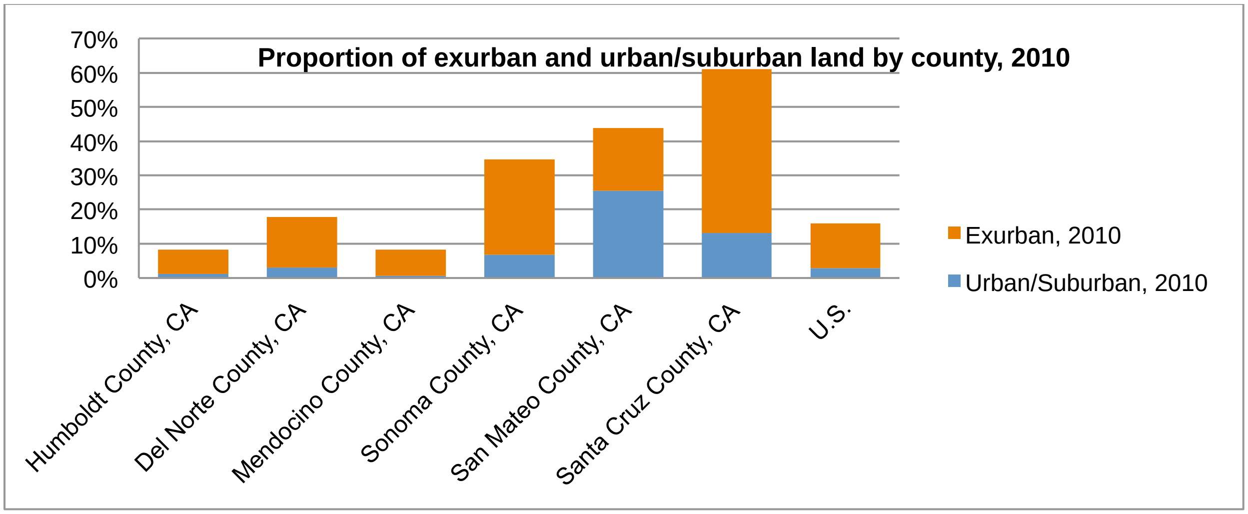


Figure 8—Percent of private land in residential development. Exurban development is defined as units from 1.7 to 40 acres per unit. Urban/suburban development is defined as units up to 1.7 acres.

Forest Industry in the Redwood Region

Figure 9 shows the relationship of timber harvest in the northern and southern counties, as compared to the rest of the state. Since 1978, statewide total timber harvest has ranged from a high of 4.7 billion board feet in 1988, to a low of just over 800 million board feet in 2009. Throughout this time period, the amount of timber harvest in the combined northern and southern counties ranged from a high of 40 percent of the total state harvest (in 1996) to a low of 21 percent (in 2009).

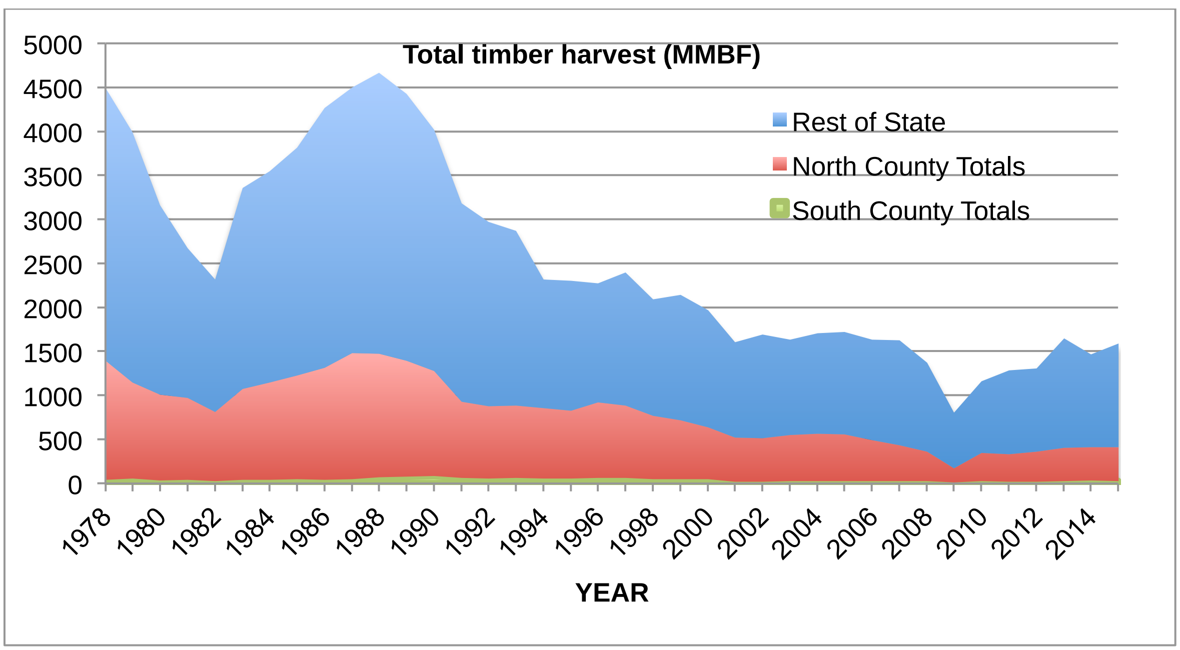


Figure 9—Total timber harvest in redwood region as compared to the rest of California, in million board feet, Scribner. Source: California State Board of Equalization various years.

The type of timber harvest has changed dramatically over the past 30 years. For example, in 1978, old growth represented almost 70 percent of the total timber harvest in the redwood region. This decreased to less than 20 percent by 1996. The State Board of Equalization stopped reporting old growth harvest by 2000 because the harvest was almost exclusively young growth timber by that time. In the redwood region, redwoods have consistently accounted for roughly half of the total harvested volume (BBER 2016).

Accompanying the decline of timber harvest in the region is the decline in mill capacity (fig. 10). Capacity to utilize raw timber across all types of wood products manufacturers (measured in board feet, Scribner) has experienced steep declines since the 1980s. In recent years, capacity has been relatively stable with very few mills dismantled and permanently removed from the manufacturing base. However, the share of capacity actually being utilized declined significantly as a result of the Great Recession, and it has been slow to rebound. Adding to the low rates of capacity utilization is the closure of a handful of larger mills in California in recent years—two of which occurred in the redwood region. As a result of these and other closures, capacity in the redwood region has declined by 31 percent just in the last decade and capacity utilization has dropped by nearly half from 75 percent in 2006 down to a low of 43 percent in 2016.

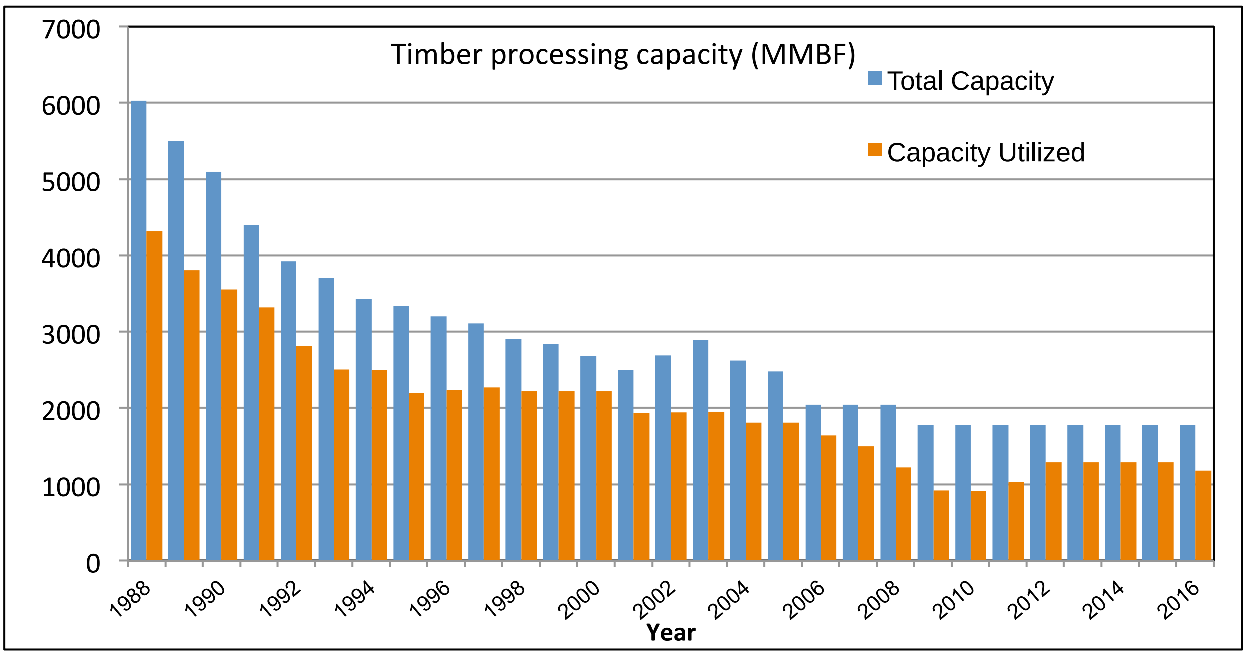


Figure 10—Redwood region timber processing capacity and use, in million board feet, Scribner 1988-2016 (BBER 2016).

Timber harvest in the redwood region takes place almost exclusively on private lands (BBER 2016). In the northern counties, the timber harvest has been heavily concentrated on industrial ownerships where they have provided upwards of 65 percent or more (table 1, fig. 11). In the southern counties, industrial and non-industrial private ownerships have each provided roughly half of the harvest over time (table 2).

Table 2—Redwood Region timber harvest by ownership class, selected years (BBER 2016)

Ownership	2000	2006	2012
		Million board feet[a]	
Private	**698.0**	**509.2**	**343.1**
Industrial	468.2	367.3	265.6
Nonindustrial private	229.8	136.3	69.4
Tribal		5.6	8.1
Public	**30.9**	**1.3**	**28.5**
National forest	8.0	1.3	3.7
State	22.8	0	24.7
BLM	0.1	0	0.0
Other public		0	0.1
Total	**728.9**	**510.5**	**371.6**

[a] Volume in Scribner Decimal C Log Rule, Eastside variant.

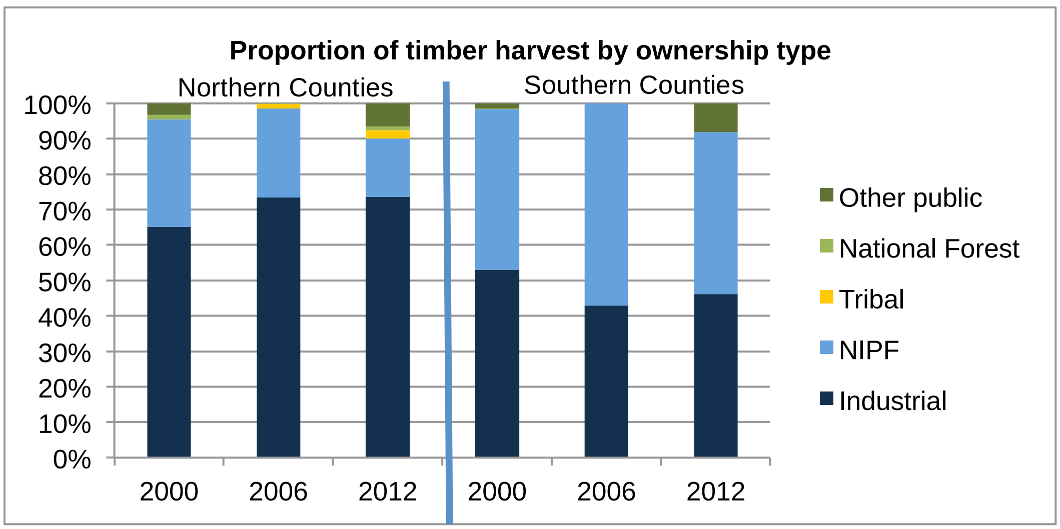


Figure 11—Volume of timber harvested (MMBF) in redwood region, redwood and all other species, selected years (source: BBER 2016).

Most of the harvested timber of the region was used for sawlogs, though a relatively high proportion (33 percent) of publicly-harvested timber was used for bioenergy (table 3). On private timberlands, only 2.3 percent of harvested timber was used for bioenergy.

Table 3—California redwood region timber harvest by ownership class and product type, 2012 (BBER 2016)

Ownership source	Sawlog	Veneer and other[a]	Bioenergy	All products
		Million board feet[b]		
Private timberlands	**332.7**	**2.4**	**7.9**	**343.0**
Industrial	255.2	2.4	7.9	265.5
Nonindustrial and Tribal	77.5	-	-	77.5
Public timberlands	**15.8**	**3.3**	**9.4**	**28.5**
National forests	0.4	3.3	-	3.7
Other public	15.4	-	9.4	24.8
Total	**348.5**	**5.7**	**17.3**	**371.5**

[a] Other product types include houselogs, firewood, furniture logs, and utility poles.
[b] Volume in Scribner Decimal C Log Rule, Eastside variant.

Capacity has exceeded timber harvest in the region as evidenced by the region's history of being a net importer of timber from other regions. This trend reversed slightly in 2012 when the distance timber traveled in all regions of California declined (BBER 2016). In general, most of the timber harvested in the region is being processed in the region (fig. 12).

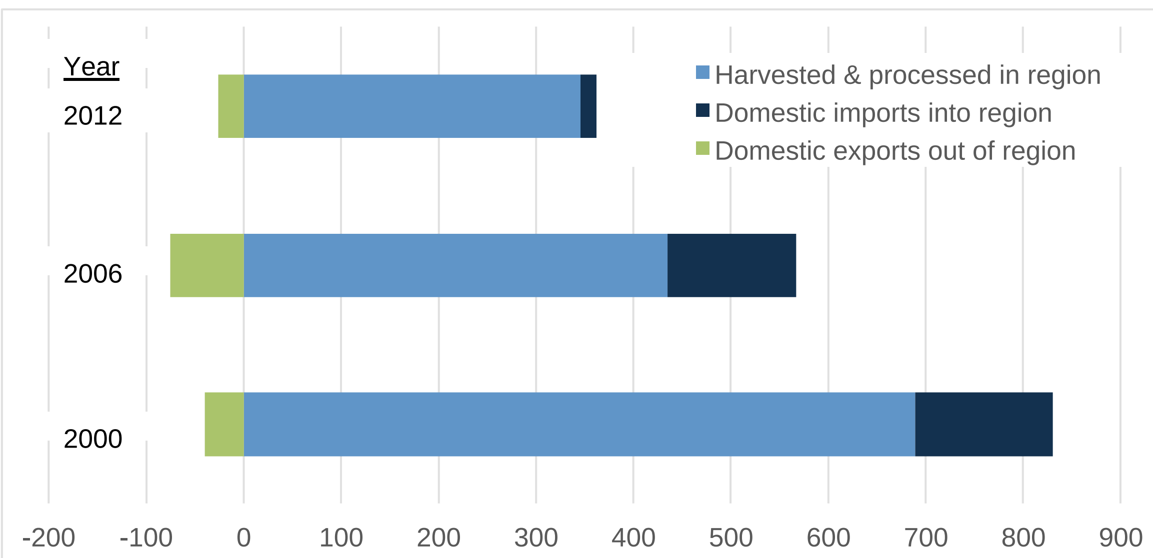


Figure 12—Volume of timber harvested (MMBF) in redwood region, redwood and all other species, selected years (source: BBER 2016).

Discussion

The redwood region mirror many trends of the American West and the United States economy as a whole, particularly in its shift in economic sectors away from commodity production (including forest products) toward service sectors. This shift has led to divergent economic opportunities between urban and rural places. Levels of human capital, as measured by federal indices, are generally low in the northern counties. This may not be surprising (it reflects the rural "brain drain" witnessed elsewhere) but it is a problem for these counties to address in an economy which increasingly values skills and expertise.

Several challenges are evident for rural western counties, including the northern counties. The challenges associated with over-reliance on single sectors, in particular those that fluctuate with commodity markets, such as the timber industry, were evident in the recent national recession (Gude et.al. 2012). Counties that were more timber-dependent tended to lose jobs at a faster rate during the recession. Since the recession, job concentration in metropolitan counties has accelerated as most new business formation is occurring in cities (Economic Innovation Group 2016).

Dependence on natural resources, and the risks dependence entails are further heightened by fiscal policies related to revenue from federal timber harvests. The northern counties historically received more than 10 million dollars annually in revenue sharing payments from the U.S. Department of Agriculture, Forest Service (USDA FS) and later from appropriated payments through the Secure Rural Schools and Community Self-Determination Act (SRS) and Payments in Lieu of Taxes (PILT) (Gorte 2010). More recently, SRS expired in 2015 and payments from the USDA FS will revert to revenue sharing payments equal to 25 percent of the gross value of commercial receipts earned from commercial activities on the National Forests. Sharp declines in the value of federal timber harvests mean that a return to revenue sharing will reduce overall payments to counties. In total, the three northern counties would see USDA FS and PILT payments decline to about 3 million dollars annually. As payments have declined, pressure has mounted on Congress to reauthorize appropriations or to reform federal timber management to maximize receipts. These options may each be unattainable for the redwood region. The decline in payments represents another challenge for rural counties seeking to stabilize and diversify their economies.

While the northern counties have maintained a forest sector that continues to supply a high proportion of the state's timber, there has been consistent decline in timber harvest and mill capacity in the region. Reasons for this decline include the exhaustion of profitable old-growth timber, increasing forestry regulation, and changing markets, such as the high costs of shipping from the redwood region and lower wood production costs elsewhere. Notably, non-industrial private timber harvest has declined from 230 million board feet in 2000 to under 70 million in 2012 (table 2). This decline in harvest is at least partly attributable to shifting landowner objectives away from timber production to home development, recreation, aesthetics, and other objectives.

This decline in the timber industry, however, does not account for the many non-timber forestry jobs that have been created in the region. The restoration economy provides an important source of jobs and revenue in Humboldt County (Baker and Quinn-Davison 2011). As the forestry sector expands to include jobs in carbon sequestration, watershed restoration, and other activities, it may be important to capture changes in forestry employment using more comprehensive definitions of the forestry sector. Currently, these jobs are not counted as "forestry" positions by North American Industry Classification System.

The southern counties display very different patterns, with high levels of human and financial capital as a result of their proximity to a major metropolitan area (San Francisco) and a booming technology sector. In the southern counties, extensive exurbanization indicates that relationships between people and land have shifted from productive uses to consumptive uses, wherein the land is valued for aesthetics and other non-productive purposes.

While this overview of the socioeconomics of the redwood region provides a broad picture of regional change, it could be improved in several key ways. We looked at county-level data, but smaller scales could paint a different picture, demonstrating variability within the northern and southern county sections. For example, the low levels of human capital within the northern counties may not be evenly distributed. Even more importantly, one of the largest sectors of the economy of the northern counties, cannabis production, has been excluded from our consideration. Because cannabis is illegal under federal law, data for the sector are difficult to obtain and virtually impossible to integrate with other sectoral data.

The economic opportunities for the counties in the redwood region will continue to diverge, and economic development policies should also be sensitive and targeted to the types of opportunities that exist in different counties. The profound shift in how and where our economies generate value, jobs, and income represents an opportunity for California's cities, but challenges for rural parts of the state that remain more reliant on sectors that have shed jobs due to productivity gains, increased competition, and challenging regulatory environments.

Acknowledgments

We wish to thank the reviewers of this paper, who helped to improve its overall quality and readability.

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A GIS Approach to Identifying the Distribution and Structure of Coast Redwood Across its Range[1]

Peter Cowan,[2] Emily E. Burns,[2] and Richard Campbell[2]

To better understand the distribution and current structure of coast redwood (*Sequoia sempervirens* (D.Don) Endl.) forests throughout the range and how it varies by land ownerships, the Save the Redwoods League has conducted a redwood specific analysis of a high resolution forest structure database encompassing the entire natural coast redwood range. Using this analysis, we are able to identify those areas most likely to contain coast redwood and the probable stature of those forests. Previous League databases were built by expert evaluation of aerial photographs, requiring substantial time and monetary resources, limiting the frequency of updates. In this extended abstract we discuss approaches for using satellite based remote sensing tools to estimate forest composition and structure throughout the coast redwood range. We further explore some of the structure of redwood forests by region and ownership.

Database requirements include the ability to identify areas dominated by redwood consistently throughout the range from northern San Luis Obispo County, California to Curry County in Oregon. To aid in differentiating forest with restoration potential, the League also requires delineation of forest structure characteristics such as basal area, quadratic mean diameter (QMD), and trees per acre (TPA). The Oregon State University Landscape Ecology, Modeling, Mapping, and Analysis (LEMMA) team's Gradient Nearest Neighbors (GNN) structure geodatabase, fulfills these requirements http://lemma.forestry.oregonstate.edu/data/structure-maps.; Ohmann and Gregory 2002).

Our project area was defined as any Hydrologic Unit Code (HUC) 12 watershed, the smallest hydrological units comprehensively mapped by the U.S. Geological Survey, within 1.6 km (1 mile) of known existing natural redwoods (CALVEG 2004; Save the Redwoods League, unpublished data). When clipped to this region the Landscape Ecology, Modeling, Mapping and Analysis Gradient Nearest Neighbor (LEMMA GNN) structure map (hereafter LEMMA) contained 3,867 pixel classes, with each pixel corresponding to a 30 m x 30 m LANDSAT pixel (Ohmann and Gregory 2002). These pixel classes were classified into 24 species and structure classes. Forest species was biased toward redwood by first categorizing any pixel with > 10 percent of basal area of redwood as redwood, likewise any remaining pixels with > 70 percent of basal area douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), with > 50 percent of basal area tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh) were classified as those species and the remaining pixels as mixed conifer-hardwood.

Structure classification was based on the LEMMA data for TPA and dominant QMD. Structural classes are based on Spies and Franklin (1991), with the biomass accumulation stage subdivided into three classes and old growth replaced by 'large complex forest'. Large complex forests (LCF) are forests which have structural complexity similar to old growth forest, but may actually consist of largely second growth trees.This distinction is important, in part, because remote sensing is unreliable at determining forest age. To improve the identification of LCF, we first compared the pixel classes to the Save the Redwoods League old growth database (unpublished data) identifying those classes that accounted for a significant proportion of the area in the database (> 1 percent) and were specific to those areas (more than 12 percent of that class found within the old growth database). Other forest structures classes were defined as follows: canopy closure, 50 percent of stems < 20.3 cm (8 inches);

[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.

[2] Save the Redwoods League, 111 Sutter Street, 11th Floor, San Francisco, CA 94104.

early biomass accumulation (ba), QMD < 40.6 cm (16 inches) or 300 TPA and QMD < 81.3 cm (32 inches); mid ba, QMD < 61 cm (24 inches) or 200 TPA and QMD < 81.3 cm (32 inches); late ba, QMD < 81.3 cm (32 inches); and maturation, 50 percent of stems < 122 cm (48 inches). Figure 1 shows the classification results for the Redwood National and State Parks region of the range.

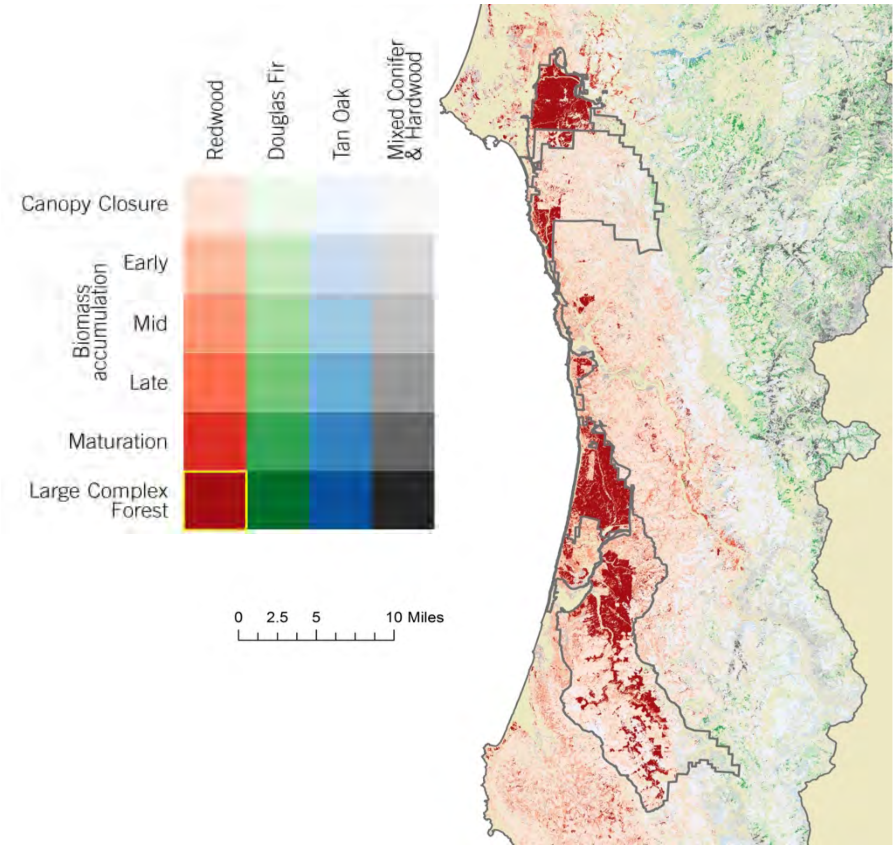

Figure 1—Example classification of the LEMMA species and structure map. The key describes four classes of species dominance and six structural classes. See text for classification details.

The resuling project boundary encompasses approximately 3.2 million ha (8 million ac), of which LEMMA identifies 2.1 million ha (5.1 million ac) as forested. As classified above 0.6 million ha (1.6 million ac) of the project area have redwood comprising greater than 10 percent of basal area, with 45, 729 ha (113,000 ac) being large complex redwood forest. Interestingly, nearly half of the LCF acres are aggregated into 14 complexes of 404.7 ha (1000 ac) or more. Futhermore over 75 percent of existing LCF is found in Del Norte, Humboldt, and Mendocino counties and more than 20 percent in the Bay Area counties, with comparatively little LCF south of Santa Cruz County.

We also explored the ownership of redwood forests throughout the range (fig. 2). Protected areas were idenfied as having "park type" protection it the California Protected Areas Database (CPAD (2015) database; we also identified state and national forests, tribal forests, as well as industrial and other private ownership types (unpublished data). While the majority of LCF exists in protected lands, large proportions are under industrial and other private ownerships, primarily as small isolated patches. Conversly, smaller structure redwood forests are widely distributed across the major ownership types within the coast redwood range, namely protected areas, industrial timberlands and other private landowners (fig. 2).

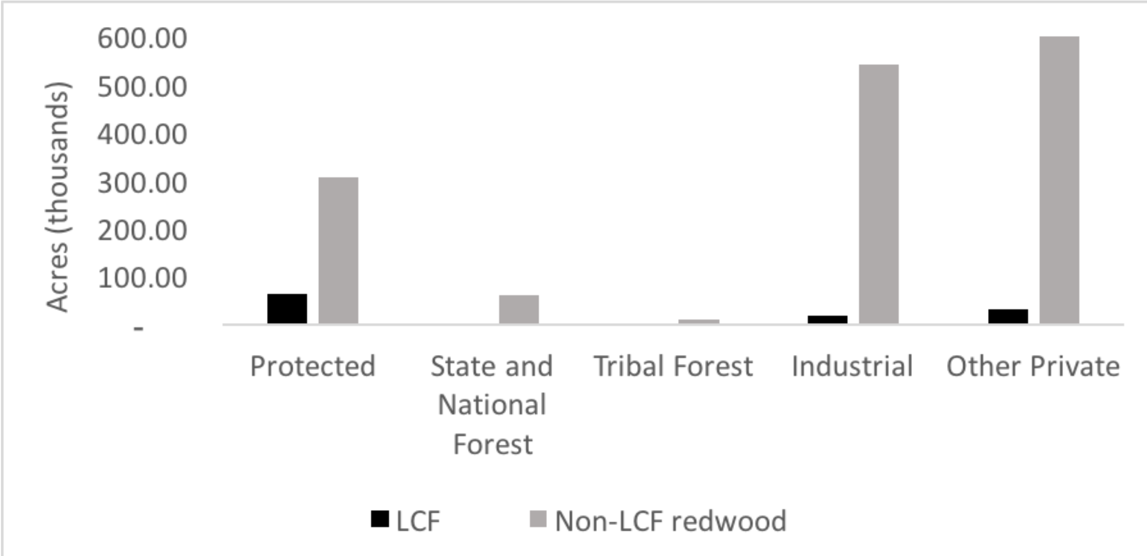

Figure 2—Acres of large complex forest (LCF) redwood forest and smaller stature redwood forests by ownership type throughout the coast redwood range.

Utilizing a high resolution species and structure map based on remote sensed data, we were able to identify the distribution of redwood forests and their structure throughout the range. The largest most complex redwood forest structure class is highly aggregated into several large complexes, mostly in protected areas and the northern portion of the range. Unsurprising, the extent of smaller structure classes is much larger and comprised the majority of redwood forests across all ownership types, including protected areas.

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The Listing of Coast Redwood as Endangered Under the IUCN Red List: Lessons for Conservation[1]

Erin Clover Kelly[2]

Abstract

In 2013, redwood (*Sequoia sempervirens* (D.Don) Endl.) was listed as endangered under the International Union of Conservation of Nature (IUCN) Red List. While this listing has no political or legal consequences for the management of redwood, it could have economic consequences as land and mill owners of the redwood region have sought to link redwood to sustainable practices in the marketplace. This paper argues that the listing of redwood, however, is fundamentally flawed by the metrics of the Red List, and that the listing misses the chief conservation challenges related to redwood, which center on ecosystem functionality, not continued existence of individuals in the wild. The IUCN, which maintains the most globally comprehensive list of threatened species, and which seeks to influence conservation actions, could address this flawed listing by creating multiple lists, including a "threatened ecosystem" list.

Keywords: conservation, endangered species, *Sequoia sempervirens*

Introduction

Concern about species extinction has increased in light of the profound consequences of human activities that have contributed to increasing extinction rates (Barnosky et al. 2011). Proposals to conserve species and biodiversity have ranged from local to global, with accompanying policy and logistical mechanisms at each level. At the global level, one of the most far-reaching efforts has been conducted by the International Union for Conservation of Nature (IUCN), an organization that has been compiling the Red List of Threatened Species (Red List) since 1964. The Red List informs conservation policies and planning by functioning as "a clarion call to action" and has become regarded as a tool that provides consistency in labeling species in need of conservation investment (Vie et al. 2008).

In 2013, coast redwood (*Sequoia sempervirens* (D.Don) Endl.), a commonly-utilized commercial timber species of the west coast of the United States, was determined to be endangered under the IUCN categorization system, meaning it is deemed to be facing "a very high risk of extinction in the wild." This paper is intended as a discussion piece, asserting that the listing of redwood as endangered is flawed and misstates the real threats to redwood ecosystems – which revolve not around the continued existence of the species, but to ecosystem functionality. The listing of redwood as an endangered species under the IUCN Red List criteria deserves the attention and critique of experts within the redwood region. Within this context, this paper aims to fulfill two objectives: 1) to inform researchers and managers of the redwood region of the listing, and 2) to raise questions about the listing of redwood, its legitimacy and potential consequences. I begin with an overview of the IUCN Red List and brief history of redwood conservation, detail the rationale for listing redwood as endangered, and consider the implications of listing redwood as endangered—for the IUCN Red List, and for redwood managers. I end with suggested modifications for the IUCN Red List, which is tasked with maintaining a scientifically credible list of globally threatened species that can be used by scientists and policy makers.

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[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California

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IUCN: Creating a Global List of Endangered Species

The committees of scientists who created and update the IUCN Red List have assessed a small percentage of the world's total species (~2.5 percent), but it is nonetheless the most comprehensive list of threatened species in the world (Vie et al. 2008). The Red List is updated yearly and available online. The list began in the 1960s as Red Data Books, and over time its categorization system has shifted in an effort to be scientifically defensible, objective, and useful for conservation purposes; and to create categories that are comparable across species and regions (De Grammont and Cuaron 2006, Mace et al. 2008). The Red List has been extensively cited by academics—there are over 1000 citations of the 2004 Red List alone, and many prominent studies have utilized Red List data to analyze global trends of biodiversity and conservation (for example, Brooks et al. 2006).

However, controversy and contention have surrounded some listings. Some have questioned the transparency of supporting documentation, going so far as to claim that analyses have "degenerated into assertion based on secret science" (Mrosovsky 1997, p. 436). Others have critiqued more narrowly the validity of certain listings, which potentially "detract attention from those populations that are truly threatened with extinction (Broderick et al. 2006, p. 25) and "create a scenario where limited resources may not be efficiently focused on those specific regions or populations that are declining and in need of rapid conservation action" (Godfrey and Godley 2008, p. 156; Seminoff and Shanker 2008).

Redwood Conservation: an Overview

Redwood conservationists of the late 19th century, often patrician visitors from urban places, were romantics who "advocated scenic protection" for isolated redwood groves; old growth redwoods were described as majestic, but valued much more highly for their timber as the urban centers of the West Coast were built (Schrepfer 1983, p. 6). The Save-the-Redwoods League was established in 1918 by academics who gained financial support from businessmen and professionals to purchase small redwood groves, and later established several redwood parks (Schrepfer 1983). These piecemeal conservation efforts were replaced in the mid- to late 20th century by concerted efforts at maintaining the last remnants of old growth, including boycotts, legal action, and direct action such as tree sits (Bevington 2009).

As of 2016, more than 90 percent of remaining stands of old-growth redwoods were located in protected areas, and privately-held second- and third-growth redwood stands were managed under one of the most stringent private forest land regulatory systems in the United States. Strategies for conservation have therefore shifted from preserving pockets of old-growth reserves to maintaining contiguous redwood stands with increased structural diversity. This involves protecting redwood forest around the old-growth patches, and working to increase heterogeneity in remaining stands, which lack multiple canopy layers and the complex crown structure found in large trees and needed by many wildlife species[3] (Lorimer et al. 2009, Van Pelt et al. 2016).

The IUCN Listing of Redwood

The protocols for listing species under the IUCN Red List have been updated seven times since 1990, most recently in 2001 (version 3.1). Current categories for listing range from "least concern" to "extinct," with three categories for Threatened species: Critically Endangered, Endangered, and Vulnerable (fig. 1).

[3] Correspondence with Emily Burns, Save the Redwoods League.

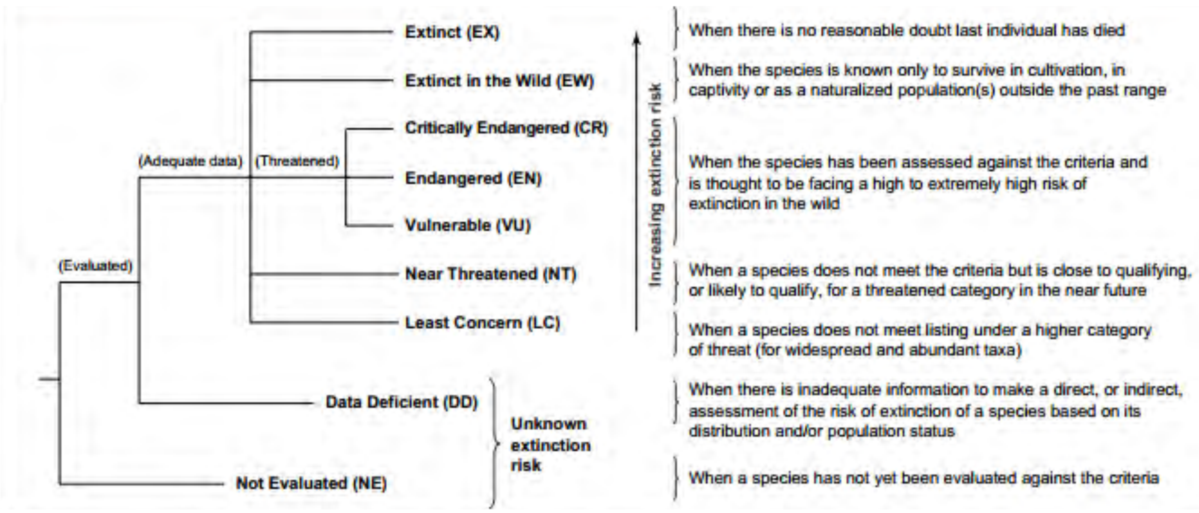

Figure 1—IUCN Red List categories, from Rodrigues et al. (2006).

Redwood was listed as endangered by the IUCN, part of an assessment of 615 conifers. Of those conifer species, 211 (34 percent) were listed as one of the three categories of threatened.[4] The overall framework of the IUCN Red List Version 3.1 is complex and beyond the scope of this paper, but the detail provided here is sufficient to understand the listing status of redwood. Redwood was listed as endangered under the categorization of *A2acd*—the significance of each of these designations is described in table 1.[5]

Table 1—Criteria or subcriteria for the listing of redwood as endangered under the IUCN Red List

Redwood listing criteria/subcriteria	Explanation
A: (criteria)	The population of mature individuals has been reduced.
2: (subcriteria)	The population of the species has declined by ≥50% over the last 10 years or three generations, whichever is longer. The reduction has not ceased, is not understood, or may not be reversible.
The subcriteria of a, c, and d refer to the basis for listing the species:	
a: (subcriteria)	The species was listed as a result of direct observation.
c: (subcriteria)	The species has had a decline in area of occupancy, extent of occurrence (the area of all of the known or potential present occurrence of the taxon), and/or quality of habitat.
d: (subcriteria)	The species was listed because of actual or potential levels of exploitation.

The Language of Listing

Though the categories for IUCN Red List Version 3.1 are clear, the reasons for listing redwood are much less clear. Claims of the IUCN conifer group are listed with direct quotations from the listing and a brief explanation of problems with the listing, based on available literature.[6]

The Decline of Individuals

According to the Red List criteria, the number of mature individuals has declined by more than 50 percent over three generations (table 1). Maturity is defined as "capable of reproduction." A

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[4] http://threatenedconifers.rbge.org.uk/.
[5] From http://www.iucnredlist.org/static/categories_criteria_3_1.
[6] http://www.iucnredlist.org/details/34051/0.

generation is the “average age of parents of the current cohort.”[7] In redwoods, three generations could be anywhere from hundreds to thousands of years. If the estimate of generation length is based on industrial rotations of about 60 years, then three generations could be 180 years. If second-growth redwood is considered, the “parents” would be old-growth redwood, which can live for thousands of years.

With either timeline, the decline is measured in terms of number of individuals. However, it is not clear that the number of mature individuals in fact has been reduced. The extent of redwoods is essentially unchanged though may be shifting slightly at its margins, with contraction at the southern end and enlargement at the northern end of the range (Sawyer et al. 2000a). In terms of number of mature individuals, second-growth redwood trees, which are smaller and occupy less space, may actually result in more individuals per acre than old growth. Young redwood stands (second or third growth) are “a mosaic of dense, stem-exclusion stands dominated by sprouting redwoods” (Thornburgh et al. 2000, p. 240). Thus the number of mature (in IUCN terms) redwood trees may not be diminished, though the structure of the forest is much changed.

The Purposeful Replacement of Redwoods With Other Species

IUCN Red List statement: *The proportion of redwood in commercially exploited forests containing this species is still declining, due to deliberate or accidental replacement by more competitive species in the early phases of succession after clear-felling, especially Pseudotsuga menziesii [Douglas-fir]”*

IUCN Red List statement: *Its late successional to climax dominance coupled with shade tolerance means it is easily replaced by more light demanding conifers such as Pseudotsuga menziesii* [*Douglas-fir]. This can be made ‘permanent’ if forests are chosen to be so managed, as indeed they tend to be in commercial forestry operations.*

These two statements appear to be the crux of the argument surrounding the IUCN claim that individuals of the species are declining. Redwood stands reach exceedingly old ages and can then maintain multiple ages within shaded stands (Busing and Fujimori 2002). But redwood is unusual among conifer species for its ability to reproduce via sprouting after disturbance, whether natural or manmade. Some authors have suggested that redwood requires disturbance (such as fires or floods) to regenerate (Barbour et al. 1980, Lorimer et al. 2009). In some second-growth stands, redwood has been proportionately diminished relative to other sprouting species such as tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh), but its widespread replacement by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) has not been documented, nor is it documented by the IUCN Red List. In fact, harvesting in the redwood region tends to lead toward continued redwood dominance, either through seeding or sprouting, though Douglas-fir will seed in stands that previously had the species (Boe 1975).

The IUCN Red List did not provide citations to support the claim that commercial operators in the redwood region are managing to favor non-redwood conifer species.[8] The prices for redwood exceed all other conifer species in the region, according to the California Board of Equalization, which places redwood values well above Douglas-fir (table 2) and other conifers. Other authors have noted the value of coast redwood, a species “highly valued for wood quality, rot resistance, and fast growth” (Jameson and Robards 2007, p. 171).

[7] http://www.iucnredlist.org/static/categories_criteria_3_1
[8] Follow-up emails with the listing author did not result in further citations.

Table 2—Harvest values for redwood and Douglas-fir timber, estimated by the California Board of Equalization (2016)

Species	Volume per log, board feet	Log size	Timber value, dollars (variation dependent on geographic region)[9]
Redwood	>300	Large	$570-$700
	150-300	Medium	$560-$600
	<150	Small	$490-$520
Douglas-fir	>300	Large	$100-$360
	150-300	Medium	$80-$330
	<150	Small	$60-$310

In addition to the economic incentives for commercial timber production, there are regulatory requirements for retaining and enhancing redwood reproduction on a long term basis. Most of the redwood region falls within the state of California, which regulates commercial forest operations under the Z'Berg-Nejedly Forest Practice Act of 1973. The California Forest Practice rules require replanting within 5 years following harvest. Further, the major industrial forest landowners in the redwood region have long term sustained yield plans on their properties providing further stability of redwood forests.

The Need to Maintain Old Growth and Regulate Logging

Under listed threats, the IUCN Red List highlights the need to "preserve" old growth, a suggestion that is unlikely to meet with much objection today.

IUCN Red List statement: *The conservation issues involving Sequoia sempervirens pertain largely to the necessary preservation of the remaining 'old growth' Redwood forest for ecological reasons and involve much less questions about survival in the wild of the species.*

While the criteria for listing redwood as endangered were based on number of individuals, this statement points to the more relevant conservation issue of redwood, that of maintenance of old growth stands, which have dramatically diminished in extent since the arrival of Euro-American settlers and the establishment of commercial logging regimes (Sawyer et al. 2000a). Old growth redwood forests consist of trees of many ages, including large and old trees, and structural complexity and variability (van Mantgem and Stuart 2012). However, almost all remaining old-growth redwood forests are protected—either in the region's system of state and national parks, or through agreements with the U.S. Fish and Wildlife Service designating protected areas for the federally-listed marbled murrelet (*Brachyramphus marmoratus*).

IUCN Red List statement: *Logging the species should be under stricter regulation allowing regeneration to maturity of this species.*

As a proposed conservation action, the IUCN Red List notes that the species should reach maturity in order to allow for regeneration. Maturity is defined by the IUCN as 'capable of reproduction.' Rotation ages in the redwood region vary, but even at the very lowest range of harvest ages, around 50 years old, the species is capable of reproduction. Maturity, however, is a very young age in the full course of redwood development, and at this age redwoods may not serve the many habitat functions of older redwood trees.

Under the California Forest Practice Rules, harvesting is restricted in terms of extent, timing, and cumulative impacts, including cumulative impacts to late seral habitat. Each harvest on private and state lands within California is documented in a Timber Harvesting Plan, created by a professional forester, and available to the public for review and comment. Multiple agencies (e.g., the California Dept. of Fish and Wildlife, Regional Water Boards) review each plan, including an on-the-ground

[9] Estimated for June-December 2014, http://www.boe.ca.gov/proptaxes/pdf/20142H_Final.pdf. Douglas-fir values were limited to the redwood region for geographic comparability.

assessment of impacts to public trust resources. Regulations for Timber Harvesting Plans are updated annually by the California Board of Forestry, a multi-stakeholder board approved by the Governor of California. As an example, recent regulations have clarified the responsibilities of the state's forestry sector to sequester additional carbon as part of the state's innovative and progressive global warming mitigation efforts.

Changing Area of Occupancy As a Result of Urbanization

IUCN Red List statement: *A second cause of decline of area of occupancy for redwoods is urbanization, at present a relatively minor factor, but predicted to increase much in the next few decades.*

In the listing, subcriteria c notes that the species has had a decline in 'Area of Occupancy,' defined as the extent of the occurrence of the species, itself defined as an area around which an imaginary boundary can be drawn to encompass all the known or projected sites of occurrence for the species. Sawyer et al. (2000b) state that redwoods are still found throughout their historical range. There is little evidence to suggest that urbanization or other anthropogenic land use changes threaten the area of occupancy for redwoods, as the IUCN has defined its terms.

Historically, conversion occurred in the redwood region in the 19th century for agricultural purposes, though many of these lands have reverted to redwoods because of the "vigorous sprouting ability" of redwood (Sawyer et al. 2000a, p. 32). Conversion also occurred in the middle 20th century, with timberland conversion permits totaling almost 7,689 ha (19,000 ac) throughout the region in 1970 and slowing in subsequent years (Shih 2002).

Today, the northern part of the redwood region remains remote and sparsely populated, and is mostly identified as low priority for risk from population growth and development by the California Fire and Resource Assessment Program (CDF 2010). By contrast, the southern part of the redwood region has experienced significant human development, particularly exurbanization—large-scale lots scattered throughout the forest. Exurbanization likely does not reduce the number of individual trees, but creates fragmentation of habitat.

Discussion

The concerns raised here about the veracity of the redwood listing on the Red List as endangered are not entirely new, as the "mismatch of the risk of extinction predicted from applying the IUCN criteria and that predicted from a common sense evaluation of status" has been voiced before (Godfrey and Godley 2008, p. 155). The listing itself concedes that redwood does not meet the IUCN Red List definition for endangered, as redwood conservation issues revolve around the need to maintain habitat, rather than ensure the continuation of the species in the wild.

Now that the species has been listed as endangered, it is worthwhile to examine the lessons of the listing, and its possible consequences. While the listing has had limited impacts legally and economically thus far, its intended impact (to prioritize conservation efforts) has been diluted because it ignores the conservation efforts and challenges being faced by managers and researchers in the redwood region.

Legal and Economic Consequences of Listing Redwood

There are virtually no legal or political consequences of an IUCN listing in the United States. There is no direct regulatory impact because the IUCN Red List is a guide for conservation and policy setting; it is not administered by a government or used directly as a policy tool. Some species listed on the IUCN Red List have been consequently listed on the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). CITES lists 5,600 species of animals and 30,000 species of plants in order to protect them from overexploitation through trade restrictions. However, conifers are exceedingly rare on the CITES list, and there are no conifers from the United

States listed on CITES.[10] Redwoods have never been proposed as candidate species for the United States Endangered Species Act (ESA) list, and would not likely qualify because the ESA targets species that are facing extinction.

Economic repercussions, however, are possible because of the reputation and status of the IUCN Red List and its utility as shorthand for assessing species' vulnerability. As an example, on the Wikipedia page for *Sequoia sempervirens*, the conservation status of the species is prominently listed as Endangered based on the IUCN listing. Should redwood become more widely perceived as endangered, it could undermine the image of responsible stewardship that has been cultivated by redwood producers. Of the five industrial companies operating in the redwood region, four are certified by the Forest Stewardship Council, widely considered the most rigorous standard, and the remaining industrial owner is certified by the Sustainable Forestry Initiative.[11] Though the timber companies of the redwood region operate in a restrictive regulatory environment, they have modified their management practices even further to maintain social license and to build reputations for sustainability in the marketplace.

Threats Facing the Redwood Forest

Focusing on whether redwoods as a species will exist in the future serves to distract from conservation issues facing redwood forest ecosystems, as elaborated by the Save the Redwoods League, which points out that it is the old-growth redwood habitat, and many of the species contained therein, which is endangered.[12] Most of the watercourses of the region have been listed as impaired under the Clean Water Act (303(d)) for sedimentation as a result of logging, and multiple species of the redwood region – particularly those associated with old-growth forests – have been listed as threatened or endangered under the ESA. Fire was historically an important disturbance agent within the redwood region, and its suppression has resulted in structural changes within the forest such as increases to litter and brush biomass (Brown and Baxter 2003).

Another concern regarding redwood forests is fragmentation and particularly parcelization. While the current rates of urbanization do not pose a threat to the *existence* of the species, the redwood ecosystem has been and will continue to be threatened by parcelization because of sales from commercial forest landowners in the face of greater economic opportunities—or "higher and better uses." These economic opportunities may be from real estate developers, or from agricultural uses (including marijuana, which is a common crop in the redwood region). Rural dispersed settlement creates threats through the introduction of non-native species, and fragmentation as a result of road building and lawns (Gobster and Rickenbach 2004). While fragmentation is not likely to reduce the *number* of redwoods, human settlement patterns may further impact wildlife and ecosystem processes within the redwood region.

The most commonly-used tool for maintaining species' existence in the face of decline is to impose limitations on trade. However, lowering the economic viability of redwood timber markets does not address its conservation needs, and may in fact exacerbate them. The working forests of the redwood region have maintained relatively contiguous stands over large areas, with both industrial and non-industrial owners providing the ecosystem services that accompany large, intact forests. However, their ownerships are dependent upon viable markets, which have diminished as a result of regulation, product substitution, changes to redwood products as harvests move from old growth to second-growth wood, and changing consumer preferences.

Finally, there are potential threats to redwood as a result of climate change and shifting fog patterns, as redwoods are dependent on fog as a source of moisture (Johnstone and Dawson 2010), and so could decline or decrease at their margins, in drier areas. This threat was not noted in the IUCN Red Listing.

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[10] See full list of CITES species: www.cites.org/eng/app/appendices.php.
[11] FSC: Green Diamond Resource Company, Humboldt Redwood Company, Mendocino Redwood Company, Big Creek Lumber; SFI: Sierra Pacific Industries.
[12] https://www.savetheredwoods.org/blog/wonders/celebrating-the-endangered-species-act/.

Implications for the IUCN Red List

The IUCN maintains its Red List in the hopes that it will contribute to the conservation of species in an era of unprecedented extinction levels. The IUCN Red List has great value as a global categorization system to help in the conservation of species that are threatened with extinction. However, the endangered status of *S. sempervirens* on the Red List raises concerns about the utility and objectivity of the list. Though the Red List does not mandate conservation actions, which can only be done through policy processes that weigh economic and other benefits of species, it does highlight species that are in need of conservation action. Listing species as endangered that do not meet the criteria of the listing organization raises red flags that distract from legitimate conservation concerns. In the case of redwood, the listing focuses on the continued existence of a species that does not appear imperiled, and yet does not address the ecosystem processes that are in need of conservation attention.

The redwood listing highlights a lack of local expert input. Redwood is a species that has garnered substantial conservation attention for over 100 years. It has a non-profit conservation organization dedicated entirely to its continued success (Save the Redwoods League). Yet local experts do not appear to have been consulted. One solution is to incorporate the expertise of scientists intimately familiar with species, rather than relying on broad specialist groups that tackle groups of species worldwide. The Conifer Specialist Group assessed over 600 conifer species worldwide, and its findings (at least regarding the listing of redwood) came as a surprise to many local experts.

For the purposes of informing policy, the Red List could split into multiple lists that better incorporate different ecological concerns. Mrosovsky (2003) suggests three lists: one to assess risk of extinction, a second to indicate whether the species is fulfilling its ecological role, and a third regarding economic utility (or loss of utility). Such a system may be more useful for some species that are not at risk of extinction but that may have diminished in terms of providing ecosystem services. An ecosystem risk system could use the same logic of the Red List—to make global assessments of risk possible, and to prioritize conservation efforts across regions—and could cover more ground than single-species listings. Maintaining a list of threatened ecosystems could inform a wide variety of policies that extend beyond single-species listings, such as land use policies that slow or prevent conversion. Ecosystem-level approaches provide a policy opportunity to address connectivity and function across jurisdictional, political, and ownership boundaries.

Finally, the listing of redwood highlights an important oversight of the IUCN Red List: the utility of working landscapes that connect protected areas in order to maintain ecosystem processes over large acreages. The findings of the Red List authors imply that old-growth reserves are central to redwood conservation, yet ignore the vast majority of working landscapes that could implement (and are implementing) management to create structural diversity and habitat. The Save the Redwoods League has developed the Redwoods and Climate Change Initiative, bringing together private landowners, non-profits, and governmental agencies to work on research and outreach to maintain redwood ecosystem function.[13] As part of this vision, working forests serve to connect the patches and isolated groves of protected old-growth forests and may be critical to long term sustainability of redwood ecosystem functions. Incentives to grow forests for long periods, with investments in creating more old-growth habitat and maintaining or restoring habitat connectivity, could benefit redwood ecosystems as a whole.

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Economic Contribution of Timber Harvesting and Manufacturing to North Coast Redwood Region Counties[1]

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Abstract

The ability to secure local government and public support for redwood timber sustainable redwood forest management can be enhanced by articulating local economic benefits resulting from harvesting and product manufacturing which ripples through the entire economy. Input-output modeling is used to estimate the economic contribution by summing the multiplier effect (i.e., ripple effect) of timber harvesting and manufacturing as measured by employment, wages and salaries, and value added in the local region. The activity and region of interest is redwood region timber harvesting and product manufacturing in California's North Coast counties (Del Norte, Humboldt, Mendocino, and Sonoma).

Input-output modeling and contribution analysis is conducted using IMPLAN software and 2013 data. The analysis examines how the direct effect (e.g., expenditure and employment related to harvesting, biomass, sawmill, and veneer operations) generates indirect effects (i.e., purchases by supporting sectors of the economy) and induced effects (i.e., household spending by direct effect and indirect effect employees) that result in a total effect or cumulative benefit to the local economy. Thus, the true economic benefit is greater than the value of timber harvesting and manufacturing alone as the multiplier effect of these activities impact many other sectors of the economy. This information can be used to communicate the economic benefit to North Coast counties that results from continued support of redwood region timber harvesting and product manufacturing.

Keywords: economic benefits, IMPLAN, multiplier effect, public policy, timber harvesting

Introduction

Timber harvesting and forest products manufacturing are part of the economic base for California's North Coast counties of Del Norte, Humboldt, Mendocino, and Sonoma. The predominant species harvested are coast redwood (*Sequoia sempervirens* (D.Don) Endl.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). Those involved in the forest products industry are intuitively aware that timber harvesting and associated manufacturing are "important" to the local economy; however, advocates of forestry need precise estimates of that economic importance when attempting to communicate the importance of forestry with policy makers, government officials, and the public. Input-output analysis can be used to quantify the economic importance of any sector of the economy by tracking the multiplier effect, which indicates how spending in one sector of the economy stimulates spending in other sectors. This study uses input-output analysis to provide an estimate of the economic importance or contribution that redwood timber harvesting and manufacturing has to the regional economy by quantifying the multiplier effect of this sector in terms of sales (total output), value added, employment, and wages and salaries. The economic contribution analysis was done using IMpact Analysis for PLANning (IMPLAN) economic impact assessment software originally developed by the U.S. Department of Agriculture, Forest Service and now maintained by IMPLAN LLC., formerly Minnesota IMPLAN Group (MIG) (MIG 2004). The forestry-related sectors (e.g., logging, wood energy, solid wood manufacturing, wood furniture manufacturing, etc.), as described

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in the IMPLAN study area data, represent the direct effects (e.g., jobs and value of production in the forestry and forest products manufacturing sector) which is the starting point of the analysis.

The activity of the forestry sector (i.e., purchasing inputs such as utilities, parts, etc. and labor) generates a multiplier effect creating indirect effects (e.g., supporting sectors purchase more inputs to meet demand which supports jobs in other sectors of the economy as a result of forestry-related sector activity) and induced effects (e.g., purchases in other sectors benefiting from household spending by direct and indirect employees). The induced effect also includes inter-institutional transactions, such as transfers from businesses to households (e.g., interest and dividend payments), transfers from people to government (e.g., payment of taxes), and transfers from governments to people (e.g., social security) (MIG 2004). Impacts are generated by calculated multipliers and economic impacts are estimated in terms of direct, indirect, and induced impacts. These sum to a total effect which provides an estimate of the full economic contribution or importance of the redwood region forestry and forest products manufacturing sector to the regional economy. The reported metrics of the economic contribution analysis include sales (total output), value added, employment, and wages and salaries. Total industry output refers to total value of production. Value added, which is akin to Gross Domestic Product, is the sum of payments made by industry to workers, interest, profits and indirect business taxes. Employment refers to the total number of full-and part-time jobs which is calculated based on average output per employee; therefore, this is total employment needed to support any industry and is a combination of both full and part-time jobs. Wages and salaries represents all forms of employment income and is the sum of employee compensation and proprietor income indicated in the I-O model (MIG 2004).

Forestry and associated manufacturing sectors are important to regional economies that have abundant forest resources and forestry is one of the major contributors to employment in rural America (Alvarez 2007). Globally the forest products industry contributed to over 1.1 percent of gross domestic product (GDP) and 1.2 percent in total employment opportunities to the global economy in 2014 (FAO 2014). In 2012 there were approximately 52,200 workers, earning $3.3 billion annually, employed in the California forestry and forest products industry which included primary and secondary wood and paper products, private sector forestry and logging, and forestry sup-port activities (McIver et al. 2015). Given the importance of the forest products industry, numerous studies have been conducted in the United States at the regional and/or state level, with some regularity (Abt et al. 2002, Aruna et al. 1997, Brandeis et al. 2012, Dahal et al. 2012, Dahal et al. 2015, Deckard and Skurla 2011, Hodges et al. 2005, Joshi et al. 2014, McConnell et al. 2016, Laaksonen-Craig et al. 2003, Li and Carraway 2009, Tilley and Munn 2007, Young et al. 2007). Similar but few studies have also examined the economic importance of forestry at the county level (Henderson and Munn 2013, Jackson 2015, Jeuck et al. 2014). Economic impact assessments of redwood region timber harvesting to California's North Coast are occasionally conducted particularly in consideration of policy actions with major timber harvesting reductions (Agee 1980, Berck et al. 2000, Burton and Alpert 1981, Clatterbuck 2007, Fowler 1974, McKillop 1977, Stewart 2007, Vaux 1973) to better understand potential impacts to regional employment and income as well as impacts on government revenue (McKillop 1978).

Methods

Economic contribution of the redwood region forest products industry was estimated using IMPLAN, a non-survey-based computer software and modeling system that constructs regional economic accounts and regional input-output tables. Economic contribution analysis utilizes input-output (I-O) models to track the multiplier effect of an existing sector or group of sectors to a defined economy (i.e., national, state, or county level). IMPLAN can be used to construct I-O models that are used to depict economic benefits of contributions by specific industries or activities to a specified economy. I-O multipliers describe the response of an economy to some stimulus (i.e., a change in demand or production). The stimulus for this analysis is the total output and employment of redwood timber harvesting and associated forest products manufacturing. IMPLAN models the multiplier effect or interconnections between industries, households and the government and tracks the flow of money from sector to sector. The Minnesota IMPLAN Group began developing IMPLAN databases since 1987 (MIG 2004) and provides yearly IMPLAN data and software. IMPLAN is now used in various fields to estimate impacts of economic activities for specified economies (e.g., county, state, nation). The forestry-related sectors in the IMPLAN study area data represent the direct effects (e.g., forestry jobs and value of production), which purchase goods and services from other sectors of the economy. These other producers, in turn, purchase goods and services to meet the demand of the direct effect. These indirect purchases or indirect effects continue until leakages from the economy (imports, wages, profits, etc.) end the cycle. The indirect effects and the effects of increased household spending (induced effects) can be mathematically calculated from model multipliers derived using the Leontief inverse (Leontief 1986). The resulting sets of multipliers describe the change of output for each industry caused by a change in final demand for any given industry. (MIG 2004).

Direct effect data on total output or sales for timber harvesting and related manufacturing were analyzed using IMPLAN V3.0 software which incorporates a 536-sector input-output transaction table based on North American Industrial Classification System (NAICS). Input-output models were constructed for each of California's North Coast counties and the combined four county region to examine direct effects and generate associated Social Accounting Matrix (SAM) multipliers. This type of multiplier accounts for household spending, social security and income tax leakage, institution savings, and commuting. It also accounts for inter-institutional transactions (e.g., transfers from businesses to households (interest and dividend payments), transfers from people to government (payment of taxes), and transfers from governments to people (social security, unemployment compensation among others) all result in an induced effect. These can be summed into a total effect, which is an estimate of the greater value or importance of the forestry and forest products industry to an economy. The four North Coast counties include Del Norte, Humboldt, Mendocino, and Sonoma. Forest-related industries were aggregated into three broad primary sectors: logging, lumber and wood products, and wood energy (table 1).

Table 1—Forestry products industry aggregated sector scheme and source IMPLAN sectors and description of each included in the four aggregated sectors indicating the corresponding NAICS sector classification (Note not all sectors exist in each county)

Sector aggregation	IMPLAN Sector	IMPLAN description	NAICS 2012
Forestry and logging	15	Forestry, forest products, and timber tract production	1131-2
	16	Commercial logging	1133
Wood biomass	47	Electric power generation - Biomass	221117
Solid wood	134	Sawmills	321113
	135	Wood preservation	321114
	136	Veneer and plywood manufacturing	321211-2
	137	Engineered wood member and truss manufacturing	321213-4
	138	Reconstituted wood product manufacturing	321219
	139	Wood windows and door manufacturing	321911
	140	Cut stock, resawing lumber, and planing	321912
	141	Other millwork, including flooring	321918
	142	Wood container and pallet manufacturing	32192
	144	Prefabricated wood building manufacturing	321992
	145	All other miscellaneous wood product manufacturing	321999
Wood furniture	368	Wood kitchen cabinet and countertop manufacturing	33711
	369	Upholstered household furniture manufacturing	337121
	370	Non-upholstered wood household furniture manufacturing	337122
	373	Wood office furniture manufacturing	337211
	374	Custom architectural woodwork and millwork	337212
	376	Showcase, partition, shelving, and locker manufacturing	337215

When more precise data is available, the IMPLAN study area values should be adjusted when possible. For this analysis the value of the logging (IMPLAN sector 16) was adjusted to reflect timber harvest values by county as reported by the California State Board of Equalization (2015).

Employment, wages and salaries, total output, value added and associated SAM multipliers were derived for each of forestry-related sectors. The 2013 data were used to examine the economic contribution of the North Coast's redwood forest products industry; economic contributions were measured in nominal values and reported in nominal dollars. Economic contribution analysis, not to be confused with economic impact analysis, examines the gross change in a region's existing economy that can be attributed to a given industry (e.g., redwood forest products industry) while economic impact analysis is used to examine net changes to the economic base of a region that can be attributed new revenues that otherwise would not occur (e.g., a new sawmill) (Watson et al. 2007).

This study examines the economic importance of the existing industry and is thus an economic contribution analysis. The direct effect will include the value of production and jobs for all redwood timber harvesting and forest products manufacturing sectors. The value of these sectors will be used to shock the input-output model to generate a multiplier effect to estimate the resulting indirect and induced effects. These will be summed to provide an estimate of the total effect of the industry to the regional economy. The economic contribution analysis was conducted for each of the four North Coast counties and for one regional model comprising the four counties.

Results

The study area data includes all sectors of the four county North Coast region economy and are presented in table 2. The forest products sector directly accounts for over 3,665 full- and part-time jobs with wages and salaries amounting to over $224 million and nearly $790 million in sales with value added in excess of $309 million.

Table 2—Study area data for California's North Coast four county region (Del Norte, Humboldt, Mendocino, and Sonoma) indicating values for sectors of the economy including forestry-related sectors and all other sectors aggregated at the 2 digit NAICS level

	Employment	Labor income ($M)	Output ($M)	Value added ($M)
Forestry and logging	1,316.3	85,282	149,746	92,908
Wood biomass	60.3	9,129	129,477	78,164
Solid wood	1,949.0	109,748	461,101	117,644
Wood furniture	339.6	20,257	49,224	20,707
Subtotal (forest products industry)	3,665.2	224,416	789,548	309,424
11 Ag, forestry, fish & hunting	17,217.7	1,179,511	1,874,007	1,231,859
21 Mining	856.5	81,775	206,868	114,797
22 Utilities	890.7	268,939	397,264	311,743
23 Construction	22,725.1	1,378,507	3,984,745	1,413,004
31-33 Manufacturing	24,593.3	2,714,859	9,580,141	3,101,738
42 Wholesale trade	11,873.3	1,250,485	2,692,315	1,755,887
44-45 Retail trade	43,346.3	1,887,695	3,652,533	2,420,901
48-49 Transportation & warehousing	9,878.5	572,521	1,268,252	599,105
51 Information	5,264.6	1,105,971	2,042,149	1,126,601
52 Finance & insurance	15,672.2	750,360	2,243,112	806,528
53 Real estate & rental	23,571.6	5,266,741	7,712,428	5,812,744
54 Professional- scientific & tech services	29,648.2	1,942,526	3,485,709	1,983,314
55 Management of companies	2,390.7	295,431	516,830	302,245
56 Administrative & waste services	20,299.7	838,260	1,298,809	862,259
61 Educational services	5,526.1	111,515	213,482	119,650
62 Health & social services	51,041.7	2,788,023	4,545,035	2,842,508
71 Arts- entertainment & recreation	11,884.3	353,171	734,706	368,650
72 Accommodation & food services	33,509.9	1,024,245	2,056,624	1,156,588
81 Other services	23,967.8	944,357	1,676,117	1,074,642
92 Government & non NAICS	50,687.4	4,315,910	5,304,670	4,246,492
Subtotal (rest of region economy)	404,845.7	29,070,803	55,485,797	31,651,253
Total economy	408,510.9	29,295,219	56,275,345	31,960,677

Monetary values in 2013 dollars and expressed in thousands ($M).

The study area data values for the forest products industry (table 2) indicated in the table row titled Subtotal Forest Products Industry represent the direct effect for the contribution analysis. This direct effect generates a multiplier effect resulting in indirect and induced effects on all other sectors of the economy (table 3). The economic contribution (i.e., total effect or the sum of the direct, indirect, and induced effects) of the forest products industry to the four county North Coast economy amounts to a $1.57 billion industry generating $787.9 million in value added and 10,073 full- and part-time jobs with wages and salaries of $517.3 million. Sectors of the economy that benefit most include construction, wholesale and retail trade, real estate, professional services, health services, and government, with each realizing an additional $50 to $110 million in output because of the forest products industry (table 3).

Table 3—North Coast (Del Norte, Humboldt, Mendocino, and Sonoma) forest products industry's economic contribution indicating direct effect values for forestry-related sectors and resulting indirect and induced effects on all other sectors aggregated at the 2 digit NAICS level (The total effect is indicated along with the total county economy size, the total effect expressed as a percentage of the total county economy, and the multiplier value of the forest products industry)

	Employment	Labor income ($M)	Output ($M)	Value added ($M)
Forestry and logging	1,316.3	85,282	149,746	92,908
Wood biomass	60.3	9,129	129,477	78,164
Solid wood	1,949.0	109,748	461,101	117,644
Wood furniture	339.6	20,257	49,224	20,707
Subtotal (direct effect)	3,665.2	224,416	789,548	309,424
11 Ag, forestry, fish & hunting	147.5	6,048	11,296	7,833
21 Mining	7.6	300	1,822	1,017
22 Utilities	5.6	723	2,981	2,009
23 Construction	341.2	20,434	59,771	21,176
31-33 Manufacturing	34.7	2,250	14,410	3,641
42 Wholesale trade	242.6	16,390	55,013	35,878
44-45 Retail trade	695.3	23,756	58,914	39,019
48-49 Transportation & warehousing	246.1	13,528	32,385	14,750
51 Information	96.5	6,080	37,048	19,651
52 Finance & insurance	284.4	12,297	40,778	15,029
53 Real estate & rental	313.7	5,383	122,710	91,926
54 Professional- scientific & tech services	501.4	23,622	58,946	35,014
55 Management of companies	55.9	5,872	12,090	7,071
56 Administrative & waste services	367.4	11,180	23,482	15,489
61 Educational services	107.4	2,482	4,148	2,328
62 Health & social services	791.8	41,434	69,702	43,559
71 Arts- entertainment & recreation	175.1	3,055	10,483	5,198
72 Accommodation & food services	685.7	16,659	41,722	23,511
81 Other services	427.3	17,620	30,504	19,519
92 Government & non NAICS	880.5	63,806	87,504	74,834
Subtotal (indirect & induced effects)	6,407.7	292,919	775,709	478,453
Total (total effect)	**10,072.9**	**517,335**	**1,565,257**	**787,877**
Total county economy	408,510.9	29,295,219	56,275,345	31,960,677
Total as % of four county region	2.5%	1.8%	2.8%	2.5%
Multiplier	2.75	2.31	1.98	2.55

Monetary values in 2013 dollars and expressed in thousands ($M).

The economic contribution analysis for each of the four individual counties are reported in tables 4 to 7. Del Norte's forest products industry economic contribution amounted to over $9.08 million in sales or output, which was the smallest of the four counties, while Humboldt's economic contribution of forestry-related sectors amounts to nearly $657.4 million in sales and has the highest contribution value of the North Coast counties. The Sonoma and Mendocino forest product industry generates an economic contribution amounting to $310.2 million and $329.7 million in sales, respectively.

Table 4—Del Norte forest products industry's economic contribution indicating direct effect values for forestry-related sectors and resulting indirect and induced effects on all other sectors (the total effect is indicated along with the total county economy size, the total effect expressed as a percentage of the total county economy, and the multiplier value of the forest products industry)

	Employment	Labor income ($M)	Output ($M)	Value added ($M)
Forestry and logging	25.0	1,252	2,452	1,397
Wood biomass	0.0	0	0	0
Solid wood	10.7	697	3,598	908
Wood furniture	0.0	0	0	0
Subtotal of direct effects	35.7	1,949	6,050	2,305
Indirect & induced effects	24.6	1,079	3,034	1,865
Total (total effect)	60.3	3,028	9,084	4,170
Total county economy	11,232.8	728,040	1,270,078	783,147
Total effect as % of county	0.5%	0.4%	0.7%	0.5%
Multiplier effect	1.69	1.55	1.50	1.81

Monetary values in 2013 dollars and expressed in thousands ($M).

Table 5—Humboldt forest products industry's economic contribution indicating direct effect values for forestry-related sectors and resulting indirect and induced effects on all other sectors aggregated at the 2 digit NAICS level (the total effect is indicated along with the total county economy size, the total effect expressed as a percentage of the total county economy, and the multiplier value of the forest products industry)

	Employment	Labor income ($M)	Output ($M)	Value added ($M)
Forestry and logging	769.1	42,002	79,030	46,462
Wood biomass	60.3	9,129	129,476	78,164
Solid wood	823.8	55,059	212,265	58,000
Wood furniture	65.4	2,703	8,460	2,807
Subtotal of direct effects	1,718.5	108,893	429,232	185,432
Subtotal (indirect & induced effects)	3,320.2	136,234	228,160	376,994
Total (total effect)	5,038.7	245,127	657,393	562,426
Total county economy	68,409.1	4,275,694	8,389,036	4,722,941
Total effect as % of county	7.4%	5.7%	7.8%	11.9%
Multiplier effect	2.93	2.25	1.53	3.03

Monetary values in 2013 dollars and expressed in thousands ($M).

Table 6—Mendocino forest products industry's economic contribution indicating direct effect values for forestry-related sectors and resulting indirect and induced effects on all other sectors (the total effect is indicated along with the total county economy size, the total effect expressed as a percentage of the total county economy, and the multiplier value of the forest products industry)

	Employment	Labor income ($M)	Output ($M)	Value added ($M)
Forestry and logging	394.6	38,409	57,597	40,696
Wood biomass	0.0	0	0	0
Solid wood	566.9	27,080	136,184	30,099
Wood furniture	12.0	474	1,476	492
Subtotal of direct effects	973.4	65,962	195,257	71,287
Subtotal (indirect & induced effects)	1,203.1	44,961	134,427	80,285
Total (total effect)	2,176.5	110,923	329,684	151,572
Total county economy	49,115.6	2,975,548	5,968,931	3,329,705
Total effect as % of county	4.4%	3.7%	5.5%	4.6%
Multiplier effect	2.24	1.68	1.69	2.13

Monetary values in 2013 dollars and expressed in thousands ($M).

Table 7—Sonoma forest products industry's economic contribution indicating direct effect values for forestry-related sectors and resulting indirect and induced effects on all other sectors (the total effect is indicated along with the total county economy size, the total effect expressed as a percentage of the total county economy, and the multiplier value of the forest products industry)

	Employment	Labor income ($M)	Output ($M)	Value added ($M)
Forestry and logging	127.7	3,620	10,667	4,353
Wood biomass	0.0	0	0	0
Solid wood	547.6	26,912	109,053	28,637
Wood furniture	262.3	17,080	39,288	17,409
Subtotal of direct effects	937.5	47,612	159,008	50,399
Subtotal (indirect & induced effects)	1,209.1	57,588	151,270	93,356
Total (total effect)	2,146.6	105,200	310,277	143,756
Total county economy	279,753.4	21,315,935	40,647,298	23,124,884
Total effect as % of county	0.8%	0.5%	0.8%	0.6%
Multiplier effect	2.29	2.21	1.95	2.85

Monetary values in 2013 dollars and expressed in thousands ($M).

Discussion and Conclusion

The economic contribution of the forest products industry differs greatly by county for two reasons. The total effect or contribution value is reflective of the direct effect value (i.e., size of the forest products industry) and the multiplier value (i.e., how the economy responds to spending). For example, Del Norte's direct effect from forestry-related sectors was about $3.6 million in sales or output with an associated output multiplier of only 1.31. Meaning that each $100 in forest products industry output generates another $31 in output form other sectors of the economy. Compare that with

Humboldt's direct effect of over $429 million and an output multiplier of 1.53. Thus, Del Norte's forest products industry has an economic contribution amounting to over $9.08 million output, while Humboldt's economic contribution of forestry-related sectors amounts to over $657 million in sales. Del Notre County, as compared with the other North Coast counties, has a smaller forest products industry and overall economy with fewer local businesses to capture expenditures by forest sector employees. As expected the multiplier effect for output, not surprisingly, is largest for Sonoma at 1.95 and lowest for Del Norte at 1.50. Sonoma realizes the largest multiplier effect and Del Norte the smallest of the 4 counties as each is the largest and smallest, respectively, of the North Coast county economies. This reflects that a larger economy is generally a more diverse economy with more opportunities to respond to changes in final demand rather than loosing input purchase spending as leakages when input purchasing occurs outside of the defined economy.

Humboldt has the largest employment multiplier at 2.93 indicating that each 100 jobs in the forest products sector contributes another 193 jobs in other sectors of the economy. The multiplier effect is larger for the four county North Coast regional economy than any individual county as a larger regional economy captures more potential leakage that would occur in a smaller county economy which cannot respond to all of the input purchases required by the forest products industry and its supporting sectors. As expected, Sonoma realizes some of the largest multiplier effects and Del Norte the smallest. Humboldt's multiplier values are also among the largest which is a reflection of the county having the largest and most diversified forest products industry of the four North Coast counties but also an economy that is able to capture more potential leakage. This demonstrates that the Humboldt economy has a comparably greater share of sectors that are supportive of the forest products industry than other North Coast counties, which is to be expected given the size of Humboldt's forest products industry. The magnitude of the economic contribution differs as size of the forest products industry and the overall economy varies greatly across the four North Coast counties. However, the forest products industry makes an important economic contribution to each of the North Coast counties.

Forestry provides numerous economic benefits to local economies (e.g., sales, jobs, income), and the ability to practice forestry depends upon access to publicly maintained infrastructure (i.e., road and bridges) to transport harvested and manufactured forest products. However, the economic benefit that results because of forestry-related harvesting and manufacturing is not always understood or fully appreciated. Having access to periodic assessments of the economic contribution of forestry and forest products manufacturing can empower advocates to better communicate the economic importance of forestry to policy makers, elected officials, and the public. The economic contribution analysis of forestry and forest products manufacturing and associated economic multiplier effects expressed in terms of employment, wages and salaries, value added, and total output to California's North Coast counties of Del Norte, Humboldt, Mendocino, and Sonoma presented here can serve as a powerful aid in effectively communicating the economic importance of the practice of forestry.

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Cannabis (*Cannabis sativa* or *C. indica*) Agriculture and the Environment: a Systematic, Spatially-Explicit Survey and Potential Impacts[1]

Van Butsic[2] and Jacob C. Brenner[3]

Abstract

Cannabis (*Cannabis sativa* or *C. indica*) agriculture is a multi-billion dollar industry in the United States that is changing rapidly with policy liberalization. Anecdotal observations fuel speculation about associated environmental impacts, and there is an urgent need for systematic empirical research. An example from Humboldt County, California, a principal cannabis-producing region, involved digitizing 4,428 grow sites in 60 watersheds with Google Earth imagery. Grows were clustered, suggesting disproportionate impacts in ecologically important locales. Sixty-eight percent of grows were > 500 m from developed roads, suggesting risk of landscape fragmentation. Twenty-two percent were on steep slopes, suggesting risk of erosion, sedimentation, and landslides. Five percent were < 100 m from threatened fish habitat, and the estimated 297,954 plants would consume an estimated 700,000 m^3 of water, suggesting risk of stream impacts. The extent and magnitude of cannabis agriculture documented in our study demands that it be regulated and researched on par with conventional agriculture.

Keywords: California, drug production, land use change, marijuana, research agenda, satellite imagery

Introduction

Illegal drug production and distribution are multi-billion-dollar global industries (UNODC 2014) with potential to transform ecosystems (Benessaiah and Sayles 2014, Mcsweeney et al. 2014). Drug supply chains are generally thought to involve production in the Global South to satisfy demand in the Global North, but this assumption no longer holds true for cannabis (*Cannabis sativa* or *C. indica*) (Decorte et al. 2011). The geography of cannabis agriculture is shifting, with import substitution now observed in almost every developed country in the world (Potter et al. 2011).

In the United States, cannabis agriculture has been understudied and underestimated in scope and magnitude (Weisheit 2011). Research on cannabis agriculture systems is especially urgent in light of recent policy liberalization (Crick et al. 2013), which is facilitating a transition in cannabis from an illegal drug to a licit agricultural crop. Cannabis is still federally illegal in the United States as a Schedule 1 drug according to the Drug Enforcement Agency (http://www.dea.gov/druginfo/ds.shtml), and this classification has stymied research on cannabis production methods and their environmental impacts (Eisenstein 2015). However, over the last 2 decades the majority of states have liberalized cannabis policy (Cole 2013), ranging from decriminalization to medical, permitting to the creation of retail markets for recreational use. The latest federal spending bill prohibits federal agents from interfering with the enactment of state laws allowing medical cannabis use. States are likewise left to address any collateral impacts of the burgeoning medical cannabis industry. State-level regulations have at times included explicit environmental protections, such as laws approved in late 2015 in California meant to hold cannabis agriculture to the same standards as other crops (State of California 2015). In general, policymakers are challenged to keep up with the rapid changes in cannabis agriculture on the ground.

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Legal United States markets for cannabis were estimated to be worth $2.7 billion in 2014 and projected to reach $11 billion by 2019 (Arcview Market Research 2014). This expanding market, coupled with new opportunities to grow cannabis free from threat of federal enforcement, suggest significant near-term shifts in production. Even with new regulatory protections for the environment and their embrace by many growers (McGreevy 2015), a boom in cannabis agriculture promises serious environmental implications (Carah et al. 2015).

Building on other scholars' (Carah et al. 2015, Eisenstein 2015, Sides 2015) recognition of cannabis production as a topic of growing environmental concern and their calls for more rigorous research, we present here a study on the expansion and intensification of land use for cannabis agriculture. Our study, as an example of what could be done anywhere cannabis agriculture takes place, illustrates the value of a systematic environmental research approach.

In the current era of policy liberalization, the seat of cannabis agriculture in the United States is a region known as the "Emerald Triangle" in northern California (Corva 2014). Consisting of Humboldt, Trinity, and Mendocino counties, the Emerald Triangle is arguably the birth place of modern cannabis production in the United States, and Humboldt County might be the top cannabis-producing region in the world (Corva 2014).The Emerald Triangle is also home to outstanding natural resources including large stands of old-growth California redwood (*Sequoia sempervirens* (D. Don) Endl.) and relatively uninterrupted runs of endangered and threatened anadromous fish, such as steelhead trout (*Oncorhynchus mykiss*) and Chinook salmon (*Oncorhynchus tshawytscha*). The potential conflict between the rapidly growing cannabis industry and the habitat needed by these protected species is thus a federal-level, as well as a local-level, environmental concern.

Popular media speculation about environmental impacts of cannabis agriculture in this region, especially impacts on water, is widespread (Bland 2014, Harkinson 2014, Ryzik 2014), but empirical research is limited (Carah et al. 2015). The small body of scientific research points to profound negative consequences, including decreased stream flows (Bauer et al. 2015), rodenticide poisoning of rare carnivores (Gabriel et al. 2012), and high carbon emissions from greenhouses (Mills 2012). While these studies show negative impacts of cannabis production, they are all based on limited, non-random sampling in areas where cannabis production is known to be high. Thus, they cannot be used to infer impacts at broader scales.

In order to identify the extent of land-use change for cannabis production and other potential impacts on the environment, we systematically mapped grow sites in a random sample of 60 watersheds in and boarding Humboldt County that statistically represent the county a whole. See supporting online information for sampling details. We used our map results to answer four questions about cannabis agriculture and its potential impacts on the environment:

1) How many cannabis grows are in the study area, and what are the attributes of these grows?
2) Are there statistically significant spatial patterns of cannabis production within and across watersheds?
3) Do grows threaten natural areas by being located on sensitive sites far from developed infrastructure?
4) Do grows pose a risk to threatened species due to their water consumption and location near critical habitat?

Methods

Study Area

Our study area consisted of 60 randomly sampled (out of 112 total), ecologically representative watersheds within and bordering Humboldt County (12 digit WBD) (USDA NRCS 2015) (fig. 1). The area is characterized physically by steep terrain (34 percent of land with slope > 30 degrees), large areas of forest, and > 160 km of Pacific Ocean coastline. Coastal areas are consistently cool with summer high temperatures seldom exceeding 26 °C. By contrast, inland valleys and uplands are warmer in summer and cooler in winter (California 2015).

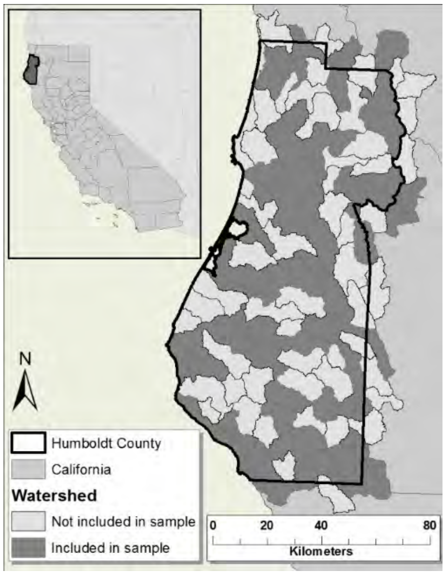

Figure 1—Sampled watersheds within and adjacent to Humboldt County, California.

Excluding cannabis, agricultural sales in Humboldt County totaled nearly $270 million in 2013. Livestock production contributed $76 million, followed by timber ($72 million), milk and dairy products ($61 million), nursery stock ($49 million), field crops ($5 million), and fruit, nut and vegetable crops ($3 million) (Humboldt County 2015). Over 50,000 ha of land are in organic production. Humboldt County participates in the Williamson Act, which reduces property taxes for owners who commit their land to agricultural uses. In forested areas with high timber value, Timber Production Zone designations reduce property tax in exchange for limiting land development potential. Humboldt County producers have access to state, regional and international markets for their products.

Methods of cannabis production are not well known to researchers due to the traditionally illicit nature of the product. Since the prohibition of cannabis in the 1930s, research into horticultural and agronomic methods has been prohibited in the United States. Thus, there is no published literature on the modes of production used in our study area. Popular accounts point to three main cannabis production modes in our area: indoor cultivation with artificial light; greenhouse cultivation where light may be natural, artificial, or both; and outdoor cultivation with natural light. Growers report the importation of enhanced soils to make up for poor-quality natural soils throughout the county. There is no research documentation of fertilizer or pesticide use in cannabis production in our area, though both are reported to be used elsewhere (Carah et al. 2015).

Data

We located and mapped greenhouse and outdoor grow sites with high-spatial-resolution satellite imagery in Google Earth. The fine spatial grain of this imagery allowed us to visually detect even small, sparsely planted grows, which are not easily captured using spectral remote sensing (Daughtry and Walthall 1998, Kalacska and Bouchard 2011). These grows make up a large proportion of the cannabis agriculture operations in our study area.

Data on critical steelhead trout and Chinook salmon habitat locations were provided by the California Department of Fish and Wildlife (California Department of Fish and Wildlife 2015). In our study area, these salmonids are listed as threatened under the federal Endangered Species Act (National Oceanic and Atmospheric Administration 2015). We choose to feature these species because they are vulnerable to low flows (imposed by water withdrawals), soil erosion, and agrochemical contamination.

Data on slope and zoning were developed and provided by Humboldt County (http://www.humboldtgov.org/1357/Web-GIS). We used the Watershed Boundaries Dataset at the Hydrological Unit Code (HUC) 12 level (USGS 2015). The LANDFIRE dataset was used to determine land cover type (USDA 2013).

Identifying and Delineating Grow Sites

Outdoor grows and greenhouses can be visually detected in high-spatial-resolution satellite imagery (fig. 2) (Bauer et al. 2015). We used fall images from 2012 and 2013, because cannabis plants are mature at this time and can be distinguished from other vegetation based on their size, arrangement, and color. We demarcated grows using heads-up digitizing within a systematic grid pattern overlaid on each watershed. For outdoor grows, we counted the number of plants. To estimate plants in greenhouses, we followed Bauer et al. (2015) in assuming one plant needs 1.115 m^2 of greenhouse area. We assumed all greenhouses are used for cannabis production based on a 19-fold increase in greenhouses 2004 to 2014, and a simultaneous decrease in nursery crop production (Humboldt County 2015).

Figure 2—Image from Google Earth showing cannabis plants and greenhouse from 2012.

Spatial Distribution and Clustering of Grows

We analyzed the distribution and clustering of grow sites (outdoor and greenhouse grows combined) at two scales, within and across watersheds. Across watersheds, we calculated a global Moran's I statistic to test for spatial autocorrelation among watersheds with respect to plant density (number of plants/watershed area). We then carried out an optimized hotspot analysis to calculate Getis-Ord Gi* statistics for the study area and for each individual watershed (Getis and Ord 2010). At least 30 grows had to be present in a watershed to calculate the Getis-Ord Gi* statistic, and 26 of 60 watersheds met this standard. The ArcGIS Optimized Hotspot Analysis Tool and Global Moran's I tools were used for these analyses (ESRI 2015).

Threats Due to Remote and Steep Grow Sites

We overlaid ancillary spatial data in a GIS to derive proxies for potential threats to natural areas. First, we calculated the distance from each grow to the nearest developed road as a proxy for fragmentation caused by land clearing and road building. Next, we overlaid grows on a > 30 percent slope layer as an indicator of potential for erosion, sedimentation, and mass wasting (landslides, and others).

Potential Impacts on Threatened Freshwater Species

To better understand potential impacts on threatened species we calculated the number of plants and grows located within buffers around critical habitat of Chinook salmon and steelhead trout. We complemented our spatial analysis with total water use estimates. To quantify water use in our study, we applied published water use rates per plant (Bauer et al. 2015) to the number of plants identified in our mapping exercise. Our assumptions were thus 22.7 liters per plant per day over a 150 day growing season (Humboldt Growers Association 2010).

Results

Number and Extent of Grows

We located 4,428 grow sites in our study area containing an estimated 297,954 plants. The average grow contained 67 (SD 75) plants. Greenhouse grows (n = 2407) contained more plants on average (85.77, SD 88.81) than outdoor grows (n = 2021) (45.23, SD 45.266). The largest outdoor grow had 757 plants, while the largest greenhouse grow had an estimated 960 plants. An average watershed in our study area would contain 70 grows (SD 102) and 4,770 plants (SD 6,448). The maximum number of grows in one watershed was 481, and the maximum number of plants in one watershed was 26,677. We identified zero grows in 11 watersheds (table 1, fig. 3)

Table 1—Summary statistics for individual grows and watersheds

Type	Mean	Std. deviation	Minimum	Maximum
Outdoor grows				
No. plants	45.26	45.38	2	757
Water use (m^3)	104.56	104.83	4.62	1748.67
Greenhouse grows				
No. plants	85.77	88.81	1	960
Water use (m^3)	198.15	205.16	2.31	2217.60
All grows				
No. plants	67.28	75.06	1	960
Water use (m^3)	155.43	173.39	2.31	2217.60
Summarized at watershed scale				
No. grows	71	102	0	481
No. plants	4770	6448	0	26677
Water use (m^3)	11000	14900	0	61600

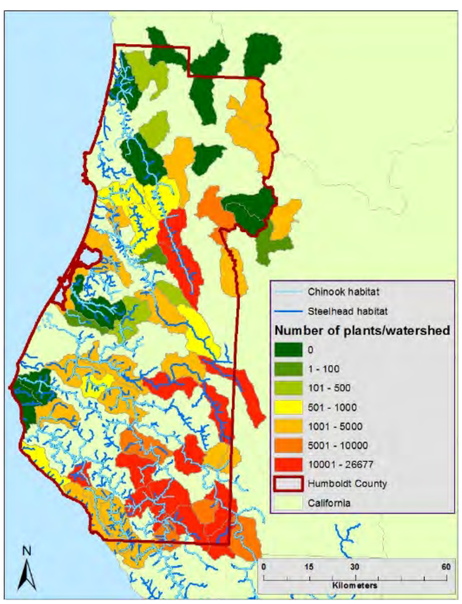

Figure 3—Number of plants per watershed and location of critical habitat for steelhead trout and Chinook salmon.

We discovered strong spatial clustering across watersheds in our study area and within watersheds. At the scale of the study area, there is statistically significant positive spatial autocorrelation among watersheds with respect to the density of plants (number of plants/watershed area). The Moran's I was 0.371 (z-score 4.194, p-value 0.000027). The optimized hot spot analysis applied to the full study area resulted in the identification of three hotspots and one cold spot (fig. 4). The optimized hot spot analysis conducted at the individual watershed scale also showed strong clustering, with hot spots present in all 26 watersheds analyzed.

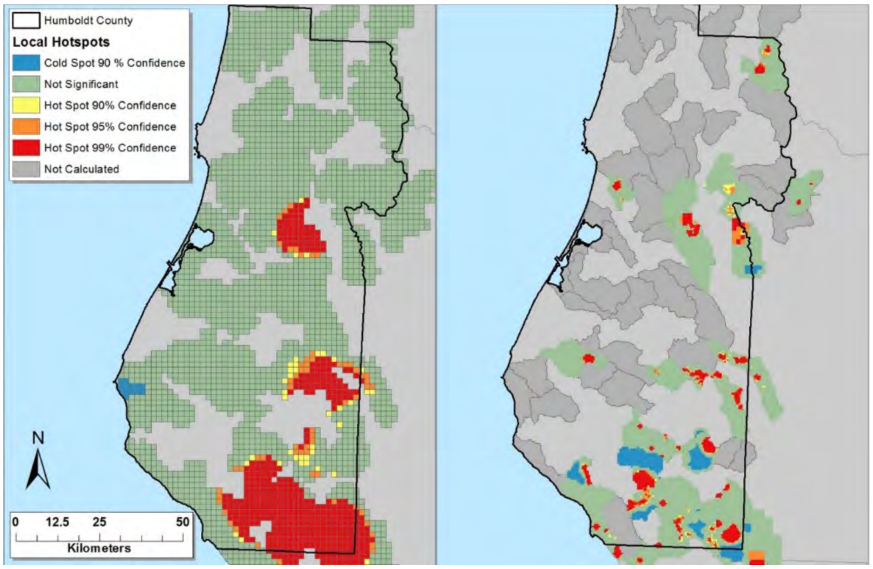

Figure 4—Hot spots of cannabis cultivation in Humboldt County. A) Results of the optimized hotspot analysis for the whole county. B) Result of the optimized hotspot analysis run individually for 26 watersheds.

Proxies for Habitat Threats

Over 68 percent of grows were located more than 500 m from a developed road (fig. 5C), while 15 percent were within 100 m. Total cultivated area covered by greenhouses and outdoor grows totaled 1.2 km^2. Twenty three percent of grows were located on slopes measuring > 30 percent. Equal percentages of outdoor and greenhouse grows were located on steep slopes.

Potential Impacts on Threatened Freshwater Species

We calculated the number of grows located within buffers of steelhead trout and Chinook salmon habitat. Twenty five percent of the grows we identified were located within 500 m and 6 percent were located within 100 m of Chinook salmon habitat (fig. 5D). Nineteen percent of grows were located within 500 m and 4 percent were located within 100 m of steelhead trout habitat (fig. 5D).

Because water use is a linear function of the number of plants, water use followed the same distribution as number of plants across space. In total we estimated 688,000 m^3 of water used annually to irrigate cannabis in our study area. The largest greenhouse consumed 2,218 m^3 of water and the largest outdoor grow consumed 1,740 m^3 of water. At the watershed scale, an average of 11,000 m^3 of water was used to irrigate cannabis grows, with a maximum of 61,600 m^3 (table 1).

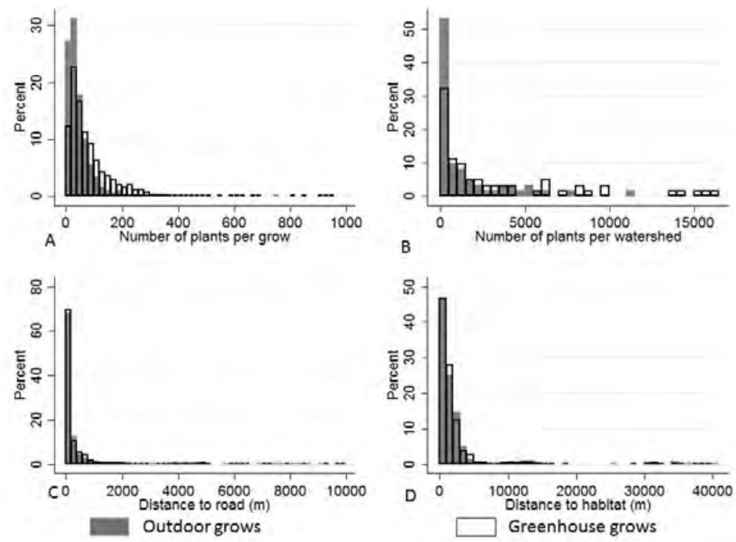

Figure 5—Distribution of A) plants per grow, summarized by outdoor and greenhouse grows. B) plants per watershed, summarized by outdoor and greenhouse grows. C) Distance from grow sites to developed roads. D) Distance from grow sites to critical habitat for Chinook salmon and steelhead trout.

Discussion

Our results, which show abundant grow sites clustered in steep locations far from developed roads, potential for significant water consumption, and close proximity to habitat for threatened species, all point toward high risk of negative ecological consequences associated with cannabis agriculture as it is currently practiced in northern California. Cannabis production was ongoing as of 2014 in 83 percent of sampled watersheds, suggesting that cannabis agriculture is already widespread. The footprint under cultivation is relatively small (122 ha compared with > 50,000 ha of organic farmland), but the associated environmental impacts may extend far beyond the grow sites themselves (Carah et al. 2015). Given the current profitability of cannabis production, we expect that cannabis agriculture will expand into other sites with suitable growing conditions throughout the region.

The spatial clustering of grows in environmentally sensitive areas within individual watersheds suggest that cannabis production will have disproportionate impacts in certain locales, such as those highlighted previously by Bauer et al. (2015). California's ability to mitigate these impacts requires an understanding of not only where cannabis production takes place, but also the conservation values of grow sites, as well the mechanisms linking cannabis agriculture with local ecosystems. Past work on water use impacts during sensitive periods of drought stress in headwater streams (Bauer et al. 2015) is a good example of the type of research that could be advanced by a systematic survey such as ours, which shows a range of impacts on different watersheds. We join these other researchers in arguing for ecological monitoring of cannabis hotspots as a top priority.

Explanation of the patterns we observed is an important task for future research. The drivers of spatial clustering in cannabis production are almost completely unknown. One might hypothesize a combination of biophysical factors, such as access to water for irrigation (Bauer et al. 2015), and social factors, such as law enforcement activities (Corva 2014). Other factors that might explain cannabis agriculture patterns include land tenure (Polson 2013), local land-use regulation (Polson 2015), and agglomeration economies (Pflüger 2004). Land-use science on cannabis agriculture lags behind research on other crops, but advances in the field will be crucial for predicting future cannabis expansion and moderating its impacts.

Historically, cannabis is often exempt from the regulations that govern other agricultural crops (Stone 2014). Conservationists and growers alike have called for regulation of cannabis production (Harkinson 2014), often due to fears of environmental impact (Carah et al. 2015). Bills recently signed into law by the Governor (Assembly Bill 243, Assembly Bill 266, and State Bill 64) represent a defining moment in California's history of cannabis production by: a) requiring municipalities to develop land use ordinances for cannabis production; b) forcing growers to obtain permits for water diversions; and c) introducing seed-to-consumer tracking.

However, bringing the industry into compliance is no small task. Many grows are located in remote areas and access can only be granted through private roads, making access for audits and other measures of regulatory enforcement difficult, if not impossible. In addition to the remote and semi-clandestine nature of many grow operations, cannabis agriculture is practiced primarily by widely dispersed, small producers. (We suspect there is a minimum of 5,000 producers in the Emerald Triangle and the number may be twice as high. For comparison, there are roughly 400 wineries in Napa County). Much of the newly proposed regulatory regime relies on self-reporting.

Currently, there is a lack of basic information on cannabis agriculture as it is currently practiced. We know of no water-balance models based on actual cannabis water use. Our water use estimates therefore should be interpreted with caution. Anecdotal evidence suggests growers can reduce water use by 70 percent by cultivating small plants that mature quickly, although there is no suggestion of the implications of this production system for yields (Walker 2015). Likewise, we know of no published research on the agrochemical intensity of cannabis agriculture, although work has shown that anti-coagulant rodenticides are used at some sites (Gabriel et al. 2012, Thompson et al. 2014). Popular media and anecdotal observations suggest a movement toward organic production methods (Troung 2015).

In our study area we documented two different production methods – outdoor grows and greenhouse grows—as well as heterogeneity within each of these cropping systems. For outdoor grows, plants are often grown in planters or raised beds, presumably using imported soils. Likewise, some greenhouse grows appear to use artificial light while others do not. These differences indicate to us that there are likely widely different impacts from different production systems. For instance, we might expect less erosion from greenhouses than outdoor grows since soils within greenhouses are not exposed. At the same time, we note that many greenhouses are surrounded by large clearings created during construction with exposed soils subject to erosion. Expanded field research into the differences in production systems is needed to better understand this heterogeneity.

Like the lack of environmental regulation of cannabis production, the lack of research on cannabis agricultural practice is strongly tied to the federally illegal status of cannabis as a Schedule I drug, a fact that prevents all but a few researchers from conducting field and laboratory studies. As licit cannabis production under the aegis of medical and recreational uses spreads through the United States it is crucial for federal oversight to allow researchers to keep pace with developments in the field. Field based measurements of water use, chemical use, cropping systems, and yields are all needed to inform effective agricultural policy.

Greater research is also needed on the social systems underlying cannabis agriculture. Very little is known about the relationship of land tenure and cannabis agriculture. Further, we know of no systematic survey of growers to identify predominant demographic and socio-economic characteristics. Such information is important for understanding social drivers of the boom in cannabis agriculture, as well as prospects for compliance with regulations.

It is important to put the impact of cannabis production in perspective with the production of other agricultural commodities. For example, our water use estimate of 668,000 m^3 is comparable to the irrigation demand of 40 ha of almonds in other parts of California (Connel et al. 2012). This is a relatively small amount considering that there are over 320,000 ha of irrigated almonds in the state (USDA 2012). Likewise, the cultivation of cannabis in our study area occupies less than 2 km^2 (23 ha under greenhouses), a miniscule proportion of the Humboldt landscape. It is thus apparent that the total stock of land or water resources consumed is not in itself troubling. Rather, it is the spatial distribution of cannabis agriculture that determines environmental harm. Locating grows in areas with better access to roads, gentler slopes, and ample water resources could significantly reduce threats to the environment. Future cannabis policy liberalization should take into consideration the potential for mitigating environmental impacts through land-use planning.

The economic impacts of cannabis agriculture should also be compared to other agricultural products. For example, the annual profit from 40 ha of almonds could be up to $422,000 (Connel et al. 2012). Using a conservative 0.45 kg/plant average (Walker 2015), and a market price to growers of $1,100/kg, our research suggests a wholesale economic value of around $150 million and an annual retail value of ~$1 billion (at $7,400/kg) for just the cannabis produced in the proportion of Humboldt County included in our study (Wang 2015). This estimate exceeds twice the total value of timber, livestock, dairy, nursery, and vegetable crops grown in Humboldt County in the same year (Humboldt County 2015). Therefore, while potential threats to the environment from cannabis agriculture are clear, there may also be opportunities for sustainable rural development (Polson 2015). Indeed, sustainable cannabis agriculture might provide a unique and significant opportunity for land sparing and nature preservation.

The goal of our study was to document the extent of cannabis agriculture and highlight potential environmental threats. Moving forward, integrated research on biophysical and social drivers of cannabis agriculture is needed to better understand why grows appear where they do, who is developing these grows, how these grows impact ecosystems and biodiversity, and what are the economic prospects for this industry in the future. We believe that the proper characterization of cannabis as an agricultural crop coupled with greater legal access for researchers to production sites could enable the growth of a research field centered on cannabis agriculture as an important human-environment system.

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Estimating the Impact of Cannabis Production on Rural Land Prices in Humboldt County, CA[1]

Benjamin Schwab [2] and Van Butsic[3]

Abstract

Amenity values, development potential, commodity prices and productive capacity largely determine rural land prices. For rural lands used in timber and agricultural production, capacity and expected future commodity prices play primary roles. For rural lands that are used as second homes or recreational properties, amenities—such as being near lakes or having scenic views—drive pricing. Here, we examine the impact of cannabis production on rural property values in Humboldt County, California, the largest cannabis producing county in the country and also home to both productive and recreational rural lands. We hypothesize that lands that are best for cannabis production will be impacted by two competing forces. On one hand, areas with high cannabis capacity should have higher prices if potential returns to growing cannabis are highest in these areas. On the other, these areas may have social disamentities that provide downward pressure on property values (e.g., higher levels of crime, transient workers, etc.). Using a hedonic model that accounts for land's productive capacity as well as the presence of potential disamenties, we find the density of cannabis production has a positive relationship with property prices. Our results suggest that a doubling of the median existing cannabis density in a watershed is associated with a 3 to 4 percent increase in the sales price of undeveloped land in Humboldt County.

Keywords: amenity value, hedonic models, illegal markets, marijuana

Introduction

Over the last half century many rural areas in the United States have undergone broad social and cultural transformations. Behind much of these changes lies a shift in land use (Radeloff et al. 2005). Many rural lands which once were used for resource use and extraction, are now primarily used for tourism and second home ownership (Brown et al. 2005). Such shifts have not only impacted local communities, but land markets as well.

The prices for productive and recreational lands are determined by different economic forces. For timber and ranching lands, land prices are based on the productive capacity of the land (either in terms of timber production or livestock) and expectations of future returns to management (Conrad 2010). For recreational properties, land prices are largely driven by amenity values such as hunting and fishing opportunities, scenic values, and wildlife viewing (Smith et al. 2002). Likewise, there is strong evidence that disamenities, such as crime or degraded environments, negatively impact these amenity based properties (Boggess et al. 2014).

Cannabis production is an increasingly important rural land use in many parts of the country. Now legal as either medicine or for recreational purposes in over 30 states, cannabis production is a multi-billion dollar industry and much of the production takes place in rural areas where traditional natural resource uses mingle with the new wave of economic activities (Arcview Market Research 2014). Economic theory suggests that cannabis production may have competing impacts on property prices. On one hand, properties that are well-suited, either socially or biophysically, to production should experience increased property prices based on the potential for high future returns. At the same time, these properties may be less attractive to buyers who see the potential cannabis production on

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neighboring parcels as a disamenity, which can lower the amenity value of a parcel or cause land owners to invest in ameliorative actions. Given these competing forces, whether cannabis production increases or decreases property prices remains an empirical question.

We examine the impact of cannabis production on property values in Humboldt County, California, one of the largest cannabis production regions in the country, and likely the world. Combining a dataset of over 3,000 arm's length property transactions with data on the location of cannabis farms, we identify the impacts of cannabis production using a hedonic pricing model. The results of our empirical investigation suggest that areas that are best suited for cannabis production have statistically significant higher prices than similar parcels in parts of the county that do not produce cannabis.

Methods

Study Area

Our study area consists of 54 randomly selected watersheds in Humboldt County that are representative of the area as a whole (see Butsic and Brenner 2016, for comparative statistics). Humboldt County is located in northern California (fig. 1) along the Pacific Coast and is considered the leading cannabis producing county in the United States, if the not world. The county is heavily forested with a mix of coniferous and hardwood forest, with pockets of open rangeland. Timber production contributes about $72 million in direct sales to sawmills, secondary manufacturing, and biomass energy plants that generate additional value added products (Humboldt County 2015). The harvesting and processing of wood has historically been a major center of economic activity. Due to the steep terrain and poor soils, traditional agriculture is limited to a relatively small area of the county. Livestock, dairy, and nursery production are the largest agricultural sectors ($76, $61, $41 million dollars in sales in 2014) and make up over 95 percent of all agricultural production by value. In comparison, the wholesale value of cannabis production is likely over $300 million, although no official figures exists (Butsic and Brenner 2016).

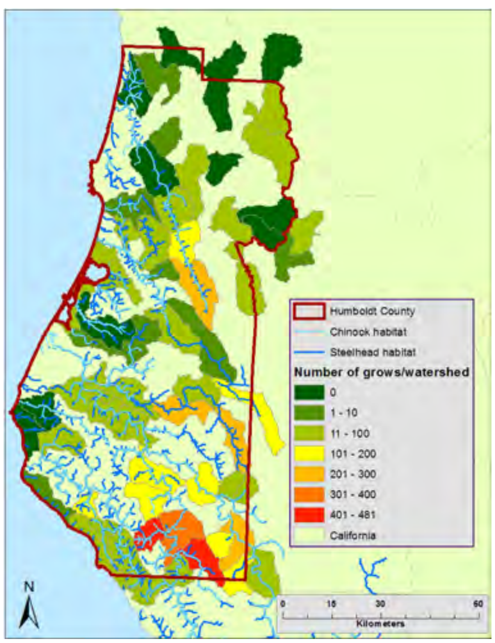

Figure 1—Humboldt County and number of grows per watershed.

Second home ownership and tourism is an increasing important part of the Humboldt County economy. Located about 4 hours north of the Bay Area, and comprised of scenic terrain and ample coastline, Humboldt County is an attractive area for outdoor enthusiasts. Redwoods State Park and parts of Six Rivers National Forest are well known for their spectacular old growth forest and attract nearly three-quarters of a million visitors a year. These visitors spend nearly $3.2 billion dollars a year, making tourism one of Humboldt County's leading industries.

Cannabis Production

Cannabis can be legally cultivated in California for medicinal purposes, although the federal government still considers cannabis an illegal Schedule I drug (McGreevy 2015). Producers must be documented care givers and can supply their crop either to individuals who have physician approval to use cannabis or to dispensaries, which can sell cannabis to patients. Under the Obama administration, federal law enforcement agencies have not strongly enforced federal cannabis laws nationally, although there is precedent for federal actions on dispensaries and growers (Zilversimt 2016). Federal law typically enforces a 5 year prison sentence for cultivation sites larger than 99 plants, hence anecdotal evidence suggests that many farmers stay under that number in case of federal intervention (California Normal 2016). Currently, there is no organized program in California to track cannabis cultivation siting, production, or sales, even in the legal market. New laws passed in 2015 aim to establish such a system by 2018 (McGreevy 2015).

There is little documentation of actual practices of cannabis production in the scientific literature (Carah et al. 2015). Nevertheless, researchers have anecdotally observed several tendencies of cannabis production that are relevant to our modeling exercise. First, production takes place both outdoors and in greenhouses. Outdoor production is reliant on natural sunlight and plants are typically grown in groups or individually in raised beds. Greenhouse production allows for light to be diminished with shades or enhanced with artificial light. The manipulation of light allows growers to precisely control flowering, which gives greater control over production in terms of both the schedule and the amount produced. Finally, for most agricultural crops, soil quality is a driver of crop choice. In Humboldt County, poor-quality agricultural soil covers nearly 90 percent of the county. Therefore, many growers import soil and amendments for both outdoor and greenhouse grows. While there is no documentation of how much soil is imported, various local businesses supply soil in large quantities (e.g., www.humboldtnutrients.com, www.royalgoldcoco.com).

Past land use analysis have shown that cannabis production is clustered at the watershed scale, with some watersheds having high levels of production and others no production at all (Butsic and Brenner 2016). Cannabis production is more prevalent in the south and east of the county. Most production takes place in remote areas of the county, and many of these areas are not suited for traditional agricultural production.

Disamenities from cannabis production may reduce property values. Many cultivation sites are located off the grid, and thus rely on generators for power. Many Humboldt County landowners have complained about the constant humming in remote areas of the county caused by these generators (Stansberry 2016b). Also many growers use artificial lights to increase yield, and these lights can lead to light pollution that may be unattractive to rural residents (Stansberry 2016a). And overall, there may be an unease for some potential landowners about purchasing property near cannabis cultivation, both for cultural reasons and because cannabis cultivation is still federally illegal.

Data

Our primary dataset of property sales was purchased from Core Logic and contains over 3000 arm's length transactions. We developed this dataset from a larger list of sales, by limiting our analysis to properties where at least 90 percent of the property price was the value of the land. We limited our dataset in this way because we were not able to obtain detailed information on improvements, such as the size of a structure, number of bathrooms, or number of bedrooms. We also limited our data temporally to sales that took place from 2000 to 2015.

In order to identify what features of the property impacted sales price, we merged the sales data with a host of spatial variables. For each parcel we calculated: the percent of the property in coniferous forest, the percent in hardwood forest, the percent in mixed forest, percent in agriculture, and percent in barren land; the distance to the nearest town of at least 10,000 people, the size of the parcel in acres, the distance to the ocean, latitude, and distance from nearest paved road. In addition we calculated the percent of the parcel with slope over 30 percent, and percent of the parcel with a southern aspect.

Because cannabis is often produced on imported soils and takes little land to grow, many variables typically used to explain agricultural productivity may not fully account for whether an area is actually well suited for cannabis production. Therefore, to quantify if a parcel is well suited for cannabis production, we identified the density of cannabis plants in each of the 54 watersheds. To do this, we used the dataset developed by Butsic and Brenner (2016). Cannabis density in a watershed acts as a proxy for overall suitability of a watershed for cannabis production.

Estimating the Hedonic Model

In order to identify the impacts of cannabis production on property prices, we employ a hedonic pricing model (Rosen 1974). The intuition behind the hedonic model is that a piece of property is comprised of various attributes that contribute differently to the overall price of the property. Often, these attributes are grouped as structural, locational, neighborhood, and environmental characteristics (Champ et al. 2003). By including these parcel characteristics in a regression framework, we can identify the contribution of each to the overall price. In our specific case we estimate the hedonic model of property *i* in watershed *w* sold in year *t* as:

$$\ln(price)_{itw} = B_1 + \beta eco_{iw} + \phi property_{iw} + \theta distance_{iw} + \Phi zone_{iw} + \Gamma cannabis_w + \rho_t + e_{iw}$$

We take the natural log of price in order to limit the impact of high priced properties that may be outliers in the data. Where eco represents a vector of variables pertaining to the ecological characteristics of each parcel (percent of property with slope over 30 percent, percent of each land cover class in the parcel, percent of property with a south facing aspect slope), $property$ a vector of characteristics pertaining to the features of the property (acres, acres^2, assessed value of improvements), $zone$ a vector with the zoning classification of each parcel (including having a timber harvest plan within the past 15 years), $distance$ is the distance to various important features (town, stream, road and ocean), and $cannabis$ is the number of cannabis plants per acre in the watershed where the property is located.

Results

Results of the model suggest that higher intensity cannabis production is positively associated with land prices. In the sample that includes all properties larger than 1 acre (0.4 ha), price per acre increased by 24 percent for a one plant per acre (0.4 ha) increase in cannabis density (table 1). For the sample of properties greater than 2 and 20 acres (0.81 and 8.1 ha), the corresponding price per acre increases are 27 and 25 percent, respectively. The coefficient estimates are significant at the 5 percent level using heteroscedasticity robust standard errors. With the exception of the 1 acre (0.4 ha) sample, the results are significant at the 10 percent level when calculated with robust standard errors clustered at the watershed level (table 2).

To better interpret the coefficient estimates, note that a one plant per acre (0.4 ha) increase would be an extraordinarily large expansion in cannabis cultivation. To put the results in perspective, the median watershed in our sample has a plant per acre (0.4 ha) density of 0.14. If cannabis density doubled from that number, our results imply a concomitant property price per acre(0.4 ha) increase of 3.4 percent in the sample of all properties larger than 1 acre (0.4 ha).

Table 1: Hedonic estimates with robust SEs

	Parcels > 1 acre	Parcels > 2 acres	Parcels > 20 acres
Cannabis plants per acre (watershed)	0.242	0.273	0.245
	(0.108)**	(0.107)**	(0.117)**
Assessed improvements ($1000s)	0.025	0.024	0.021
	(0.004)***	(0.004)***	(0.003)***
Parcel size (acres)	-0.006	-0.006	-0.004
	(0.001)***	(0.001)***	(0.001)***
Acres^2	0.000	0.000	0.000
	(0.000)***	(0.000)***	(0.000)***
% slope > 30%	-0.271	-0.250	-0.224
	(0.182)	(0.186)	(0.201)
% mixed forest	-0.150	-0.084	0.005
	(0.144)	(0.148)	(0.181)
% hardwood	-0.171	-0.199	-0.038
	(0.184)	(0.198)	(0.219)
% shrub	0.391	-0.026	-0.503
	(0.454)	(0.347)	(0.630)
% coniferous	0.301	0.267	0.091
	(0.140)**	(0.145)*	(0.194)
% barren	0.094	-0.129	-0.097
	(0.256)	(0.276)	(0.397)
Ln (distance to road)	-0.141	-0.041	0.010
	(0.095)	(0.092)	(0.097)
Distance to stream (KMs)	-0.013	0.019	0.071
	(0.055)	(0.057)	(0.104)
% facing south	0.105	-0.008	0.074
	(0.130)	(0.131)	(0.157)
THP in last 15 years	0.059	0.123	0.222
	(0.108)	(0.106)	(0.113)**
Ag exclusive zone	-0.759	-0.507	-0.033
	(0.117)***	(0.114)***	(0.127)
Ag zone	0.096	0.410	0.400
	(0.126)	(0.133)***	(0.293)
TPZ	-0.886	-0.613	-0.014
	(0.131)***	(0.128)***	(0.131)
Forest/rec zone	-0.798	-0.458	0.224
	(0.125)***	(0.125)***	(0.134)*
City	0.202	0.063	-0.517
	(0.161)	(0.180)	(0.273)*
Unzoned	0.632	0.569	0.190
	(0.103)***	(0.115)***	(0.172)
Distance to ocean (100 KMs)	-1.700	-1.731	-2.025
	(0.330)***	(0.334)***	(0.369)***
ykm	-0.001	-0.001	-0.002
	(0.001)	(0.001)	(0.002)
Distance to a city (100 KMs)	-0.323	-0.384	-0.417
	(0.083)***	(0.085)***	(0.094)***
R^2	0.52	0.51	0.41
N	1,422	1,229	751

Dependent variable is recorded per acre sales price. Robust standard errors in parentheses. * $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$.

Table 2: Hedonic estimates with clustered SEs

	Parcels > 1 acre	Parcels > 2 acres	Parcels > 20 acres
Cannabis plants per acre (watershed)	0.242	0.273	0.245
	(0.225)	(0.178)	(0.130)*
Assessed improvements ($1000s)	0.025	0.024	0.021
	(0.003)***	(0.004)***	(0.003)***
Parcel size (acres)	-0.006	-0.006	-0.004
	(0.001)***	(0.001)***	(0.000)***
Acres^2	0.000	0.000	0.000
	(0.000)***	(0.000)***	(0.000)***
% slope > 30%	-0.271	-0.250	-0.224
	(0.224)	(0.200)	(0.218)
% mixed forest	-0.150	-0.084	0.005
	(0.185)	(0.121)	(0.125)
% hardwood	-0.171	-0.199	-0.038
	(0.227)	(0.178)	(0.207)
% shrub	0.391	-0.026	-0.503
	(0.403)	(0.301)	(0.613)
% coniferous	0.301	0.267	0.091
	(0.198)	(0.161)	(0.179)
% barren	0.094	-0.129	-0.097
	(0.237)	(0.143)	(0.205)
Ln (distance to road)	-0.141	-0.041	0.010
	(0.133)	(0.117)	(0.115)
Distance to stream (KMs)	-0.013	0.019	0.071
	(0.057)	(0.039)	(0.068)
% facing south	0.105	-0.008	0.074
	(0.150)	(0.142)	(0.165)
THP in last 15 years	0.059	0.123	0.222
	(0.115)	(0.104)	(0.112)*
Ag exclusive zone	-0.759	-0.507	-0.033
	(0.166)***	(0.131)***	(0.114)
Ag zone	0.096	0.410	0.400
	(0.192)	(0.156)**	(0.197)**
TPZ	-0.886	-0.613	-0.014
	(0.223)***	(0.187)***	(0.132)
Forest/rec zone	-0.798	-0.458	0.224
	(0.242)***	(0.203)**	(0.160)
City	0.202	0.063	-0.517
	(0.156)	(0.135)	(0.189)***
Unzoned	0.632	0.569	0.190
	(0.138)***	(0.132)***	(0.172)
Distance to ocean (100 KMs)	-1.700	-1.731	-2.025
	(0.500)***	(0.436)***	(0.361)***
ykm	-0.001	-0.001	-0.002
	(0.002)	(0.002)	(0.001)
Distance to a city (100 KMs)	-0.323	-0.384	-0.417
	(0.129)**	(0.097)***	(0.080)***
R^2	0.52	0.51	0.41
N	1,422	1,229	751

Dependent variable is recorded per acre sales price. Robust standard errors in parentheses. * $p < 0.1$, ** $p < 0.05$; *** $p < 0.01$.

Non-cannabis characteristics also influence the sales price of the properties in our data set. Higher proportion of coniferous forest cover increases the property value, while barren and other forms of forest cover are associated with lower sales prices. However, estimates of the impact of different forest cover types on sales price are marginally significant overall, and vary with the parcel size cutoff of the sample. For example, the positive and significant impact of coniferous forest cover is diminished greatly when looking at parcel sizes over 20 acres (8.1 ha). Indeed, the difference in the magnitude of forest type coefficients is small in the larger parcel sample more generally.

Similarly, the impact of zoning variables varies by the parcel size cutoff. In the samples that includes parcels less than 20 acres (8.1 ha), prices for property in ag-exclusive, forest/recreational or timber production zones (TPZ) are much lower than other zoning classifications. However, those effects are moderated when looking only at the sample with larger properties. Likewise, no price difference is found for a property with a history of a timber harvest plan (THP) in the smaller parcel sample, but a THP is associated with a 21 percent price per acre increase in the sample restricted to properties greater than 20 acres (8.1ha). These differences likely reflect the fact that smaller parcels are undesirable for production (either timber or agricultural) purposes, so smaller parcels with these zoning restrictions likely carry higher conversion costs to residential development (and hence fetch lower prices).

Properties further inland and further from cities are significantly less valuable. However, distance to a stream and latitude does not significantly affect price, all else constant. While properties further from a known road are significantly less valuable in the sample that includes properties less than 20 acres (8.1 ha), no effect is found in the large property sample. That difference may reflect the existence of privately constructed and maintained dirt roads that exist on large properties in more remote areas that are not visible in our dataset.

Discussion

Changes in rural economies have large impact on land prices for different land uses. Here, we investigate the impact of an expanding and economically important land use: cannabis production. Using Humboldt County as our case study, we used the hedonic method to estimate the impact of cannabis grows on property prices. Our findings suggest that the increases in productive capacity of land brought about by cannabis production outweigh the negative disamenity impacts of cannabis production and that cannabis has a positive and statistically significant impact on property prices in our study area.

The past decade has seen significant changes to state and local policy towards cannabis, and the next decade will likely bring further transitions to the regulatory framework surrounding its production and consumption. In California, a 2016 ballet initiative regarding recreational legalization has prompted considerable discussion of the future role of cannabis in the state’s economy. While much of the policy debate has centered on potential tax revenue from retail level sales, our research highlighted potential secondary economic impacts on the rural economy.

Despite our results, we are unable to definitively claim our estimated positive relationship between cannabis production and property prices is causal. If there are unobserved factors driving both property prices and the location of cannabis farms, the relationship estimated here may be biased upwards. Further, we also cannot determine whether the positive influence of cannabis on property values stems directly from the higher potential productive value of this activity, or indirectly from higher local incomes due to the crop. Further planned projects will attempt to remedy these issues by identifying exogenous sources of variation in cannabis farm location and property prices.

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Family Forest Owners in the Redwood Region: Management Priorities and Opportunities in a Carbon Market[1]

Erin Clover Kelly,[2] Joanna Di Tommaso,[3] and Arielle Weisgrau[2]

Abstract

California's cap-and-trade carbon market has included forest offset projects, available to all private landowners across the United States. The redwood region has been at the forefront of the market, creating the earliest forest carbon projects. From carbon registries, we compiled a database of all forest carbon projects in the market, in order to determine where projects were located, what types of landowners (e.g. industrial, non-industrial, tribal, timber investment management organizations [TIMO]) were participating, and how projects were being developed.

Notably, non-industrial private forest landowners or "family" forest landowners were underrepresented within the market relative to their landholdings. We conducted a survey of family forest landowners in several forested regions across California, in order to determine landowner management objectives and willingness to participate in the carbon market, including obstacles and incentives for participation, and how carbon markets coexist with management objectives. We found that, though many of the carbon market objectives align well with family forest landowner objectives, the burdens of entering the market discourage participation. Further, using cluster analysis, we grouped family forest owners in California according to three management objective types, which we labeled Amenity, Legacy, and Income groups. These three groups had different views of the carbon market and climate change. If family forest owners are to be included in this or other carbon-sequestration incentive programs, the management motivations and constraints of distinct landowner types need to be considered.

Introduction

In January 2013, California implemented the first forest carbon offset program in the United States under a regulated (cap-and-trade) market. Part of the market, through a program of Improved Forest Management (IFM), has functioned to encourage forest management that increases carbon sequestration and storage in private forests. Forest landowners voluntarily join the program, but once in the program they are in a regulatory market with stringent protocols that can entail great expense, including carbon-specific inventory requirements, third-party verification, and 100-year obligations to maintain carbon stocks. These requirements have shaped market access, with most economically marginal and small-scale landowners excluded from the market. This paper presents landowner participation data for the market as a whole[4], focused on what landowner types have joined the market and in what regions, with emphasis on non-industrial private landowners and their motivations and constraints within the cap-and-trade market. While these landowners are underrepresented within the regulated market, findings help us understand family forest owners' views toward payments for ecosystem services (PES) programs, including potentially other carbon payment schemes that could include family forest owners.

Forest land ownership in the United States includes public (44 percent) and private (56 percent) landowners (table 1). Public agencies are typically non-participating in the California forest offset

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[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.

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[4] We include all projects registered on the Climate Action Reserve and American Carbon Registry, though a small proportion of these projects are still in the voluntary market.

market, which leaves 56 percent of land eligible for the market, as forests across the country may participate in the market. The most common type of private ownership in the United States is non-industrial private forest land ownership or “family forest” land, which comprises 62 percent of private forests. Private corporate ownership, including both industrial (mill-owning) landowners and institutional investors, constitute 33 percent of private forests.

Table 1—Forest land area and Improved Forest Management (IFM) projects in the United States by ownership type, 2007 (Smith et al., 2009), and percent IFM carbon projects

Ownership type		Number of forested ha in US	Percent of forested ha in US	Percent of eligible (private) forested ha in US	Percent of IFM project by area	Percent of IFM projects
Private noncorporate	Non-industrial private forest (NIPF)	99.6 million	35%	62%	2.3%	11%
	Other (conservation, tribal, etc.)	8.5 million	3%	5%	35%	44%
Private corporate[a]		55.8 million	18%	33%	62%	45%
Public agencies		133 million	44%	-	-	-
Unknown		-	-	-	1.6%	3.2%
Total[b]		303.9 million	100%	100%	100%	100%[b]

[a] Includes industrial and TIMO-owned forests.
[b] Rounding may cause the total values to deviate from 100%.

Table 1 makes clear that a high proportion of private corporate, conservation, and tribal landowners have enrolled projects in the California market. NIPF landowners, on the other hand, have a disproportionately low percentage of their land enrolled. This confirms findings by a number of researchers that the requirements to enter carbon markets are too costly and complex for most NIPF landowners (Charnley et al. 2010, Fletcher et al. 2009, Markowski-Lindsay et al. 2011, Miller et al. 2012, Thompson and Hansen 2012). This is especially true for the California market, which has more complex, stringent requirements than previous (voluntary) markets (Schmitz and Kelly 2016).

The redwood region is in many ways central to the functioning of the California offset market. The earliest projects in the market were created by land trusts in the redwood region, experimenting with nascent protocol versions to fine tune requirements (fig. 1; see also Schmitz and Kelly 2016). The redwood region was therefore an incubator of the IFM program, with landowners outside the redwood region entering the market over time and particularly beginning in 2013 when the cap-and-trade program began.

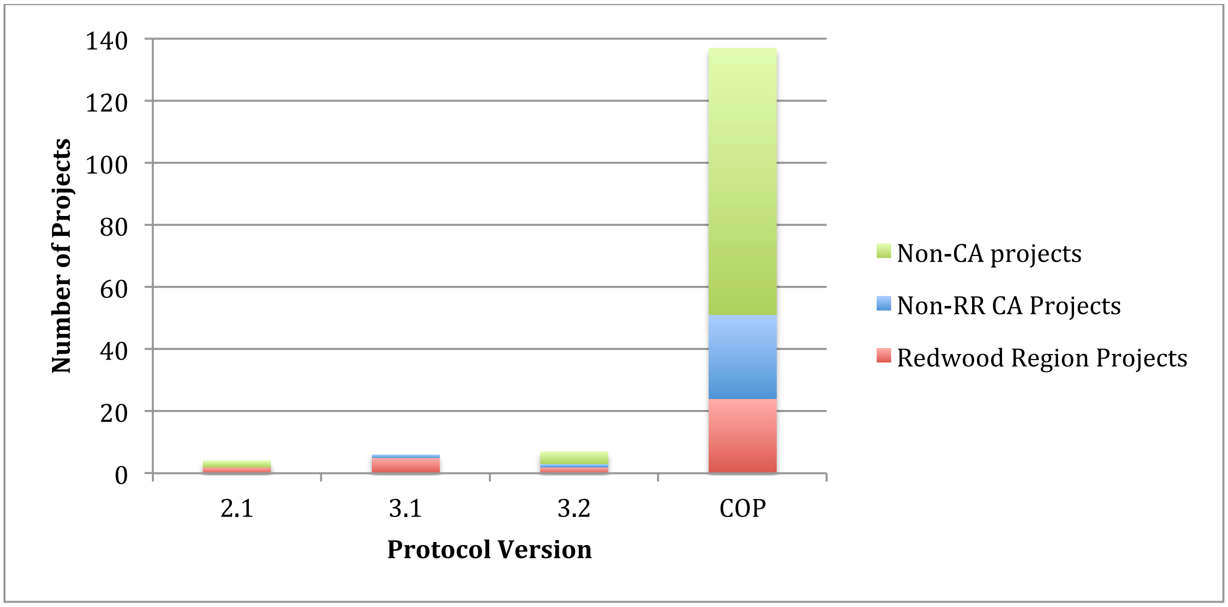

Figure 1—Projects developed under the California IFM protocols, from earliest (version 2.1) projects in 2006 through cap-and-trade (COP) versions, which were developed beginning in 2013.

The redwood region also has the greatest number of projects under NIPF ownership, with almost 70 percent of the NIPF hectares enrolled in the market. This reflects the fact that the redwood region has a landowner enrollment pattern that differs from the rest of the United States (table 2), with a much higher proportion of NIPF and conservation projects than the rest of the United States, and lower TIMO and miscellaneous projects.

Table 2—Hectares of land enrolled in the California market by each landowner type (RR = redwood region; Non-RR = rest of United States)

	RR ha in market	% of RR ha in market	Non-RR ha in market	% of Non-RR ha in market
Conservation	57,590	32.9%	118,227	6.2%
Industrial	57,552	32.9%	603,034	31.5%
Miscellaneous[a]	313	0.2%	84,470	4.4%
NIPF	33,198	19.0%	14,305	0.7%
Tribal	20,291	11.6%	410,170	21.4%
TIMO	3,659	2.1%	641,388	33.5%
Unknown	2,246	1.3%	30,646	1.6%
TOTAL	174,849	100.0%	1,916,774	100.0%

[a]Miscellaneous ownership projects include a hunting club, a church, a gated community, and several carbon developer-owned projects, among others.

We tested NIPF landowners' willingness to enter the market—and the constraints and opportunities they identified—because of their underrepresentation in the market, but also because family forest owners are known to have diverse ownership objectives (Majumdar et al. 2008), many of which may already align with IFM objectives, such as managing for longer rotations and non-timber revenue. NIPF owners may also benefit greatly from joining the market, as they are less likely than other landowners to have management plans, and they often have intergenerational transfer issues that threaten their ability to maintain their forest lands intact (Best and Wayburn 2001, Butler 2008). Finally, the views of NIPF landowners toward the IFM program and its particular constraints and opportunities allow us to consider how to better tailor other payment for ecosystem services programs in the future.

Methods

Database

For landownership data in the introduction, we compiled all project information from online offset registry databases. We included all projects registered on the two official California market databases, American Carbon Registry and Climate Action Reserve, up to September 2016, excluding projects that were double-counted, voluntary, or completed, for a total of 155 projects.

Survey

We mailed a survey with questions related to forest ownership objectives, forest uses, future management plans, and viewpoints regarding forest carbon offset project development. We incorporated questions from the National Woodland Owner Survey regarding general forest ownership and management information (Butler et al. 2005), and questions related to climate change attitudes developed by Markowski-Lindsay et al. (2011). We also included an insert describing California's carbon market and the role of forest offsets in meeting emission reduction goals. Survey materials were reviewed by experts (n = 15), including University of California extension researchers, local foresters, offset project verifiers, university professors, and a local forest landowner.

We targeted non-corporate forest landowners in California with more than 40.5 ha (100 ac), identified as smaller than the minimum parcel size that could theoretically support a financially viable offset project. We included financially non-viable acreages because of the possibility of future aggregation protocols. We sampled from five northern California counties previously surveyed by Ferranto et al. (2011), representing three forest bioregions:

1. Klamath/North Coast bioregion: Humboldt, Mendocino, and Shasta (western region) counties.
2. Modoc bioregion: Shasta (eastern region) and Plumas (Northwest) counties.
3. Sierra bioregion: Sierra and Plumas counties.

We utilized ArcMap to identify parcels that met our sampling parameters, and contacted all five counties to obtain GIS parcel data and parcel numbers for sampled parcels. All duplicate and corporate landowner names were dropped from the sample. We randomly selected 200 landowners from Humboldt, Mendocino, and Shasta counties, which all have large numbers of landowners. We sent surveys to all 82 Sierra County and 83 Plumas County landowners because each county had few landowners. For most of the findings of this paper, we utilized only the surveys from Humboldt and Mendocino counties, which we termed the "redwood region," though many of the projects occurred on mixed-conifer forests.

We ensured confidentiality by assigning case numbers to surveys; after adjusting for undeliverable addresses, survey packets were mailed to 754 landowners following the Dillman survey method (Dillman et al. 2009). Landowners were mailed a postcard a week after the initial mailing thanking participants for completing the questionnaire, and reminding non-responders to do so. Those who did not return the questionnaire were mailed a replacement booklet 1 month later. Data were organized and analyzed using IBM SPSS Statistics software.

Survey Results

A total of 165 completed surveys were sent back (response rate = 21 percent). We excluded land trust and industrial forest landowners, resulting in 142 usable surveys from all five California counties. We received 75 usable surveys from the redwood region.

Beliefs Regarding Climate Change and Knowledge of the Market

Forest carbon projects are intended to mitigate climate change through increased sequestration of greenhouse gas emissions. We therefore asked RR landowners how they viewed climate change, with fewer than half indicating they believed human activity is causing climate change, though more than half indicated that humans are responsible for alleviating climate change and that forests can reduce the impact of climate change (fig. 2).

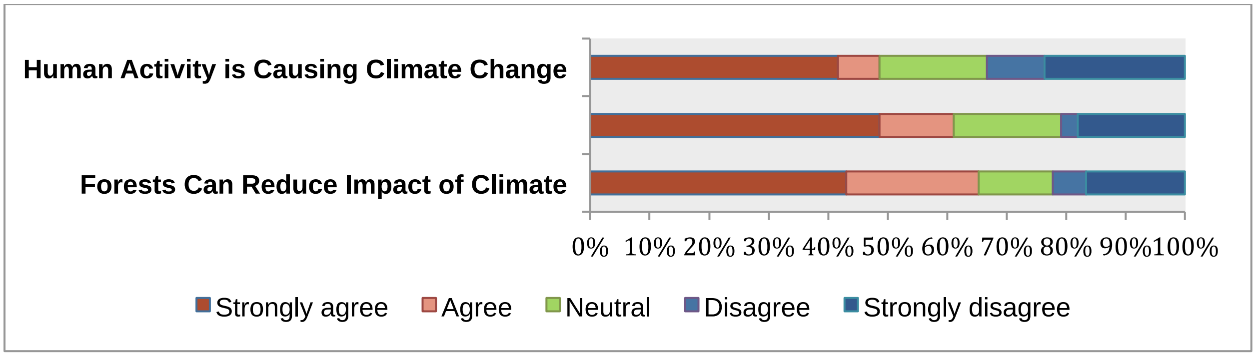

Figure 2—Landowners in Humboldt and Mendocino counties: views on climate change, and humans' responsibility to alleviate climate change.

Because the IFM program was developed in California and largely tested in the redwood region, we hypothesized that landowners of the region would have some knowledge of the program. We found that over 60 percent of landowners did have knowledge of the market prior to receiving the survey (fig. 3). Of those who had some knowledge of the market, information sources varied, with news or other media the most common source and Registered Professional Foresters (RPFs) the second most common source (fig. 3).

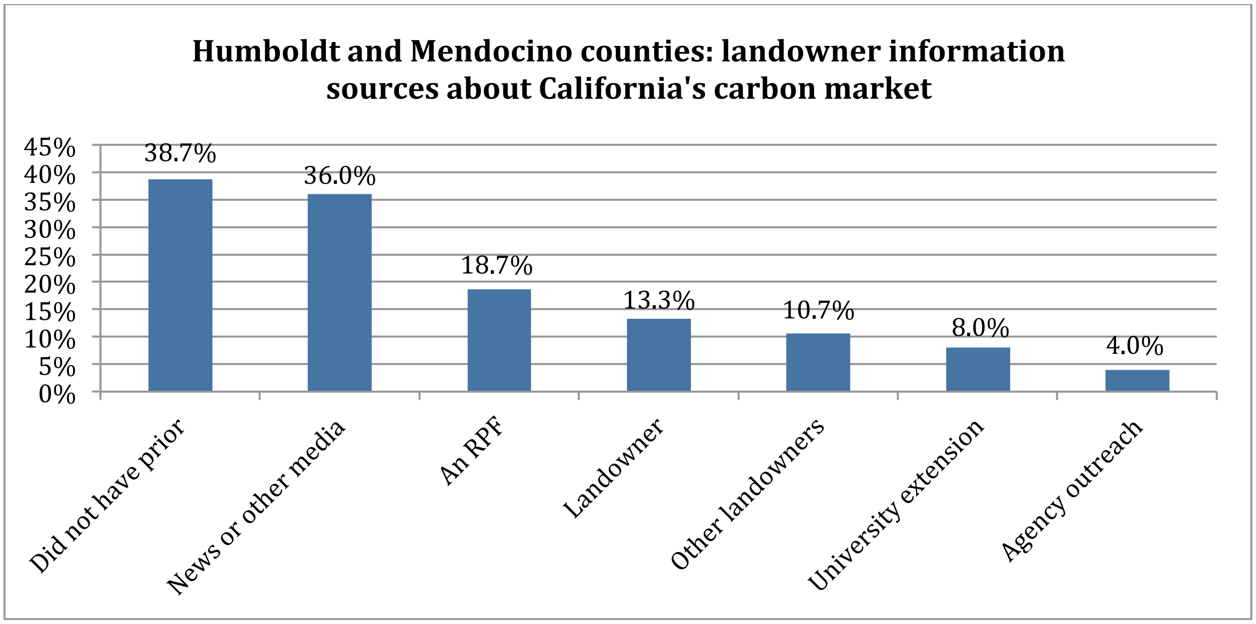

Figure 3—Sources of information regarding IFM projects and the California cap-and-trade program for landowners in the redwood region.

Willingness to Join the Market, and Constraints for Doing So

Redwood region landowners indicated they were unlikely to enroll in the market because of the obligations of the market, with or without financial assistance, though a high proportion indicated they “didn’t know” whether they would enroll (fig. 4).

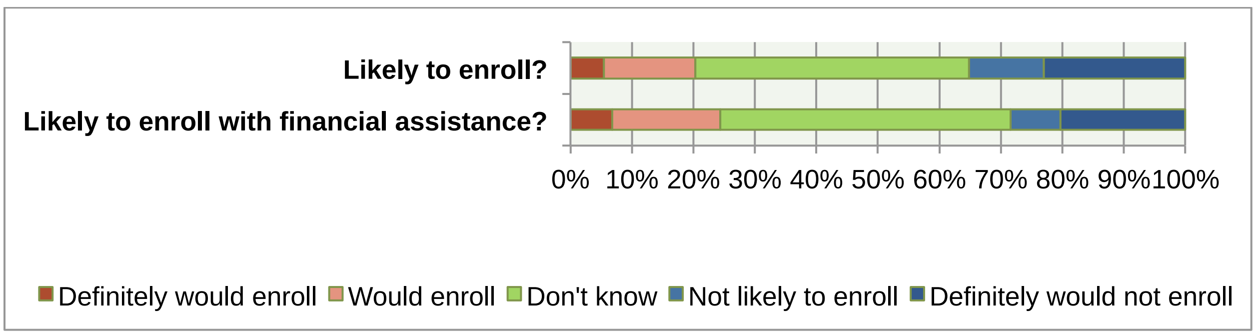

Figure 4—Likelihood of Humboldt and Mendocino county landowners to join the market.

This supports numerous previous findings and was an expected result. The survey was not only intended to gauge landowners’ perceptions of the market, but also to ask about possible motivations and constraints regarding the market, and to inform them about the market. We therefore walked landowners through market obligations, and asked landowners about the specific concerns that may constrain market participation. The survey detailed obligations related to time commitment, upfront project development, ongoing verification, and monitoring and reporting requirements. Figure 5 is a representative selection of landowner concerns, reflecting the importance of upfront and ongoing costs and the complexity of project development, with lesser concerns related to granting access to land and time commitment concerns surrounding the possibility of selling land and changing management decisions (fig. 5). For this figure, “finding time” and “granting access” were related to initial project development, but we found that concerns about finding time and granting access for verification and monitoring were similar.

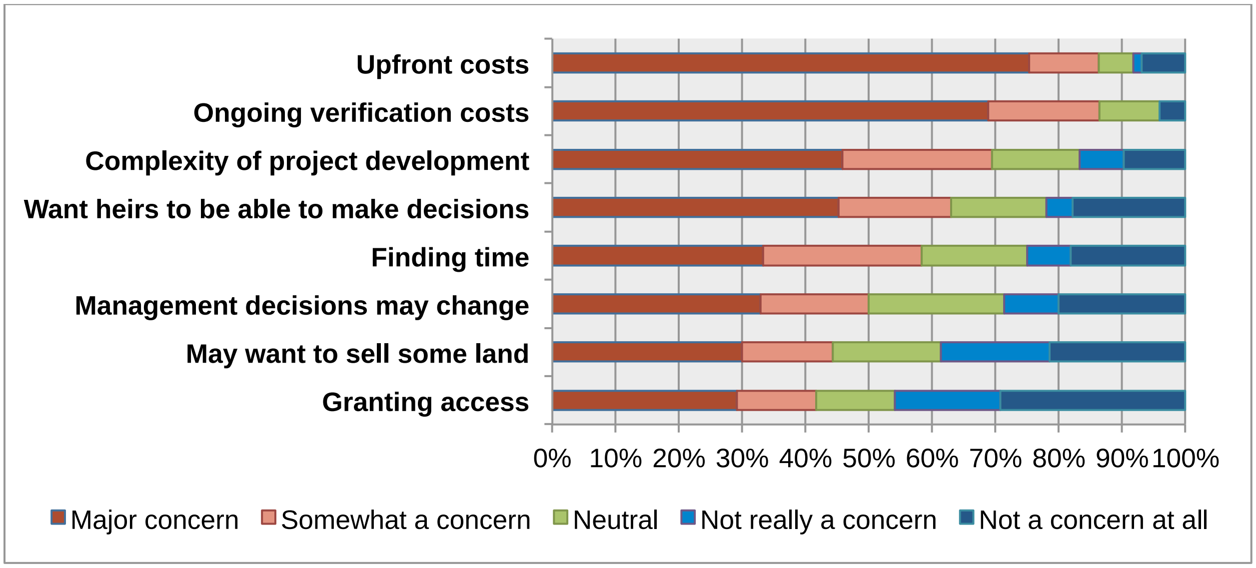

Figure 5—Redwood region landowner concerns regarding enrollment in the California market.

Ownership Objectives

Redwood region landowners have diverse ownership objectives (fig. 6). Figure 6 includes the 10 most commonly identified landowner objectives. This result was also expected; it underscores the complexity of NIPF ownership and the many considerations of NIPF landowners toward management decisions on their lands.

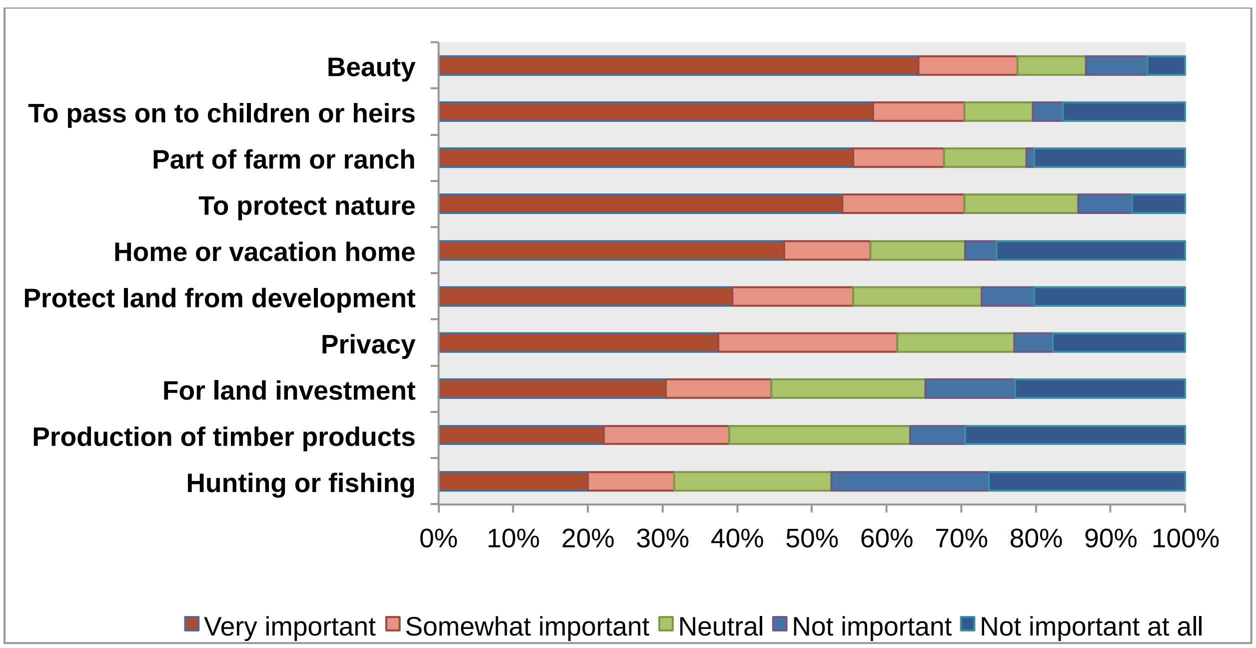

Figure 6—Ten most common ownership objectives for land ownership in the redwood region.

Because the IFM program may provide an alternative source of revenue for these multi-objective NIPF landowners, we asked about possible motivations for participating in the market (fig. 7).

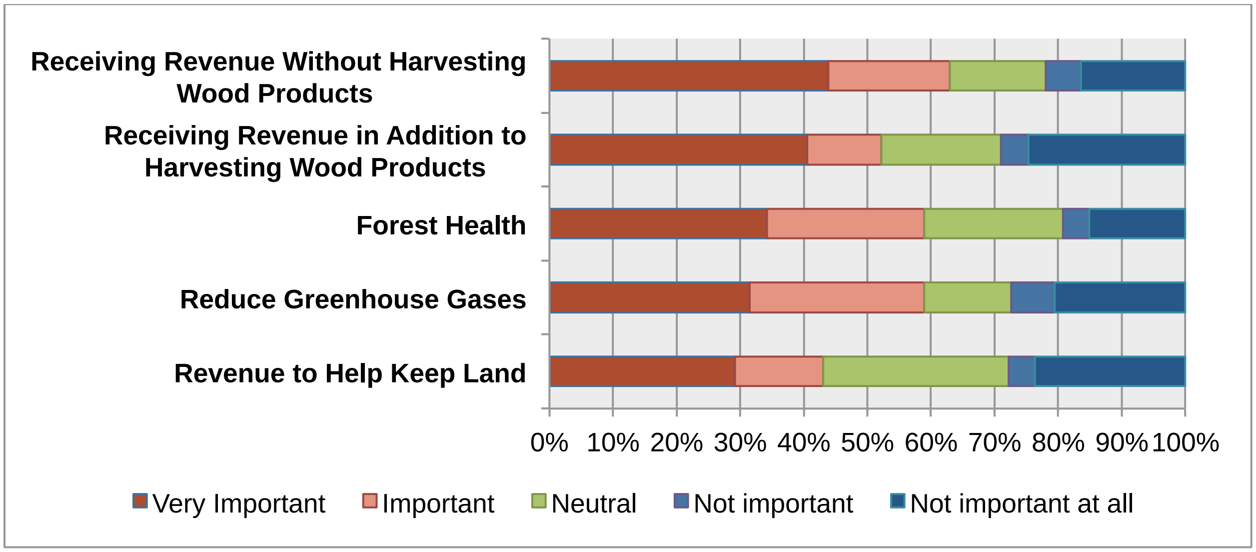

Figure 7—Landowners in Humboldt and Mendocino counties' potential motivations for joining the market.

These motivations reflect the diverse objectives of NIPF landowners, and show a particular interest of NIPF landowners in receiving revenue that is in addition to or separate from timber harvest.

Discussion

In common with previous research, we found that NIPF owners were generally unwilling to join the forest offset market, which may represent a missed opportunity for the IFM program—assuming that the program wishes to have high acreages enrolled—as family forest owners have extensive landholdings in the United States. These landholdings are possibly an opportunity for the IFM program, but also an opportunity for other payment for ecosystem services programs that aim to provide habitat, and maintain open space. NIPF landowners have diverse objectives, many of which are not related to timber harvesting, which may align well with the California carbon market protocols. But they have low levels of knowledge about the market, and concerns about entering a market with high upfront costs and complex requirements. With this and other payment for ecosystem services markets, NIPF landowners likely need incentives to participate, including monetary incentives and expertise.

This paper, among many others, has demonstrated that NIPF landowners have diverse objectives in addition to timber harvesting. We feel that these diverse objectives (including not cutting timber) could fit with the California cap-and-trade market protocols. However, some researchers have found that the forestry protocols of the California market do not result in optimal carbon sequestration because of underrepresentation of carbon sequestered in wood products and the effects of substitution (e.g. using wood instead of concrete or plastics). When wood products and substitution are taken into account, commercially-managed forests may sequester higher levels of carbon compared with less intensively managed stands (van Kooten et al. 2015; see also Perez-Garcia et al. 2007). This suggests that incentivizing *more* timber harvesting may have net carbon benefits. Thus protocol design has a significant impact on how forests are managed for carbon markets, and any findings about landowner willingness to enter these markets are limited to the protocols under consideration.

While our findings grouped NIPF landowners together, there is another way to interpret landowner motivations, by clustering landowners into objective types. Elsewhere[5], we used responses

[5] Di Tommaso, J.; Kelly, E.C.; Gold, G. The willingness of family forest owners to enter California's carbon offset market. Manuscript in review.

from the entire California sample and created a typology based on 16 landowner objective questions. We utilized cluster analysis to place landowners into discreet groups following an altered clustering method developed by Kuuluvainen et al. 1996. We found three landowner types based on objectives: an amenity group, characterized by concern for non-timber land ownership values, including aesthetic values and protecting nature; a legacy group, primarily motivated by concerns about passing land on to children and maintaining a farm or ranch; and an income group focused on the production of timber products and other economic benefits such as land investment. Among these groups, we found some distinctions in attitudes toward the market, its constraints, and possible motivations for joining it. For example, we found distinctions between amenity landowners, who were more likely to believe in climate change and to be motivated to potentially join the market in order to curb greenhouse gases. On the other hand, legacy landowners were less likely than others to believe in climate change, and indicated that curbing greenhouse gas emissions was not a motivator. Income landowners were the most familiar with the market, and most likely to receive information from RPFs; they also stated lower levels of concern regarding several market obligations, including finding time to participate and granting access to their land to professionals. These distinctions among landowner types were not found when we considered the Humboldt and Mendocino landowners in isolation, but similar landowner groupings have been found elsewhere (Kline et al. 2000, Kuuluvainen et al. 1996, Majumdar et al. 2008).

Considering the diversity of ownership objectives and landowner types in the redwood region, therefore, it is important to consider a diversity of outreach and education programs for joining the IFM program, if that is an objective. Amenity landowners may be motivated by the non-extractive PES markets. Outreach to legacy landowners could focus on the importance of maintaining working lands and natural resource-based livelihoods, and the opportunities of alternative sources of income for maintaining family ownership. Income landowners could see the benefits of multiple revenue streams, especially in regions that have declining forest products infrastructure or for forests that are economically marginal in terms of wood products.

However, the underrepresentation of family forest owners within the carbon offset market does not mean that more landowners need to be pushed into a potentially unsuitable market. Rather, similar payment for ecosystem services programs could be developed that are more response to landowners' concerns and objectives.

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Plantations as a Response to the Creighton Ridge Fire: a Landscape Experiment in Cazadero, California[1]

Frederick D. Euphrat,[2] Charles Williams,[3] and Judy Rosales[4]

Abstract

During a period of unusually hot, dry weather in 1972, the Creighton Ridge fire burned 4,452 ha (11,000 ac) of forest and intermixed grasslands, as well as many residences on the recently-subdivided 16 ha (40 ac) ranch holdings in the Cazadero – Fort Ross area, north of San Francisco. In response to the fire, local work crews planted and thinned trees from 1981 to 2000 with State of California economic and technical assistance.

Plantations of ponderosa pine (*Pinus ponderosa* Douglas ex C. Lawson) and Coulter pine (*Pinus coulteri* D. Don) were to serve as 'nurse trees' allowing ingrowth of shade tolerant Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and redwood (*Sequioa sempervirens* (D. Don) Endl.). About a million pines, Doulgas-fir and redwood were planted on about 1,214 ha (3,000 ac). Where nurse trees were thinned, the desired forests were achieved. Thinning of these new forests, however, has never been a funded state or landowner priority.

Now the forests are up to 45 years old and growing vigorously as dense stands. This paper evaluates several hypotheses testable in this new forest. It also reviews the potential benefits of a forest plan to mitigate plantations' impacts while creating measurable benefits to the people and ecosystems of the affected watersheds.

Introduction

The Coast Range area of Cazadero – Fort Ross, Sonoma County, California (Coast Ridge) is unique in a host of ways. Adjacent to the Pacific ocean and north of San Francisco Bay, this is the area of the most southern claim by the Russian Empire (Fort Ross in 1812). It was further influenced by Spain and Mexico in the 1820s and 1830s. Originally populated by the Southern Pomo people, land management in this area has been strongly in flux, with the succeeding 'new owners' seeking separate and individual management and economic goals. Like many areas opened in the West, the area's history is one of meeting new goals until the land, or other resource, can no longer provide, is still playing out. The larger area of the Coast Ridge Community Forest is shown in fig. 1.

Geographically, the Cazadero-Fort Ross area has unusual resources for the region. With an annual average rainfall of 1,854 mm (73 inches) (1939 to 1971) (Western Regional Climate Center 2016), the area is among the wettest in the redwood region. Once dominated by redwoods, and later the logging industry, the unique ecosystems of this area have been under extractive pressure since the arrival of Europeans. Because it is part of the San Francisco Bay area, ex-urban privatization and subdivision threaten the ecological integrity of these forests as well. Uses of this land included native burning for food and tools, game habitat, logging for construction, livestock grazing and gardening for homesteads, orchards and viticulture.

Following resource overuse, another use was found for the land. A path of extraction, intentional or otherwise, removes fur-bearing mammals, grizzly bear, salmon, old-growth redwood (*Sequioa sempervirens* (D. Don) Endl.), soil carbon and root mass and, eventually, a portion of groundwater from the ecosystem. Today, most ranches have been subdivided into 16 ha (40 ac) parcels, and the

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[1] A version of this paper was presented at the Coast Redwood Science Symposium, September 13-15, 2016, Eureka, California.

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land is a mixture of forests, prairies, agricultural and residential uses, with vineyards expanding across the landscape.

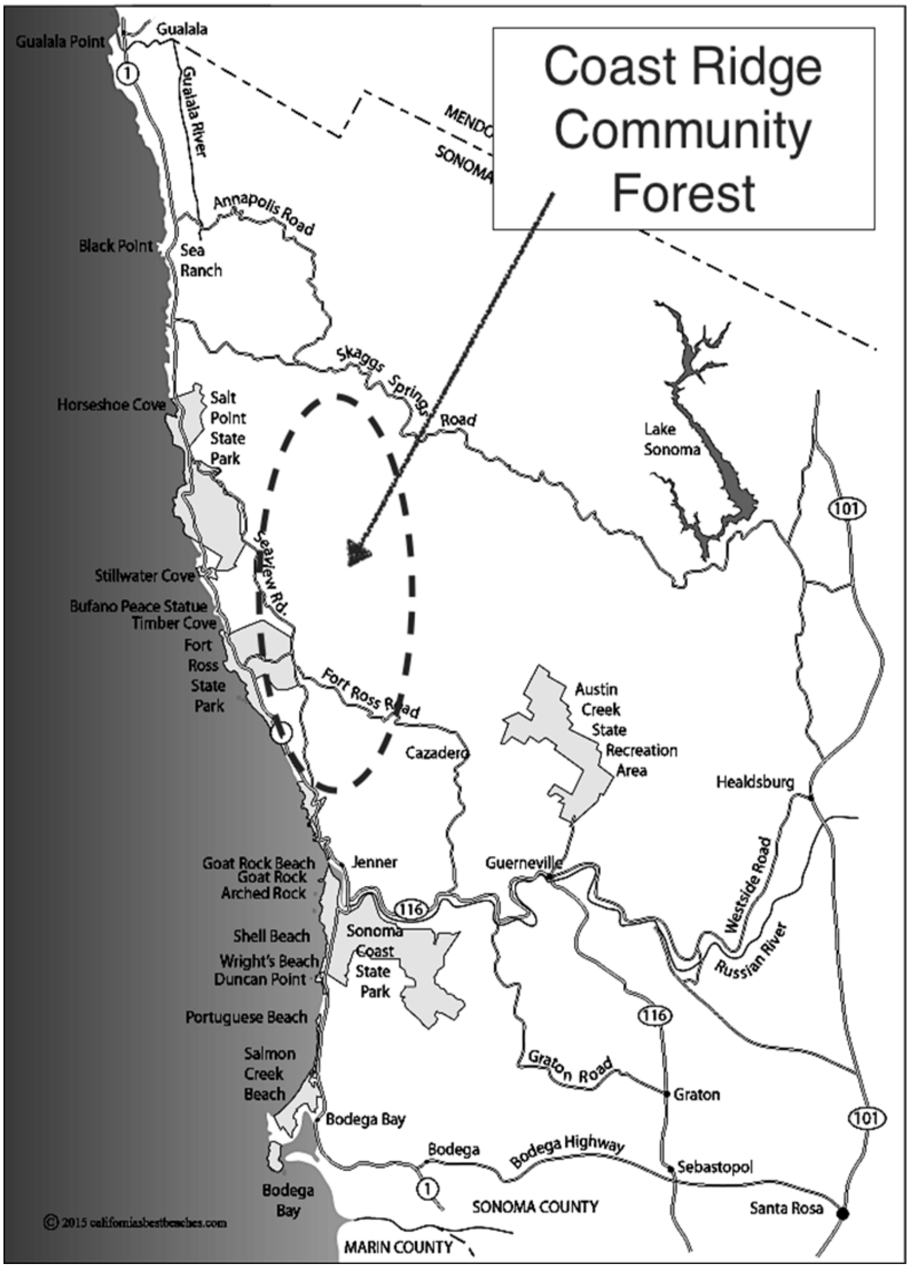

Figure 1—Location of Coast Ridge Community Forest.

Wildfire as a Landscape Pattern

Since the 1950s, seven significant 405 ha (100 ac) wildfires have burned 9, 976 ha (24,652 ac) in the Coast Ridge area. Notably, the 1978 Creighton Ridge Fire burned more than 4,452 ha (11,000 ac) and 61 structures. This is not uncommon in the region—extensive logging prior to forest practice rules created an abundance of fuels, annual grasses dominated the rangelands, and the area is remote, served by few roads. Coupled with 'fire weather,' in this case easterly breezes which block the ocean influence, the area burns readily in late summer and fall.

Figure 2 shows the array of fire on this landscape, overlapping and frequent. The years 1953, 1954, 1957, 1965 and 1978 all had large fires. Since then, fires have been quickly contained to small areas. One major reason is the development of both an on-site volunteer fire department in response to the 1978 Creighton Ridge fire. In its destruction of the lands, the fire knit the community closer.

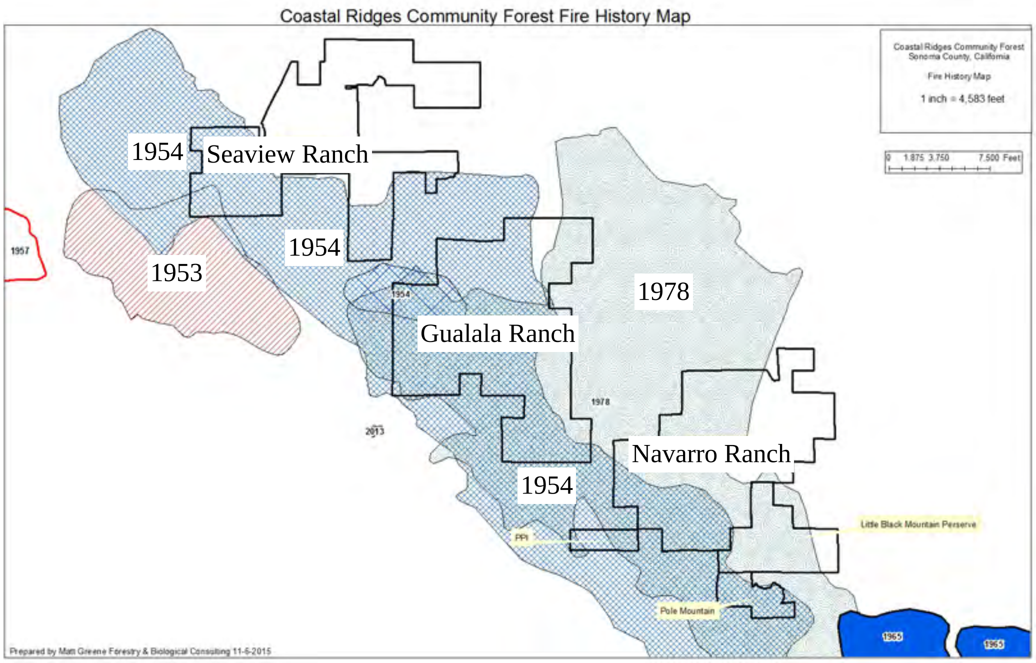

Figure 2—Fire history and subdivision, Coast Ridges Community Forest.

While a tool for natives and ranchers, fire is a devastating event on the landscape of small family farms and residences. In the Coast Ridges, four subdivisions to approximately 16 ha (40 ac) size had been created and populated by 1978: Navarro Ranches, Gualala Ranches, Seaview Ranches and Muniz Ranch. All were designed for the 'back to the land' opportunity at that time, with many urban people populating the old ranchlands. Planning for fire, apart from road sizes and turnouts as required by the county, was not in place on any newly subdivided ranch.

The new urban settlers were not cognizant of the dangers and the potential rapid spread of fire, particularly on these discontinuous ownerships. The local 'building boom' inadvertently allowed abundant regrowth on the land. When combined with residual slash, this effectively eliminated any fire-safe zones that might have been present on the earlier ranches. The 1978 fire erupted from tool use under bad fire conditions, and quickly grew to catastrophic proportions.

Community Action

After the Creighton Ridge fire, the community came together in a suite of ways. The organizational pathways for community protection have developed over time, to now include:

- Fort Ross Volunteer Fire Department
- Fort Ross environmental restoration
- Ranch associations
- Disaster preparedness points of dispensing (PODs)
- Coastal Hills community project
- Coastal Hills Land Trust
- Coast Ridge community forest
- Agency and organization partnerships

The goals of all these social outcomes are to provide fire protection, a landscape pattern of safety, community stability, and caring for the older generation. It became clear that road and vegetation management, and communication, are the critical factors in the survival of the Coast Ridges as a viable community.

The Coast Ridge Community Forest[5] describes the timeline of interactions:

1972: Fort Ross Volunteer Fire Department founded.

1978: Creighton Ridge fire burns more than 4,047 ha (10,000 ac).

1978 to 1980: Local residents join forces with Circuit Riders (a local non-profit) to restore and reforest the area devastated by the Creighton Ridge fire.

1980: Cazadero Forest Workers established to continue the reforestation work initiated by Circuit Riders and local residents.

1990 to present: Fort Ross Environmental Restoration, a cooperative corporation, established to promote sustainable forestry practices for local individuals and landowners. Projects include reforestation, restoration, forest management, and fisheries restoration.

2002 to present: Fort Ross VFD receives grant funding from Bureau of Land Management to fund three vegetation management projects within its boundaries.

2007: Coastal Hills Land Trust founded.

One of the first neighborhood actions was to start Fort Ross Environmental Restoration, a worker-owned cooperative to reforest and afforest the landscape. This group enabled the Department of Forestry to fund, with a cost-share of 90 percent, the restoration and supply it with plants from the Department seedling nurseries. It also addressed landslides and, ultimately, the reconstruction of roads (with Federal and local assistance as well). More than 1 million trees were planted on roughly 1,214 ha (3,000 ac) (David Passmore, 2016, personal communication). In addition, many individuals and organizations took advantage of this program outside of the workers' cooperative.

The rest of the landscape, which was not afforested, reforested itself with native redwoods, Douglas-fir, tanoak (*Notholithocarpus densiflora* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh) and madrone (*Arbutus menziesii* Pursh). Tanoak reforested the vast majority of untreated area, and, at a landscape scale, may be seen as the control in this experiment. The repopulation of tanoak has been extraordinary—a very large number of sprouts from root crowns, developing into a forest of small stems and a few larger trees. The nature of this forest is changing, however, with the natural and gradual introduction of the pathogen *Phytophthora ramorum*, the cause of sudden oak death (SOD), which preferentially and repeatedly kills tanoak. Managing the tanoak on the Creighton Ridge burn area has been a concern for many years.

As estimated from ground and aerial photo imagery, the Coast Ridge area is currently comprised of 20 percent pines, 20 percent tanoak, 30 percent Douglas-fir and redwood, and 25 percent grasslands. The remaining 5 percent includes farms, residential pasture and vineyards.

The Tree Planting Experiment

As readers may know, the science of tree planting for restoration has evolved since the 1970s. At that time, the California Forest Improvement Program (CFIP) required a simple management plan and guidelines for planting. Because of the belief that many trees would die in their first years, tree planting was aggressively spaced, at about 988 trees per ha (400 trees per acre), at 3 x 3 m (10 x10 ft) spacing). It was also a concern that Douglas-fir and redwood trees would not survive in the heat of the sun without shade protection, and that manual site preparation was inferior to mechanical intervention (ripping or otherwise disturbing ground).

In retrospect, some of these choices, required by the Department, were mistakes. Heavy equipment created access issues, brought in and spread scotch and French broom, compacted soils, and increased costs. The tree selection, largely ponderosa and Coulter pine with redwood and Douglas-fir, turned

[5] Website Information at coastridgecommunityforest.org.

out to be very successful, with nearly 100 percent survival (versus the 50 percent anticipated). The trees are invasive exotics to this area, however, and have succeeded in creating a 'new forest' which can seed 304.8 m (1000 ft) distant.

The concept of trees as a 'nurse crop', into which Douglas-fir and redwood would voluntarily seed and grow, was successful, though only with later intervention (enrichment planting, and thinning). The pines achieved 100 percent crown cover quickly, preventing other trees from growing with vigor unless thinned in a timely manner. Treated stands responded vigorously, with the new layer of duff acting as a mulch. The lack of a program to remove pines froze many stands at original planting dimensions. The result in untreated stands was stagnated forests without a brush layer or vigorous emerging cohort, a fire hazard.

The great benefit of the planting program was its result in community development. This project lasted for years, planting many tens or hundreds of thousands of trees (the actual number was not available for this paper). The community received revenue for hand work and machine site preparation, and a track record was established for continuing work with agencies. In addition, the community learned to work together on fire control and forest management strategies.

Experimental Results

The area not planted, which became forested mostly with tanoak, could be considered as a control. We can track the progress through historical aerial photos, we can look at individual case studies, we can evaluate a broad statistical set of randomized samples. Yet in this instance, it requires less sampling and science and more direct action; the experiment has spoken in form.

After roughly 40 years of growth without management, many current pine and Douglas-fir stands are overcrowded, with touching crowns. Stocking densities approach the original planting density, with diameters at breast height (DBH; 1.37 m) of 20.3 to 35.6 cm (8 to 14 inches).[6] There is an abundance of ladder fuels; the pine soils are allelopathic, reducing germination of other species. In short, the forests are un-thinned fire hazards spread across the landscape.

The wood produced by the forests is non-merchantable, as there is no local market for small, branchy pine trees. Some pole material and fire wood is created, but it has no commercial value, as the parcels are too small to justify the cost of logging. There is also no infrastructure to process pines, create biochar, generate electricity or otherwise capture the value of the pine. The closest market for whitewood is the Port of Oakland, where it would compete with Douglas-fir for the export market.

If the hypothesis of this experiment was to determine the possibility of growing ponderosa pine, Coulter pine and Douglas-fir in burned areas and meadows of the site location, it was success. The pine-fir forest now covers a large percentage of the Coast Ridge landscape in irregular copses of ponderosa pine.

If the hypothesis of this experiment was to test the 'nurse tree' concept of shade tolerant trees volunteering (with some amendment planting) and overtaking the nurse species, the hypothesis is rejected. The small trees were never able to outcompete their nurse crop. A pre-commercial thinning intervention may have assisted in reaching the posited outcome. Figure 3 shows the un-thinned result of successful planting.

[6] Euphrat, Frederick D. 2015. Unpublished data, SRJC forestry class survey.

Figure 3—Un-thinned planting, year 40.

If the hypothesis was that giving assistance money to a community will be a driving force in recovering from a disaster, the experiment was a success. The establishment of forest management plans, planting, and forestry work was a centralizing force in the community. Other measures, such as the volunteer fire department, the emergency PODS, the road association and its pursuit of upgraded drainage, the land trust and now the Coast Ridge Community Forest have followed with the self-organization of the community.

Associated Ecological Improvement

With the changing uses and parcelization, the soil has become degraded. The local prairie system soil built and managed with regular burning and associated with local old-growth forest soils was replaced with European grasses and western grazing systems, then burned catastrophically a set of times (to rid the grasses of the fir and redwood stumps). That degradation is interpreted as lost tilth, lost carbon, nitrogen and depth. With the passing of the perennial grasses, it is presumed that erosion increased under annuals, which may have also increased the baseline hillslope erosion rate.

To improve the prairie, the Coast Range Community Forest is implementing silvo-pastoral systems using sheep and goats. Multi pastoralism, using electric fences and moving livestock frequently allows for a controlled ecological response, promoting perennial native bunch grasses and incorporating manure into the soil. The increased carbon, tilth, and soil structure from this practice increases the fertility, stability and utility of the prairie areas (Savory 1988, Savory and Butterfield 1999).

Grazing can take place in forests and along their margins, too. The goal of this practice is to reduce invasive pines, Douglas-fir, and limit fuel loading. The selection of animals is critical, as is the grazing time window. Complete eradication of invasive species is not important, as long as enough leader and leaves are removed for reducing apical growth (on the Douglas-fir) and potential ladder fuels (on the hardwoods) (Charles Williams, 2016, personal communication).

Forests have grown abundantly on these sites. Soil has developed quickly under the dense forests, sequestering carbon, and the forests create the influences of cooler air, higher humidity, and resistance from surface erosion and mass wasting. The new forests also create habitat for endemic and introduced wildlife. If the forests are allowed to keep accumulating fuel in a highly flammable manner, these are the benefits that are at risk from fire.

Tools for Landscape-Level Response

The thousands of acres which were afforested are now in need of thinning, a market for their products, and fire prevention treatment. While similar ecologically, much of the land is divided into separate parcels of 16.2 ha (40 ac), too small and with products too low-value for ‘normal’ timber harvest practices. The pine trees from earlier plantings that received thinning have grown to be large and good timber. Unfortunately, these thinned stands are on small parcels that are not part of the commercial forest industry, and do not realize a profit for timber owners.

There are a number of programs which provide landowners cost-share funds for thinning and pruning; these require an outlay of 25 percent or more of the contract work cost. Other possibilities exist under timber harvest plans (THPs), non-industrial timber management plans, ‘La Malfa’ commercial thinning and other exemptions, a working forest management plan (still under review by the Board of Forestry) and a Programmatic Timber Environmental Impact Report (PTEIR). (CAL FIRE 2016).

Of these pathways to utilization of wood and thinning of forests across the landscape, the most reasonable approach may be the PTEIR. It is intended to create local forest rules for unique situations, such as the Coast Ridge. With a PTEIR in place, landowners could modify their forest for commercial purposes with the filing of a much-abbreviated THP.

Intrinsic to harvesting for profit is tools to get the wood from stump to market. In these steep slopes, yarding equipment is needed to bring material up and down hills. Firewood splitters, chippers or biochar equipment will be necessary for low value product. Small sawmills and stud mills would be able to utilize much of the pine product. These are big investments, however, and would are more suited to a private business or cooperative than for individual landowners.

Grazing provides a management tool for the acres of prairie amongst the forest. It creates carbon sequestration, erosion protection, fire protection and limits the spread of unwanted trees.

Realizing these landscape-level goals is the purpose of the Coast Ridge Community Forest, a non-profit membership organization focused on the lands planted in response to fire. Its website states these goals:

“In addition to improving the forest ecosystem, CRCF is striving to generate new forest products, create and expand market opportunities, and provide income opportunities for members. Working together with the overall community, CRCF will also address common land use issues such as roads, watershed, shaded fuel breaks, and invasive plants. This could include issues such as assessments, work plans and funding (CRCF 2016).

Summary

The 1978 Creighton Ridge fire in western Sonoma County, California, was a devastating blow to the community, burning 61 homes and 4452 ha (11,000 ac) of newly subdivided ranch land. The community responded with the support of state and federal agencies, planting thousands of acres and more than a million trees. The planting tested principles of reforestation in a modern context.

Forty years after the fire, there is a clear trend towards overstocking and increased fire danger. The original plantations that were not thinned ended up as very dense, tight forests with little understory success and heavy ladder fuels. The choice of tree may have been appropriate as a nurse crop for other species, but left un-thinned becomes an economic bottleneck for the landowners, and an ecological impediment for more desirable tree species.

While ecological benefits of erosion control and habitat were created, the ironic response to the original fire, planting non-endemic trees with the potential to later increase fire danger, is inescapable. There is present need for thinning and markets for this abundant wood supply.

The Coast Range Community Forest, a non-profit collaborative venture, is focused on reducing the fire danger and creating profitable revenue streams from this landscape. Its goals include management of prairies with grazing, forests with productive revenue sources, and multi-ownership projects with cooperative utilization of forest tools. The use of a PTEIR to achieve these goals may be the most functional in these small ownerships and unique ecological conditions.

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SESSION 8 – Forest Ecology

Why Are Coast Redwood And Giant Sequoia Not Where They Are Not?[1]

W.J. Libby[2]

Abstract

Models predicting future climates and other kinds of information are being developed to anticipate where these two species may fail, where they may continue to thrive, and where they may colonize, given changes in climate and other elements of the environment. Important elements of such predictions, among others, are: photoperiod; site qualities; changes in levels and yearly patterns of temperature, wind, fog and precipitation; the effects of these on interactions with other biota at each site; the effects of changes in fire frequency and intensity; the availability of seeds and seed vectors; and the effects of human activity. Examples are presented, with focus on fire and human activity. Natural migration may need assistance. Establishing groves far from the native ranges is advocated.

Keywords: assisted colonization, assisted migration, climate change, fire, *Sequoia*, *Sequoiadendron*

When preparing this talk and then paper, it became increasingly clear that it is more of an Op-Ed than a comprehensive review, and is meant for people interested in and familiar with coast redwood (*Sequoia sempervirens* (D. Don) Endl.) and giant sequoia (*Sequoiadendron giganteum* (Lindl.) Buchholz). Thus, four background references are provided, and they in turn provide detail on many of the topics covered. The final two references provide background on future speculative scenarios. Possible responses to such future scenarios are suggested.

Coast redwood's current natural latitudinal range begins with discontinuous canyon-bottom populations near the southern Monterey County border, extends north through increasingly-continuous coastal and generally-separated interior populations, and stops just north of the Oregon/California border. Where a gradient in ecological conditions becomes limiting for a species, individuals near that edge of the population usually grow less well than individuals growing in more-optimal conditions. But rather than its trees being less healthy near that northern edge, those redwoods are among the largest and most robust in its entire range, suggesting that conditions just beyond the current northern species edge would also support healthy and vigorous growth of redwoods.

Pollen deposits and other fossils indicate that redwood used to live south of its current southern population, with extirpated populations near Santa Barbara and even La Brea, and also farther north on the Oregon coast. A few planted redwoods are currently growing reasonably well in the Los Angeles Basin and, although redwood's native range stops abruptly at its northern edge, planted redwoods are thriving in some favorable locations as far north as Vancouver Island, British Columbia.

Giant sequoia's native range has a similar but latitudinally inverted pattern. Its closely-spaced native groves and largest trees are in the southern Sierra Nevada, where the climate is hotter and apparently drier than in sequoia's few and widespread more-northern native groves.

Recent fossil evidence, mostly layers of pollen deposits, indicates that sequoia has been at higher elevations during the warmer period 6,000 years ago, and lower than it is now during the last ice age. But there is no evidence of it recently or ever being north or south of its present groves within California. And like coast redwood, planted sequoias are thriving over a substantially greater

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latitudinal range, from southern Spain to part-way up the coast of Norway in Europe, and in many locations in western North America from southern California's San Bernardino Mountains to northern Oregon and beyond. Yet, there are no native sequoia groves in the Cascades and northern Sierra, and only a few in the central Sierra.

Using climate data from the native ranges of these two species, and from sites with observed performance of their planted trees in other climates, we now have a pretty good idea which climates are permissive for redwood and sequoia to survive and thrive, which are marginal for them, and which of the much larger range of climate conditions are exclusionary. If even just-modest summer rainfall is reliably well-distributed through the summer months, both species can thrive with as little as 700 mm of annual precipitation. But if summer rains are inadequate, redwood may rely on summer fogs and both species thrive on apparently-good soils with favorable hydrology supplying groundwater. Perhaps surprisingly, many well-established planted sequoias exposed to temperatures of -28 ^{o}C have survived in northern Europe, as have a few planted redwoods in southern and central Europe, and planted redwoods in California's Central Valley exposed to brief episodes of +50 ^{o}C also have survived.

So why don't they naturally occur in more of those permissive climates? First, they have to get there and, if the colonists establish, they have to successfully reproduce. For example, redwood plantations are thriving in several locations between about 1,000 and 2,000 m elevation in Hawaii. However, in remote Hawaiian plantations, thriving redwood trees fail to produce cones and (apparently) pollen. (Nearby redwoods do produce abundant cones in the presence of light breaks from buildings or automobile headlights during the night, so photoperiod seems to be important for redwood's sexual reproduction.) And of course it would have taken a strong wind or bird to get some viable redwood seeds to Hawaii naturally.

Having arrived and successfully reproduced, there may be resident insects and pathogens that harm them. For example, planted sequoias are often deformed or killed by redwood canker, a stress disease caused by the fungus *Botryosphaeria dothidia*, that infects them in near-coastal California. For reasons still unclear (to me, at least), the severity of *Botryosphaeria* damage on planted sequoias decreases with increases in elevation and latitude in both Europe and North America. It is quickly lethal on planted sequoias near sea-level in southern France, but is either benign or absent near sea-level in Denmark and Norway. In California and southern Europe, *Botryosphaeria* is not a problem for native or planted sequoias above about 800 m elevation.

Colonizing seedlings have to compete with the local vegetation. Serious competitors sequentially range from ferns, forbs and grasses to aggressive brush to other tree species, especially those trees that start faster from seed or can thrive in more shade than redwoods and sequoias can. (Small established redwoods and sequoias can *endure* many decades of overtopping shade, but unless root-grafted to overstory trees, they do not thrive unless they have full or nearly-full sunlight.)

The following observations were told to me several decades ago by Jim Rydelius, and were catalytic in my thinking about why these two species are not occupying apparently-permissive sites near their current native ranges: In the late 19th and early 20th centuries, as extensive areas of redwood forests were increasingly being harvested, ranchers often attempted to convert the newly-cut forests to grazing lands by burning the logging debris and sowing grass seeds. But many of the redwood stumps vigorously sprouted, and in typical cases many seedlings of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), plus a few of redwood and other conifers such as grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.), western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), and western redcedar (*Thuja plicata* Donn ex D. Don) established in spite of the grass. After a few years, the ranch-hands cut the encroaching young trees and then again burned the site after the grass and felled slash had dried. Each such fire killed the seedlings of the other conifers, but the redwood stump-sprouts resprouted, the larger ones producing 'fire columns' sprouting from their scorched boles. The recently-established redwood seedlings also sprouted, usually from the root-collar burl below their burned-back stem. New seedlings of Douglas-fir, redwood, and other conifers often established the following spring, but the redwood sprouts were already vigorously growing in advance of the new germinants. This process may have been repeated several more times before the rancher gave up and, in most cut-and burn cycles, additional redwoods had survived the fires. In this way, the percentage of redwood in the new forest became increasingly greater than it had been in the previous native stand, and the new redwoods enjoyed a root-size and sprout-vigor advantage over the competing conifers, which had to start over with new seedlings after each fire.

It seems possible, even likely, that as climate warms, fire severity and frequency will increase in both coastal Oregon and in the Sierra-Cascade Mountains, and the monsoons that sometimes bring useful summer rains to the southern Sierra may also become more frequent farther north. Intense stand-replacing fires may more-frequently be followed by repeated mild fires that retard competing conifers, while colonizing seedlings of both redwood and sequoia thus gain a competitive edge by sprouting and then resprouting following the subsequent fires. Increased fire intensity and frequency resulting from a rapidly warming climate may facilitate the recolonization of redwood into coastal Oregon, and the recolonization and new colonization of sequoia not only onto additional sites in the central and southern Sierra, but even into the northern Sierra and southern Cascades.

Both redwood and giant sequoia have migrated great distances in the past, and fires have probably been important facilitators of those migrations. A new natural colonization would not happen after every intense stand-replacing fire near an established population, because seeds would have to blow in, or maybe be brought in cones by such animals as squirrels, followed by favorable weather for their successful germination and establishment. Germinating seedlings of both redwood and sequoia are unusually susceptible to damping-off fungi, which are common in many soils and are killed by hot fires. Giant sequoias, in particular, retain many years' production of seeds in closed cones, which open and massively release seeds following hot fires.

It is pretty clear that natural migration by colonization of new sites is a hit-or-miss process that operated over long periods of time. Redwoods and giant sequoias have been able to thus far survive several events or conditions that led to the extinction of many other species. They have repeatedly migrated when necessary to places where they then continued to thrive. Very recently, they produced forests that inspire pleasure and awe in the humans that visit them.

Native Americans have lived in or near redwoods and sequoias for over 10,000 years, and some of them have done a pretty good job of managing the native groves with frequent burning. But there are now (mostly European-origin) American humans in the picture. Some of them create new problems, as important examples: by converting (particularly redwood) forests to such things as vineyards and/or permanent structures; by emitting greenhouse gases that rapidly warm climate; and by forest practices that favor shade-tolerant species that then outcompete and thus replace redwoods and sequoias. But some people in that high-impact invasive population of (particularly but hardly exclusively American) humans are concerned about the future of redwoods and sequoias, and are or could be doing something about it. Knowing what seems to impede their natural colonization and range extension helps some of the current humans who care about them help them continue on Earth.

There is no doubt that humans can successfully plant and husband redwoods and sequoias outside of their current native ranges. Some, most notably Sierra Pacific Industries with sequoia and, more modestly, Archangel Ancient Tree Archive with redwood, have recently been doing that with samples of sequoia and redwood from known origins of both single and multiple native-populations. They and others have the stated intention of providing and then planting new locations for redwoods and sequoias to grow and thrive, and doing so in time scales of decades rather than the centuries or millennia it historically has taken these two species to migrate long distances naturally. We have been calling such dedicated planting programs 'assisted migration', and even 'assisted colonization' when the trees successfully reproduce and a population naturalizes on and near the planted site.

It seems that it may take unacceptably long times for redwood and giant sequoia to naturally migrate to safer sites in response to unusually-rapid climate change and other changing environmental stresses, even if a warming climate results in more fires that facilitate their migration. However, helpful humans could and are successfully assisting in their migration and sometimes colonization, including locations far outside of their current natural distributions. So why do we need to be concerned about the natural ability of redwood and giant sequoia to migrate?

One answer is that we cannot be sure that active planting of these species will always be done in the future. It is conceivable that, following some catastrophic disaster, few or no surviving helpful humans will be available to continue planting forests. The effects of such a catastrophe may last for centuries or even millennia before the survivors reorganize and again establish the social and technical ability to plant and husband redwood and sequoia. Today, though, some humans have the knowledge and ability to expand these species' distributions, to thus add to their natural migration and better ensure their survival in the uncertain future.

How might entire regional forests be destroyed, or altruistic forest management be abandoned, either regionally or worldwide? Since the 1945 nuclear bombing of Hiroshima, apocalyptic worriers have accumulated some pretty realistic scenarios. We've learned that a collision with an asteroid has caused widespread loss of species and might do so again. And here are two (among several) examples of possible new self-inflicted catastrophes with contrasting implications for the future of redwoods and sequoias.

A massive use of nuclear weapons between or among the current nuclear powers may occur. Such madness will likely kill most or all humans and other living things in targeted regions, including redwoods and sequoias. The current distribution of nations with nuclear capability makes it likely that such madness will mostly affect the Northern Hemisphere, and people and forests in the Southern Hemisphere will survive. It may then take many decades or even centuries before people can again safely inhabit the northern half of Earth. There is already a magnificent 106-year-old grove of redwoods in Rotorua, New Zealand, a somewhat younger but faster-growing redwood grove near Taumarunui, New Zealand, other such planted groves might be found and dedicated, and assisted migration could establish additional groves of redwoods and sequoias in Chile, Pategonia, Australia, New Zealand and South Africa in advance of such a hemispheric extirpation.

A historical example is the 14th through 18th century black plague pandemics, which not only killed a lot of people, but also disrupted the social, political and commercial structures of nations and regions. The black plague is even credited with saving European forests. Many 14th century European forests were being converted to other uses by rapidly growing human populations and resulting commercial exploitation. The plague pandemics greatly reduced those human populations and their needs for agricultural land and wood, and forests then reclaimed much of the land. An engineered weaponized pathogen, if it is released or escapes, would likely be more efficient than the black plague was in quickly spreading and then killing humans. But, like the black plague, it would probably leave most of Earth's biota essentially intact, and perhaps even better off. And, as in most pandemics, a few humans might be resistant or escape the disease, begin to repopulate Earth, and their progeny would eventually again visit and appreciate groves of enormous redwoods and sequoias. In this scenario, conserving and perhaps expanding the redwood and sequoia populations in North America, plus some additional groves in Europe and Asia, would have been good enough.

Such an apocalyptic catastrophe may not occur for a long time, or at all. Meanwhile, in the near future, Earth's human population will continue to increase, as will population-related problems and stresses. It is noteworthy that the United Nations held one of its founding ceremonies in a Muir Woods redwood grove, probably because humans find not only pleasure and awe in such groves, but many also gain perspectives on time and feelings of peacefulness and well-being. Additional such magnificent long-lived groves of redwood and sequoia in many places on Earth could serve its hopeful future in possibly important ways.

There are at least two options for new human-assisted groves. One is to sample and thus nearly duplicate only one redwood population or sequoia grove per new planting, thus conserving the genetic structures of the different native populations and groves. A second is to combine samples of many populations or groves per new planting, thus increasing the genetic variation in the new plantations and thereby increasing their ability to better adapt to different environments.

There are many locations far outside of their native ranges where redwood and/or giant sequoia could thrive and grow to become magnificent groves. Why they are not now on such sites has until recently been because they could not get there naturally. But now that assisted migration is technically possible, human motivation and competing demands on those sites will guide the future. If enough new groves are established, it is likely that some will be in the right places to thrive and reproduce even in substantially changing conditions. Such assisted colonization seems like a good thing to do, whether or not humans survive (or other sentient creatures evolve) to appreciate these two magnificent species.

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Restoration Management in Redwood Forests Degraded by Sudden Oak Death[1]

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Abstract

We describe the foundation, objectives, and initial results from a stand-level experiment focused on restoration of redwood (*Sequoia sempervirens* (D. Don) Endl.) forests impacted by sudden oak death (SOD), caused by *Phytophthora ramorum*. Our study stands were primed for heavy impacts by SOD. Extensive harvesting which ended circa 1910 on Mt Tamalpais (Marin County, California) resulted in high densities of tanoak trees with interspersed residual redwood. The arrival of *P. ramorum* and subsequent emergence of SOD transformed these stands into tanoak shrublands with interspersed redwood trees. Pretreatment understory tanoak densities were extremely high relative to redwood forests of the north coast which have not been invaded by *P. ramorum* but both redwood advanced regeneration and overstory tree densities were low in the same respects. Mastication and hand-crew piling treatments were applied in 2015 on a randomly selected group of plots and each treatment type substantially reduced tanoak densities suggesting redwood establishment may now be possible. Our study is designed to assess tradeoffs of treatment costs with benefits resulting from fuels reduction, redwood regeneration, carbon sequestration, and water provisioning. We cannot yet make strong conclusions about these tradeoffs given the preliminary nature of our datasets. Instead, we describe areas of uncertainty and identify critical questions that must be evaluated to understand the utility and appropriateness of applying these treatments across a broader portion of the redwood forest landscape.

Introduction

Sudden oak death (SOD), caused by *Phytophthora ramorum*, has represented a significant threat to redwood (*Sequoia sempervirens* (D. Don) Endl.) forests since the emergence of the disease circa 1996. Although the causal pathogen, *Phytophthora ramorum*, can infect redwood these infections do not represent a significant threat to the health of individual redwood trees. However, mortality of redwood forest species and the resulting impacts substantial changes at the entire redwood ecosystem level; these impacts include increased ground fuels, dense resprouting of the most susceptible individual redwood trees and reduced ecosystem services provided by these forests (Cobb et al. 2012a, Metz et al. 2013). Tanoak (*Notholithocarpus densiflorus* (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh) is a unique component of redwood ecosystems; it is the dominant nut-producing species in redwood forests and is often the sole ectomycorrhizal species when Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) is not present in the stand. Tanoak is also notable for the potent combination of supporting high pathogen sporulation rates from infected tissue as well as the development of lethal bole infections that kill the above ground biomass. Thus, tanoak can both transmit the pathogen and develops the disease SOD which causes extensive mortality, increases fuel loads, and changes stand-level carbon cycling (Cobb et al. 2012a, 2016). The high densities and basal area of tanoak in many redwood stands means that many redwood forests have been impacted by this disease and many others are threatened in the coming decades (Cobb et al. 2013b, Cunniffe et al. 2016).

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Phytophthora ramorum has invaded throughout the greater San Francisco Bay Area, and damaged the culturally, ecologically, and economically important forests of Mount Tamalpais (Marin County) including many stands managed by the Marin Municipal Water District (MMWD) for water yield and recreation. Sustained inoculum loads have resulted in extensive tanoak mortality. In some stands almost 100 percent of initial overstory tanoak trees have been killed by the disease. Tanoak resprouting has formed undesirable forest structure where occasional redwood overstory trees co-occur with dense, tanoak shrub understories. Some of the most impacted redwood forests in California occur on Mount Tamalpais; these stands have significantly higher understory tanoak densities in comparison to a survey of 500 m^2 plots (172) located in uninvaded redwood forests from Sonoma to Del Norte counties (fig. 1; Cobb et al. 2012b, Metz et al. 2012). Dense understory conditions are significant management concerns from the perspectives of fuel loads, maintenance of biodiversity, aesthetics and provisioning of water resources. These concerns motivated an effort to design and test potential ecosystem restoration techniques for redwood forests impacted by this disease.

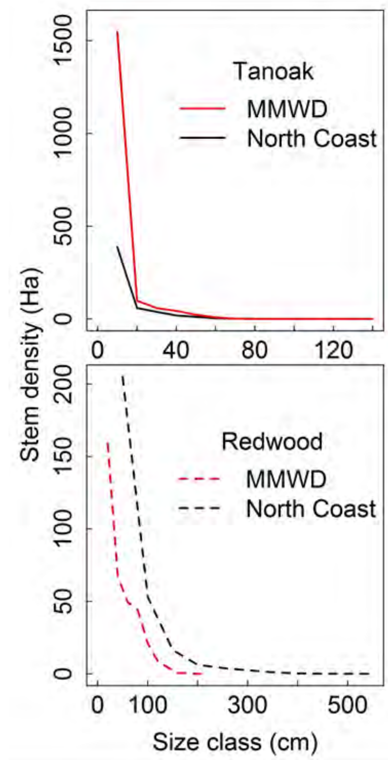

Figure 1. Tanoak (upper panel) and redwood (lower panel) stem densities in pre-treatment restoration experimental study plots (MMWD) compared to a survey of 172 - 500 m^2 long term study plots located in stands uninvaded by *Phytophthora ramorum*.

We instituted a series of replicated management experiments on MMWD lands to identify the most economically and ecologically effective actions to restore overstory trees and key ecological functions of our study area including carbon sequestration and water provisioning. Our goals were to understand if different treatment approaches have comparable benefits in terms of disease suppression and how these effects may augment or offset gains in ecosystem processes. We also aimed to understand what treatments are most effective in increasing the dominance of redwood in a set of stands where historical harvesting resulted in high densities of tanoak relative to redwood other regional forests (fig. 1). SOD kills the above ground portion of tanoak but basal sprouting can be extensive. Our restoration study sites had extremely high densities of small (~1cm diameter at breast height;1.37 m, DBH) tanoak stems with average values greater than 1500 stems ha^{-1} (fig. 1,upper panel). In contrast, North Coast redwood stands had significantly lower mean tanoak densities for the same size classes. Although redwood can dominate the overstory of upland redwood forests of our study area, the relatively low density of redwood in larger diameter classes compared to North Coast stands indicates that redwood has not regained pre-harvest dominance in the study site since extensive harvesting ended circa 1900. SOD could delay natural (e.g., without silvicultural intervention) succession at this and similar study sites if tanoak resprouting also inhibits redwood regeneration in the understory. Lower densities of redwood in small diameter classes at our study site compared to North Coast redwood forests suggests the disease may lock stands into an undesirable condition dominated by small diameter tanoak. This condition is akin to a tanoak shrubland with occasional overstory redwood trees but little or no redwood regeneration (fig. 2). Our study uses a suite of carbon cycling measurements including litterfall, soil C stocks, soil C dynamics (soil respiration, methane, and N_2O flux) to understand the extent of these restoration treatments in reducing greenhouse gas emissions, an important state-level policy goal. Therefore, our study is designed to determine the most cost-effective treatments for a set of management goals broadly applicable across the range of redwood forests at risk from SOD. In addition, the study design, both in terms of treatments and measurements, was constructed to identify potentially conflicting treatment outcomes, such as reduced water provisioning in stands with greater above ground carbon storage.

Experimental Design and Measurements

Figure 2. Tanoak shrubland-like stand conditions with interspersed overstory trees, mostly redwood. These high density tanoak stand conditions are broadly representative of pre-treatment stand structure in upland redwood forests of Mt Tamalpais. Fuels mastication treatments are shown in the foreground.

We established 30, 0.405 ha (1 ac) treatment plots across three MMWD sites and randomly assigned treatments within blocks of five plots. Randomly applied treatments included reference (no treatment), understory mastication (up to ~10 cm DBH), hand crew thinning and piling of residual materials, hand thinning with burning of piles, and mastication with follow-up removal of resprouting tanoak. Mastication was conducted using a combination of an excavator with a masticating head and/or a skid-steer with a forestry attachment (masticator head). Hand crews sought to apply similar levels of forest thinning and we found no significant differences between post-treatment stem densities for piling stem densities were 260 stems ha^{-1} ± 220 (Interquartile range - IQR) vs masticating 425 stems ha^{-1} ± 315 (IQR). The principle difference between treatments is that residual materials were concentrated in piles whereas mastication treatments conducted with heavy machinery were designed and applied with the goal of uniform distribution of residual materials. Treatments were designed and applied to remove all hardwoods while retaining conifer (redwood, Douglas-fir) regeneration. These efforts were generally successful for individual trees greater than ~1m in height.

Each plot was instrumented to measure water outflow by placing two soil moisture sensors at the parent material – mineral soil interface (between 70 and 100 cm) and at 30 cm depth. Soil moisture was measured every half hour and monitored continually with a cellular data uplink. Precipitation gauges were established in the reference treatments in four of five study blocks at the drip line of an individual redwood tree within the respective plot. We installed 15 cm length PVC collars in the shallow mineral soil (to ~12 cm depth) to conduct repeated measurements of greenhouse gas dynamics on a monthly basis. We attached portable air-tight chamber tops to each of these PVC collars for a 15 min period and withdrew soil gas samples drawn every 5 min to calculate rates of efflux. We also placed two plastic bins within each plot to quantify treatment effects on foliar litterfall, an important component of above ground productivity which can change rapidly with changes in growth and year-to-year climate variations (Cobb et al. 2013a). Lastly, we conducted quarterly samplings of shallow mineral soil (~15cm surface soil) to determine treatment effects on fine root dynamics, also an important component of ecosystem productivity sensitive to management and disturbance (Kaye et al. 2005). Roots were gently washed from soil in a low-pressure cold-water stream and sorted into live and dead fine root pools (all species between 2-0.5 mm diameter).

Results and Discussion

Mastication treatments greatly reduced fuel loads, understory density, and prevalence of sporulation-supporting species; these treatments reduced densities from 3475 stems ha^{-1} (± 2045 IQR) to 425 stems ha^{-1}. Hand crew piling was also similarly effective in reducing understory tree density and density of species which transmit *P. ramorum*; piling reduced stem densities from 1675 stems ha^{-1} (± 1260 IQR) to 260 stems ha^{-1}. The large interquartile range values (IQR), a measure of data spread, indicate the pretreatment variation in stem density across sites and also indicate that plots selected (randomly) for hand crew treatments had somewhat lower pre-treatment stem densities. Using a set of models parameterized with data from this experiment, we found that 90 percent of intact tanoak overstory trees are expected to be retained by the treatment, in part because these individual trees will be isolated from inoculum sources. Although SOD has been devastating to overstory tanoak individuals, residual tanoak trees are still present throughout the experimental area albeit at much lower densities compared to pre-treatment levels. A body of previous monitoring and modeling work suggests these individual trees will survive for much longer periods relative to untreated (high tanoak density) areas due to reduced inoculum pressure on residual trees (Cobb et al. 2012b, 2013b, Valachovic et al. 2013b). These disease suppression effects are likely to realize their greatest benefits in treatments that maintain low tanoak densities (such as follow-up sprout removal). This expectation follows from field experiments showing sprout removal provides pathogen suppression but only on the order of 3 to 5 years without maintenance of low-density conditions (Valachovic et al. 2013a).

Each understory removal treatment was effective in reducing tanoak density regardless of the treatment applied (mastication vs. piling, etc). However, the cost efficiency of applying one method over another has not yet been evaluated and is a subject of ongoing analysis. Each treatment type was effective in reducing understory dead fuels apart from unburned piles and also greatly compacted fuel beds (reduced fuel heights). Mastication treatments also tended to increase forest floor depth and mass, an effect of deliberately redistributing understory plant materials. The same effect was found for fine ground fuels, each treatment had the effect of redistributing understory biomass by transforming relatively large fuels (100 hr and above) into woody material that is generally 5 to 15 cm in length and ~2 cm diameter. Hand-crew pile treatments where piles were not burned are the sole exception among these treatments. Unfortunately, rather little is known about masticated fuel beds in terms of their effects on flame lengths and burn time although the depth and density of masticated fuel beds is likely an important driver of these fire behaviors (Kreye et al. 2014). In California, similar mastication treatments have been conducted to reduce shrub (primarily *Arctostaphylos* and *Ceanothus* species) and tanoak (Kane et al. 2009). As noted in these previous field studies and synthesis, masticated fuel beds have unique fuel composition and density for which current fire models do not adequately integrate. Although the reduced particle size and increased surface area of fuel particles in our mastication treatments could increase fuel decomposition rates, this has not been quantified in our treatments and the current state of research on masticated fuel beds does not support much more than speculation about decomposition rates (Kreye et al. 2014).

Initial data indicate these restoration treatments have also impacted the ecosystem processes we aimed to assess. Across treatments, soil moisture rapidly increased to field capacity during the onset of winter rains and all treatments showed outflow to deep soil layers during particularly heavy precipitation events. However, thinning treatments (all types) increase soil moisture of both shallow and deep soil layers, a common pattern of soil water dynamics following management or disturbances which reduce above ground transpiration. Our ongoing work aims to quantify potential differences in outflow and any changes in the proportion of water reaching the soil surface and outflow among treatments. From a longer-term and broader perspective, our study is structured to inform mastication treatment effects on water quality and quantity at the watershed-scale. This information is particularly important for the MMWD, a municipal water district that provides drinking water for approximately 187,000 people. SOD and the management responses to the disease has or will in the near future impact watersheds for thousands of others in coastal California as well as numerous fish-bearing streams in the region. Additional measurements of ecosystem dynamics (litterfall and soil GHG dynamics) were begun in summer 2016 and we have insufficient data to make robust assessments of our study aims. However, we have found a consistent net consumption of methane in our redwood soils in July and August 2016. While the timing of these observations corresponds to low soil moisture levels which generally favor methane diffusion into soils and consumption of this important GHG, methane consumption has not been documented in redwood forests; this argues for continued monitoring to determine the relevance of these observations to greater GHG policy goals.

Long-term efficacy of our treatments in restoring carbon sequestration and sustaining water yield will almost certainly depend on the reestablishment of overstory redwood given that carbon sequestration in this species is unlikely to be impacted when ground fuels can be reduced (Metz et al. 2013). It must be noted, that this reestablishment of redwood is unlikely to occur in stands similar to our study sites whether they have been impacted by disease or not. Harvesting in our study sites ended over a century ago and prior to disease these stands were primarily tanoak overstories with scattered redwood overstory trees and little advanced redwood regeneration. Post disease, the high densities of tanoak understories also lacked advanced redwood regeneration (fig. 2) indicating these conditions are unlikely to change without intervention that increases redwood regeneration. It is well known that removal of tanoak competitors is needed to increase conifer growth and dominance in the overstory (Harrington and Tappeiner 2009) and this same lesson can be applied to SOD-impacted redwood forests. Of course, in order for these treatments to be applied more broadly, cost effectiveness and tradeoffs among the various benefits (fuels, carbon sequestration, etc.) will also likely determine the optimal treatment for a particular landowner and disease condition. Restoring redwood forests impacted by SOD will clearly require a long-term adaptive management commitment.

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Redwood Seedling Responses to Light Patterns and Intensities[1]

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Abstract

Coast redwood (*Sequoia sempervirens* (D. Don) Endl.) seedlings were grown from seed in controlled environments with 16 hour photoperiods using three light patterns that mimicked full shade (constant light level), intermittent high light such as long duration sun flecks (low light with 15 minutes of intense light every 2 hours), and large openings (4 hours low light, 8 hours high light, 4 hours low light). Each light pattern contained three light intensity levels (33 percent, 66 percent, and 100 percent) with one intensity level in each pattern that provided 5.1 daily light integral (DLI). Among the treatments, the extremes of daily light were 1.6 DLI to 15.5 DLI.

Seedlings increased biomass accumulation with increasing light level with diminishing accumulation at the higher DLIs. The seedlings were most efficient at utilizing light in the full shade and large opening patterns with poor utilization of light in the intermittent sun fleck pattern. Within each pattern, increased light intensity resulted in increased seedling height, stem diameter, branching, branch length, leaf area, specific leaf area, leaf mass, root mass, total mass, root/shoot ratio, stomatal density and needle thickness. In general, maximum net photosynthesis increased with increased light intensity. Quantum efficiency did not vary with intensity within the full shade and large opening patterns. Within the intermittent sun fleck pattern, quantum efficiency was lower in the seedlings grown at the lowest light intensity.

With equal daily light, seedlings in all three patterns had similar branch length, total mass, and leaf mass. However, seedlings in the large opening pattern had greater height and greater stem diameter, but less specific leaf area than the full shade pattern. Seedlings in the intermittent sun fleck pattern had less height, stem diameter, total mass, leaf area, and leaf mass, but greater root/shoot ratios and specific leaf areas than seedlings in the other patterns.

Maximum net photosynthesis was greater in the seedlings grown in the large opening pattern. Quantum efficiency was not affected by light pattern. Photosynthetic light compensation points ranged from $12 \mu E m^{-2} s^{-1}$ for seedlings in the low intensity treatment of the shade pattern to $20 \mu E m^{-2} s^{-1}$ for seedlings grown at full intensity in the large opening pattern.

Needle morphology varied markedly among light treatments. Needles in the high light treatment of the large opening pattern had a regular pattern of rounded upper epidermal cells and a deeply safranin stained, elongated palisade layer. Needles in the low light treatments of both the full shade and intermittent sun fleck treatments had prismatic shaped cells in the upper epidermis and irregular, lightly safranin stained palisade layers.

Keywords: anatomy, coast redwood, ecology, growth, light, photosynthesis, *Sequoia sempervirens,* seedling, shade

Introduction

There is a poor understanding of the environmental conditions necessary for redwood (*Sequoia sempervirens* (D. Don) Endl.) seedling establishment. Redwood seedlings are found at the edge of redwood stands and on disturbed sites adjacent to redwood forests (Jacobs 1987) and the establishment of redwood seedlings has been found to be positively correlated with soil disturbance

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with seedling growth greater in open sites (Woodward 1986). The factors thought to be most detrimental to the establishment of redwood seedlings in the understory are: moisture stress, soil-borne fungi, and light (Becking 1996, Jacobs 1987, Muelder and Hanson 1961, Olson et al. 1990). These factors are inter-related; increased light was found to assist redwood seedlings in resisting both damping-off fungi and water stress (Jacobs 1987). Investigations of light conditions at redwood forest floors are somewhat limited, but they indicate that light is generally low and variable (Jacobs 1987, Pfitsch and Pearcy 1989, Powles and Björkman 1981, Waring and Major 1964, Woodward 1986). Sunflecks are present, even under a dense canopy, and may contribute a large portion of the integrated daily light available under the canopy, particularly the sunflecks of longer duration. Several studies have demonstrated that redwood seedlings can exist in low light, at least under experimental conditions (Baker 1945, Bates and Roeser 1928,). In natural conditions, shade has been described as beneficial to redwood seedling establishment provided root rot is excluded and other factors are held equal (Muelder and Hansen 1961). Shade was also found to be beneficial to redwood seedlings planted into open areas, although it was thought that the benefit was derived primarily through improved soil moisture (Fritz and Rydelius 1966). Additionally, it has been claimed that redwood seedlings can endure and grow slowly in heavy shade, but that their best juvenile growth is under full sunlight (Olson et al. 1990). More recent studies of redwood seedlings and light have demonstrated increased water potential and photosynthesis with fog exposure (Simonin et al. 2009) and increased light use and carbon gain from sun fleck use with increased moisture inputs (Santiago and Dawson 2014).

The purpose of this study was to investigate the response of redwood seedlings in a controlled environment varying only light. The primary objectives of this experiment were: 1) To determine the effect of light patterns on redwood growth, 2) to determine the effect of light intensity on redwood growth, and 3) to determine the plasticity of morphological and physiological characteristics of redwood seedlings to differing light patterns and intensities that mimic both openings and the redwood forest understory.

Methods

Three patterns of light were used with three levels of light intensity within each pattern for nine light treatments (table 1) within controlled environment growth chambers. Light intensity and integrated daily light treatments were selected based on reports of light levels in redwood forests in the studies cited above, as well as constraints imposed by the characteristics of the chambers available (Western Environmental, Inc. Napa, California, Model E-78HL). Intensity levels were obtained by partitioning the chambers using black shade cloth drapes (33 percent and 66 percent light transmittance, Stuewe and Sons, Corvallis, Oregon) as neutral density filters fitted above and around the open sides of a treatment area on the platforms holding the seedlings.

Five seed sources from Del Norte and Humboldt counties were used to allow for a gradient in both latitude and elevation. Eight seedlings from each of the five seed sources were used in each of the nine light treatments for a total of 360 seedlings. To allow for the observance of plasticity and avoid the confounding factor of observing and correcting for acclimation, germinated seeds with a radicle less than 2 cm were placed in 3 liter pots (10 X 10 cm top, 7.5 X 7.5 cm bottom, 30 cm height). The pots were filled with a 1:1 peat/perlite mix. Replacement planting was carried out until all treatments had a full complement of seedlings that reached 40 days in age. Temperature, moisture, and nutrient availability were held as constant as possible. Air temperature was held at 25 °C during periods of light and 15 °C during darkness. Seedlings were watered to field capacity daily and humidity was not controlled. Seedlings were fertilized three times with an application of 4 ml of Osmocote® 17-6-10 Plus Minors (Grace Sierra Horticultural Products Co., Milpitas, California) applied at the emergence of each seedling's cotyledons and at 40 and 80 days after the initial application.

Table 1—Experimental design, Light patterns and intensity treatments. All treatments had a 16-hour photoperiod with 8 hours of darkness between photoperiods. Values given as PPFD (μEm^{-2} s^{-1}) or DLI (Em^{-2} day^{-1}). Note that within each Light Pattern one treatment had an integrated daily light level of 5.1 DLI

	Light intensity treatment		
	33%	66%	100%
Shade Pattern: constant light			
PPFD	29	57	87
DLI	1.6	3.3	**5.1**
Sun Fleck Pattern: low light interrupted by 15 min high every two hrs			
Low light PPFD	20	40	60
High light PPFD	275	550	825
DLI	2.7	**5.1**	8.2
Large Opening Pattern: 4 hrs low light, 8 hrs high light, 4 hrs low light			
Low light PPFD	12	25	38
High light PPFD	165	330	500
DLI	**5.1**	10.2	15.5

Harvest of seedlings was delayed until most seedlings were large enough to have amble foliage for measurement of photosynthesis in a 5 liter cuvette. However, there was a disparity between the times allowed for growth of seedlings in different light treatments (table 2). At harvest, the largest seedlings were too tall to control their light levels within the growth chambers and to use the photosynthesis cuvette while the smallest seedlings were too small for the measurement of net photosynthesis. Proceeding with the harvest of the largest seedlings before those of smaller seedlings introduced a bias in the sampling.

Net photosynthesis was measured at harvest as the difference between CO_2 concentration in air entering the cuvette and that of air passed through the cuvette with the use of a Non-Dispersive Infrared Analyzer (Model 865, Beckman Industrial Corp., Fullerton, California). Light levels used for measuring, in order of measurement, were 0 (respiration) 11, 33, 100, 300, 600, and 1000 photosynthetic photon flux density (PPFD). Leaf characteristics were measured on 10 adjacent needles from mid-length of a branch that was at the mid-length of the seedling. One of these needles was randomly selected for the measure of stomatal density. Three needles for microscopic examination were taken from the mid-length of another branch opposite of the branch selected for needle characteristics. Height and stem diameter at the root collar were measured and the seedling partitioned into root, shoot, and leaf. Leaf area was measured with a Delta-T area measuring device (Delta-T Devices, Cambridge, England) and all parts dried at 80 °C for 12 hours.

Analysis of variance was performed using BMDP (BMDP Statistical Software, Inc., Los Angeles, California). Program 7D was used for two-way analysis and their associated contrasts. Contrasts were weighted by sample size. To control type I errors, contrasts were limited by using linear contrasts only when a significant difference in the main effect was detected in the analysis of variance. Program 2V was used for the three-way analysis of variance of photosynthesis. Schéffé's procedure for the unplanned contrast of the time allowed for seedling growth was performed using NCSS (5.X series 1992 J.L. Hintze, Kaysville, Utah). Fisher's least significant difference was used for the planned comparisons of net photosynthesis. Regression of photosynthesis data to determine quantum efficiency was performed using BMDP program 1R.

Table 2—Mean values for variables and sample size. (Asterisk* = a significant contrast ($p < 0.05$) between 33% and 100% intensity within a light pattern; within equal light treatments, values that are significantly different ($p < 0.05$) are followed by a different letter)

		Shade			Sun fleck			Large opening			Equal light treatments		
Time	**Variable**	**33%**	**66%**	**100%**	**33%**	**66%**	**100%**	**33%**	**66%**	**100%**	**Shade**	**Sun fleck**	**Large opening**
40 days	Height (cm)	2.8*	3.6	3.7	3.1*	3.1	4	3.8*	3.3	3.1	3.7^{a}	3.1^{b}	3.8^{a}
80 days	Height (cm)	6.4*	9.3	10	7.2*	8.2	11	11.1	11.3	11	10.0^{a}	8.2^{b}	11.1^{a}
	Stem diameter (mm)	0.8*	1.2	1.5	0.8*	1	1.3	1.4*	1.8	2	1.5^{a}	1.0^{b}	1.4^{a}
	Branches/cm stem	0.12*	0.37	0.59	0.05*	0.21	0.4	0.58*	0.75	0.84	0.59^{a}	0.21^{b}	0.58^{a}
	Longest branch (cm)	3.5*	6.3	8.4	2.3*	4.3	6.4	8.1*	8.9	9.6	8.4^{a}	4.3^{b}	8.1^{a}
120 days	Height (cm)	10.9*	21.2	22.3	12.7*	15.6	24.5	26.3*	31.8	33.9	22.3^{a}	15.6^{b}	26.3^{c}
	Stem diameter (mm)	1.3*	2.1	2.5	1.3*	1.8	2.4	2.7*	3.5	3.9	2.5^{a}	1.8^{b}	2.7^{a}
	Branches/cm stem	0.35*	0.57	0.68	0.35*	0.6	0.61	0.59	0.58	0.66	0.68^{a}	0.60^{ab}	0.59^{b}
	Longest branch (cm)	7.8*	13.9	16	7.1*	10.3	13.8	15.1*	18	19.9	16.0^{a}	10.3^{b}	15.1^{a}
Harvest	Sample size	28	37	37	35	34	35	35	34	35	37	34	35
	Time (days)	270*	237	223	283*	241	220	220	209	213	223^{a}	241^{b}	220^{a}
	Height (cm)	28.0*	45.9	49.6	29.3*	40.8	51.2	56.9	64	62.6	49.6^{a}	40.8^{b}	56.9^{c}
	Stem diameter (mm)	2.6*	4	4.3	2.5*	3.7	4.4	4.8*	6.1	6.1	4.3^{a}	3.7^{b}	4.8^{c}
	Leaf area 1000 cm^2	0.22*	0.58	0.79	0.21*	0.53	0.64	0.78*	1.1	1.15	0.79^{a}	0.53^{b}	078^{a}
	Leaf mass (g)	0.82*	2.51	3.74	0.75*	2.33	3.11	4.01*	7.45	7.73	3.74^{a}	2.33^{b}	4.01^{a}
	Specific leaf area $cm^2 g^{-1}$	268*	230	219	270*	236	210	201*	154	155	219^{a}	236^{b}	201^{c}
	Mass (g)	1.35*	4.24	6.04	1.19*	3.84	5.56	6.87*	13.32	13.39	6.04^{a}	3.84^{b}	6.87^{a}
	Root mass (g)	0.17*	0.61	0.89	0.14*	0.68	1.08	1.07*	2.78	2.81	0.89^{a}	0.68^{a}	1.07^{a}
	Root/Shoot	0.14*	0.16	0.17	0.13*	0.21	0.23	0.18*	0.24	0.25	0.17^{a}	0.21^{b}	0.18^{a}
	Stomatal density	2.58*	3.05	3.22	2.61*	2.95	3.19	3.43*	3.67	4.09	3.22^{a}	2.95^{b}	3.43^{c}
	Needle length (mm)	20	23	21.9	17.9*	22.9	22.8	24.5	24.2	23.3	21.9^{a}	22.9^{b}	24.46^{c}
	Needle width (mm)	1.9*	2.3	2.4	1.9*	2.2	2.2	2.2	2.3	2.2	2.35^{a}	2.15^{b}	2.2^{ab}
	Needle thickness (mm)	0.16*	0.21	0.24	0.16*	0.21	0.25	0.22*	0.28	0.29	0.24^{a}	0.21^{a}	0.22^{a}
	Needle width/thickness	12.5*	11.2	10.3	11.9*	10.6	8.9	10.5*	8.4	8.3	10.3^{a}	10.3^{a}	10.5^{a}

Results and Discussion

Seedling height, stem diameter, and biomass increased with greater light intensity within all light patterns (table 2). The differences in growth were striking (fig. 1) with higher light intensity promoting growth in nearly all treatments. The results may have been even more distinct had it been possible to harvest the seedlings simultaneously. The exception was the seedlings of the 66 percent and 100 percent intensity treatments of the Large Opening Pattern which had similar growth. The heights of seedlings grown in these two light intensity treatments compared closely with the mean height of 53.6 cm at 168 days for seedlings grown in somewhat similar conditions (Hellmers 1966).

Figure 1—Representative seedlings from each treatment at 192 to 198 days old. Treatments and DLI from left to right : Shade 33 percent (1.6) , Sun Fleck 33 percent (2.7), Shade 66 percent (3.3), Shade 100 percent (5.1), Sun Fleck 66 percent (5.1), Large Opening 33 percent (5.1), Sun Fleck 100 percent (8.2), Large Opening 66 percent (10.2), and Large opening 100 percent (15.5). Bar in background is 25 cm in length.

Higher light intensity, regardless of pattern, resulted in greater total seedling mass, root mass, leaf mass, and increased leaf area (table 2). The increase in root mass with increased light was proportionally greater than the increase in shoot mass and resulted in increased root/shoot ratios in each light pattern. Additionally, increased light resulted in a proportionally greater increase leaf mass than leaf area and a subsequent lower specific leaf area (cm^2 g^{-1} leaf tissue) in each pattern. This decrease in specific leaf area was consistent with the response in other conifers (Del Rio and Berg 1970, Klinka et al. 1992, Lassoie et al. 1985, Tucker and Emmingham 1977). Increased light increases thickening and structural changes, such as increased stomatal density, in needles while the seedling total leaf area also increased (table 2). The increase in total leaf area of seedlings exposed to increased light may have had a high cost in carbon consumption; however, these same seedlings were also able to place a higher proportion of their biomass into the development of roots and conductive tissue as measured by stem diameter (table 2). Conversely, seedlings from the 33 percent intensity treatments of the Sun Fleck and Shade Patterns did not construct as extensive root systems and conductive tissues as seedlings in higher light treatments. Seedlings grown in lower light prioritized leaf area rather than root or stem diameter growth which is consistent with observations of Baker (1945). A decline in root development in low light is probably a consistent pattern in conifers as shading is also known to increase carbon allocation to the shoot in *Psuedotsuga menziesii* (Mirb.) Franco (Brix 1967, Drew and Ferrell 1977, Krueger and Ruth 1969) and *Picea sitchensis* (Bongard) Carrière and *Tsuga heterophylla* (Raf.) Sarg. (Krueger and Ruth 1969).

The limit of the positive effect of light on seedling growth may have been approached in the 66 percent and 100 percent intensity treatments of the Large Opening Pattern. Height at 40 days was less in the 100 percent intensity of light intensity on seedling growth peaked near the intensities provided by the 66 percent and 100 percent intensity treatments. Conversely, the lower light limit for redwood seedlings was likely approached in the 33 percent intensity treatment of the Shade Pattern. In addition to the poor growth within this treatment, there was greater mortality. Twelve seedlings died after the 40 days within this treatment (table 2) and all displayed symptoms of infection by damping off fungi.

Within each light pattern, higher light intensity resulted in greater maximum photosynthesis (figs. 2A, 3A, and 4A) and greater respiration (figs. 2B, 3B, and 4B). The exception was the response curves for seedlings in the 66 percent and 100 percent Large Opening Pattern (figs. 4A and 4B) which were similar in respiration and maximum photosynthesis. Photosynthetic light compensation points ranged from 12 PPFD for seedlings in the low intensity treatment of the shade pattern to 20 PPFD for seedlings grown at full intensity in the large opening pattern (figs. 2B, 3B, and 4B). In the Sun Fleck Pattern, quantum efficiency was significantly lower in seedlings grown in the 33 percent intensity treatment (fig. 4B) but in all the other light patterns intensity treatments did not affect quantum efficiency.

Comparing responses within the Equal Light Treatments, seedlings in the Sun Fleck Pattern were smaller than those in the Shade or Large Opening Patterns (table 2). Seedlings in the Shade and Large Opening Patterns were similar in size at 40 and 80 days, but seedlings in the Large Opening Pattern had greater height at 120 days and harvest and larger stem diameter at harvest (table 2). The specific leaf area of the seedlings differed among the three patterns with the highest specific area in the Sun Fleck Pattern and the lowest in the Large Opening Pattern (table 2). Specific leaf area is known to decrease from low light to high light in conifers (Del Rio and Berg 1970, Klinka et al. 1992, Lassoie et al. 1985, Tucker and Emmingham 1977, Tucker et al. 1987). It appears that, for redwood seedlings, specific leaf area is affected by both light intensity and pattern and is not controlled simply by integrated light, or by the highest instantaneous light level to which seedlings were exposed.

There were no differences in root mass among light patterns in the Equal Light Treatments (table 2); however, the root/shoot ratio was greater in seedlings of the Sun Fleck Pattern. It could be speculated that allocation of carbon to roots in intermittent light is a mechanism that compensates for possible water stress that may occur during sun flecks. An alternate explanation is that root growth is strongly determined by integrated light whereas top growth is affected by light pattern.

Needle morphology varied markedly among light treatments. Needles from seedlings grown in high light treatments had a regular pattern of rounded upper epidermal cells and a deeply safranin-stained, elongated palisade layer (fig. 6) whereas needles from seedlings grown in low light treatments of both the full shade and intermittent sun fleck treatments had prismatic shaped cells in the upper epidermis and irregular, lightly safranin-stained palisade layers (fig. 7). This response is not unlike sun and shade leaves in western hemlock reported by others Tucker and Emmingham (1977).

In summary, both light intensity and light pattern affected the growth and morphological characteristics of redwood seedlings (table 2). The DLI had the greater, but not exclusive, effect on seedling characteristics. In general, the higher the DLI the larger the seedling grew. Given equal DLI, seedlings in the Large Opening and Shade Patterns generally had similar size and characteristics and seedlings in the Sun Fleck Pattern were smaller. The seedlings in the Sun Fleck Pattern were unable to effectively use the periods of high light provided, but this lack of growth may have been the result of the particular intermittent light pattern of light chosen. Other patterns of intermittent light, such as shorter, more frequent sun flecks or longer duration sun flecks, may be utilized by redwood seedlings in a more efficient manner. Other than a lower quantum efficiency of seedlings within the lowest light intensity of the Sun Fleck Pattern and the expected increases in respiration and maximum photosynthesis of seedlings within the high light treatments, photosynthetic characteristics did not radically differ between seedlings grown in different intensities or patterns.

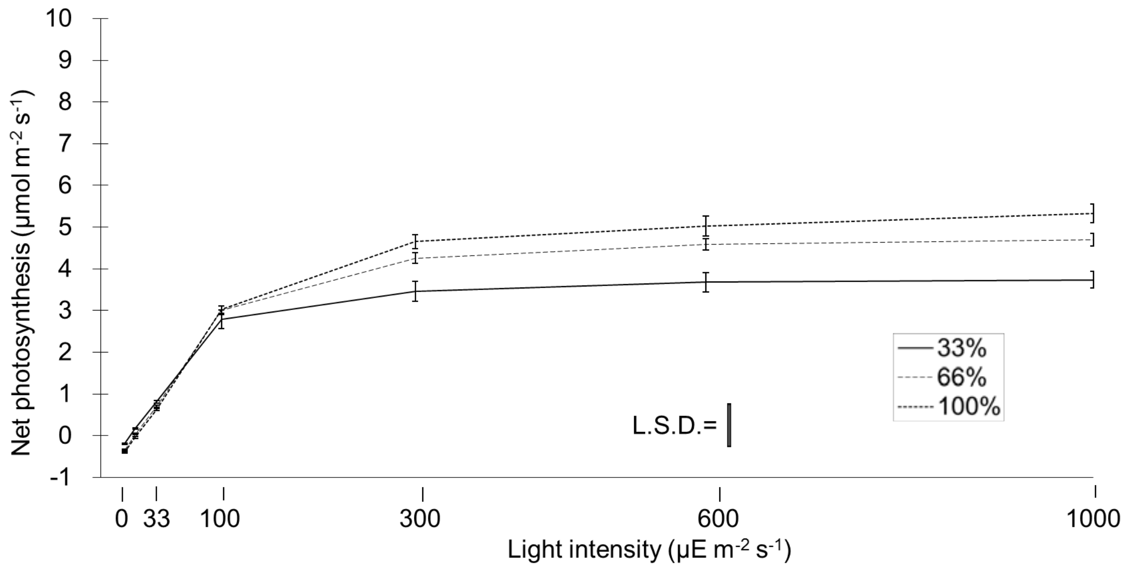

Figure 2A—Steady state net photosynthesis by redwood seedlings grown in the Shade Pattern. Bars indicate standard error of the mean and LSD bar indicates Fisher’s least significant difference among means ($p < 0.05$).

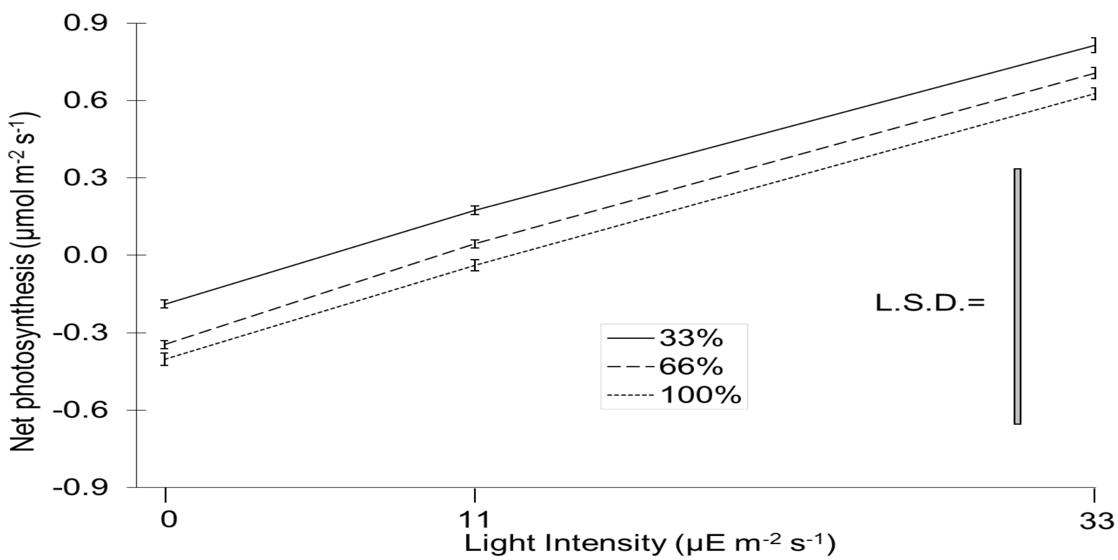

Figure 2B—Detail of steady state net photosynthesis by redwood seedlings grown in the Shade Pattern. Bars indicate standard error of the mean and LSD bar indicates Fisher’s least significant difference among means ($p < 0.05$). Quantum efficiency did not differ between different light intensities ($p < 0.05$).

With increased light, needles of redwood developed a palisade layer (figs. 6 and 7), increased stomatal density, and decreased specific leaf area (table 2). The seedlings grown in the treatments with higher light levels were able to develop greater leaf area, root mass, and stem tissue.

Redwood seedlings responded to low light environments by the development of needle area rather than root mass. The development of greater root mass and conductive tissue in higher light is a mechanism by which redwood seedlings grown in high light may avoid mortality from drought and perhaps, fungal infection.

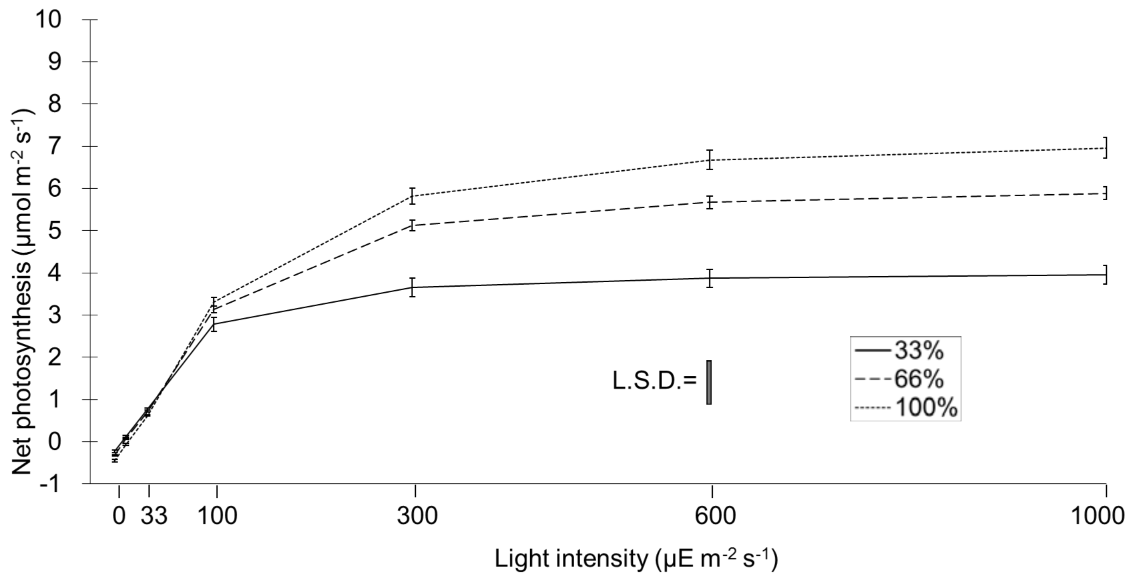

Figure 3A—Steady state net photosynthesis by redwood seedlings grown in the Sun Fleck Pattern. Bars indicate standard error of the mean and LSD bar indicates Fisher's least significant difference among means ($p < 0.05$).

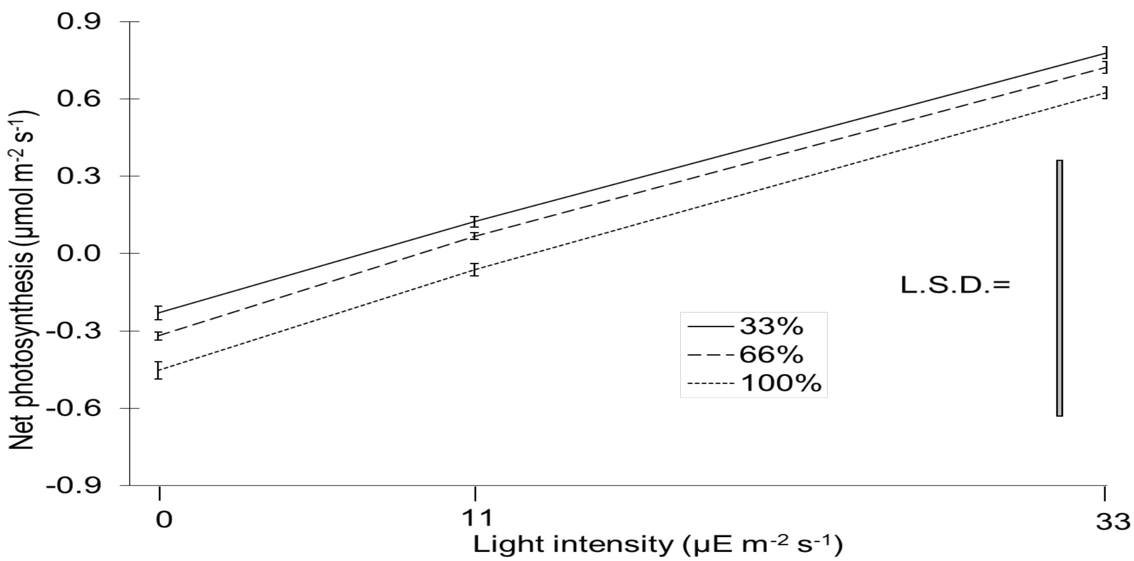

Figure 3B—Detail of steady state net photosynthesis by redwood seedlings grown in the Sun Fleck Pattern. Bars indicate standard error of the mean and LSD bar indicates Fisher's least significant difference among means ($p < 0.05$). There was a significanly lower quantum efficiency of seedlings grown in the 33 percent light intensity compared to those of both the 66 percent and 100 percent intensity treatments ($p < 0.05$).

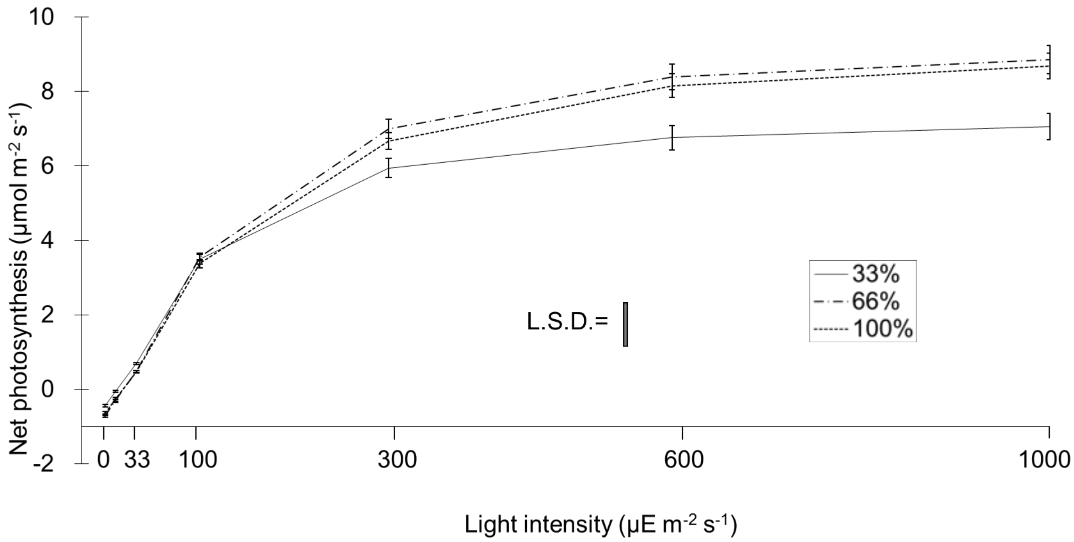

Figure 4A—Steady state net photosynthesis by redwood seedlings grown in the Large Opening Pattern. Bars indicate standard error of the mean and LSD bar indicates Fisher's least significant difference among means ($p < 0.05$).

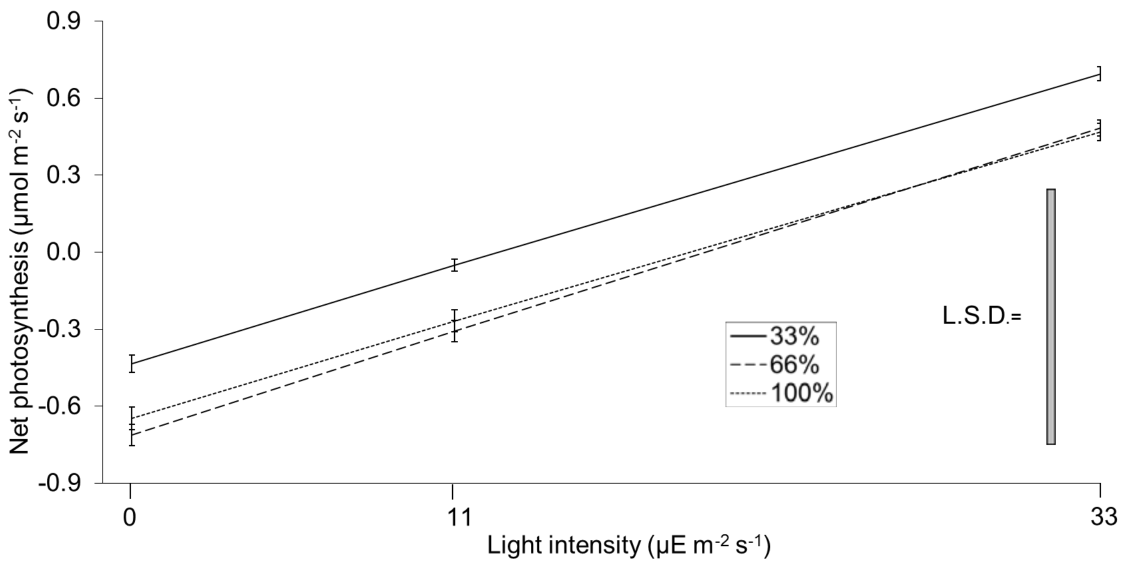

Figure 4B—Detail of steady state net photosynthesis by redwood seedlings grown in the Large Opening Pattern. Bars indicate standard error of the mean and LSD bar indicates Fisher's least significant difference among means ($p < 0.05$). Quantum efficiency (slope) did not differ between different light intensities ($p < 0.05$).

It would appear that redwood is indeed plastic, or somewhat plastic, in its response to light. Classifications of redwood generally describe it as a tolerant tree implying it is a facultative shade species that can thrive in high light. Under the conditions of this experiment, redwood seedlings did thrive in the higher light treatments, especially with a relatively consistent light pattern. However, the seedlings also displayed methods of adjustment to low light. Given that redwood seedlings thrive in high light, but display methods of adjusting to low light, it may be accurate to consider redwood seedlings as facultative sun plants rather than as facultative shade plants.

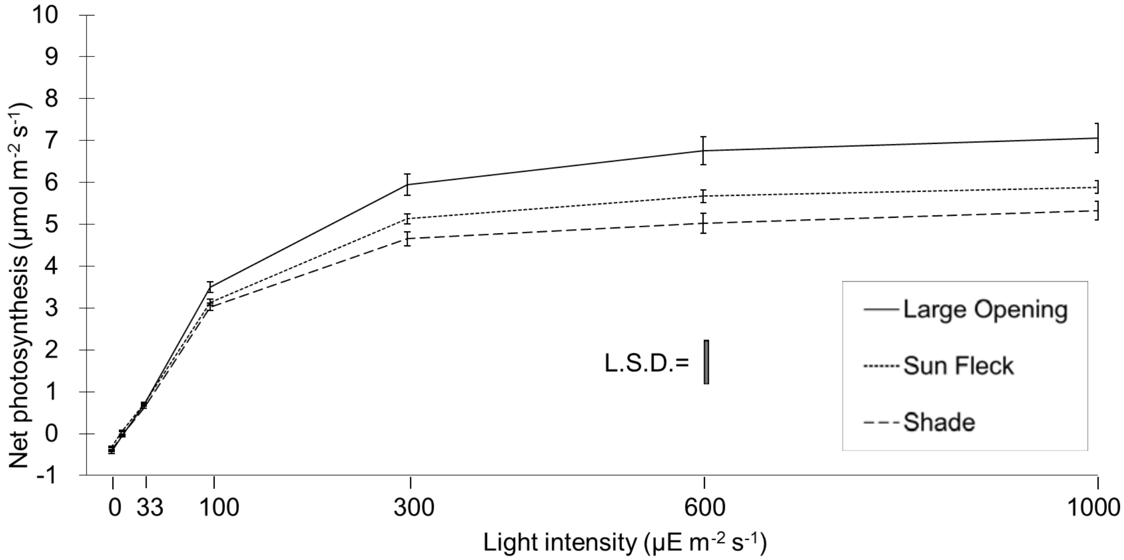

Figure 5A—Steady state net photosynthesis by redwood seedlings grown in the Equal Light Treatments. Bars indicate standard error of the mean and LSD bar indicates Fisher's least significant difference among means ($p < 0.05$).

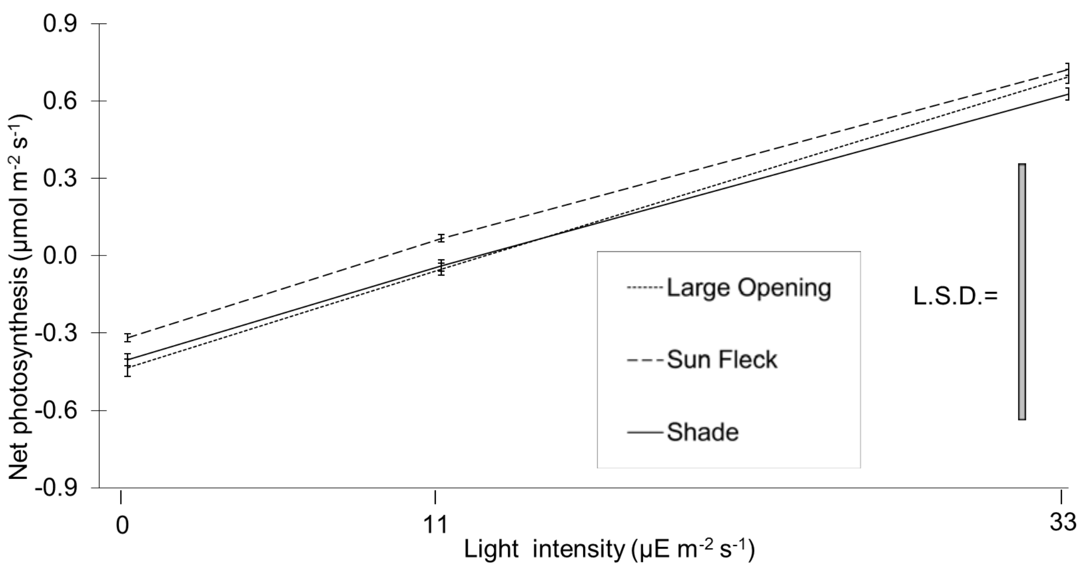

Figure 5B—Detail of steady state net photosynthesis by redwood seedlings grown in the Equal Light Treatments. Bars indicate standard error of the mean and LSD bar indicates Fisher's least significant difference among means ($p < 0.05$). There was no difference in quantum efficiency of seedlings grown in the different light patterns. Quantum efficiency (slope) did not differ between different light patterns ($p < 0.05$).

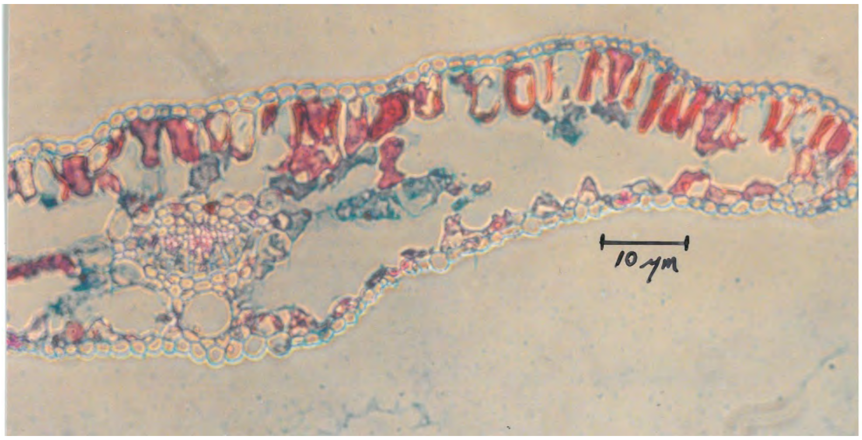

Figure 6—Needle cross section from a seedling grown in the 100 percent light intensity treatment of the Large Opening Pattern.

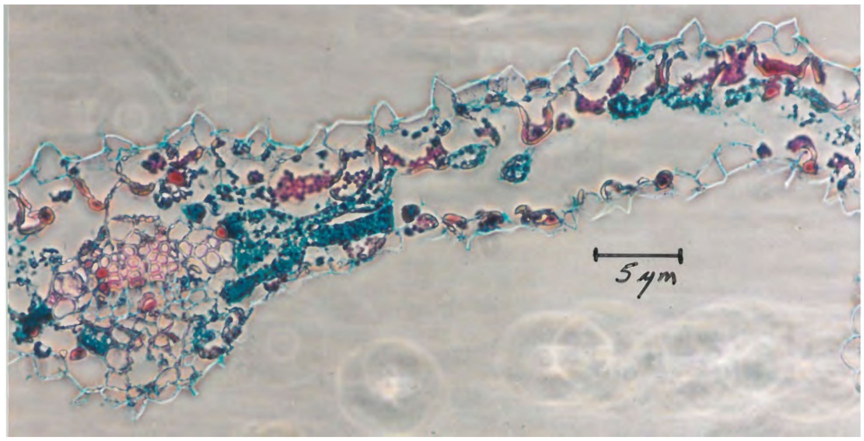

Figure 7—Needle cross section from a seedling grown in the 33 percent light intensity treatment of the Shade Pattern.

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