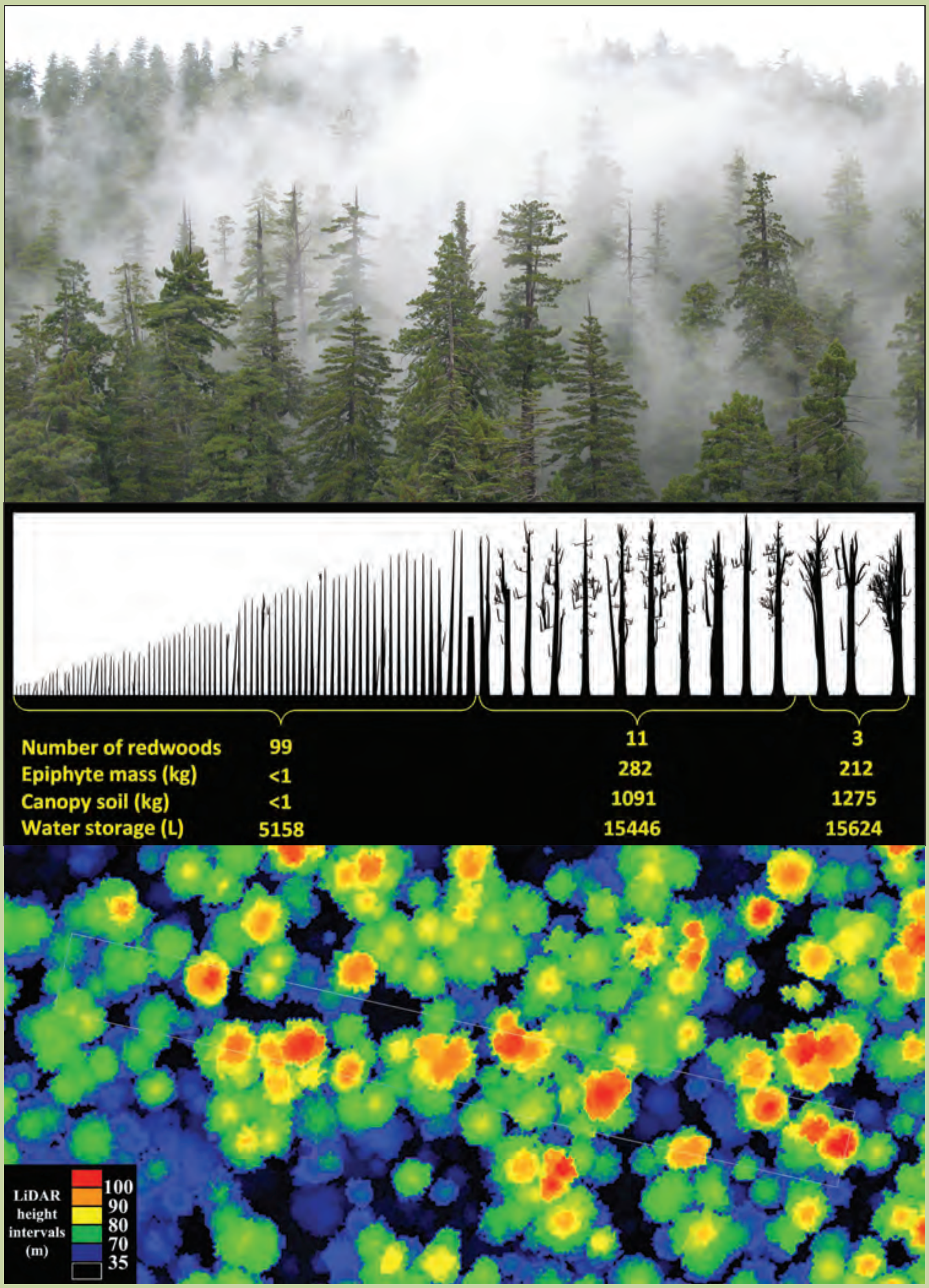
United States
Department of
Agriculture

Forest Service

Pacific Southwest
Research Station

General Technical
Report
PSW-GTR-238
March 2012

Proceedings of the Coast Redwood Forests in a Changing California: A Symposium for Scientists and Managers



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Proceedings of the Coast Redwood Forests in a Changing California Science Symposium

Technical Editors

Richard B. Standiford, Theodore J. Weller, Douglas J. Piirto, and John D. Stuart

June 21–23, 2011

University of California, Santa Cruz

U.S. Department of Agriculture

Forest Service

Pacific Southwest Research Station

Albany, California

General Technical Report PSW-GTR-238

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Proceedings of Coast Redwood Forests in a Changing California: A Symposium for Scientists and Managers

Abstract

Standiford, Richard B.; Weller, Theodore J.; Piirto, Douglas D.; Stuart, John D, technical coordinators. 2012. Proceedings of coast redwood forests in a changing California: A symposium for scientists and managers. Gen. Tech. Rep. PSW-GTR-238. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture. 626 p.

The Coast Redwood Forests in a Changing California Science Symposium was held June 21-23, 2011 at UC Santa Cruz with just under 300 registrants in attendance. Participants ranged in background from graduate level students to university forestry faculty, land managers, and conservation groups, public agencies, and land trust members. The symposium was strategically held in Santa Cruz, near the Southern end of the redwood region. Designed to present the state of our knowledge about California's coast redwood forest ecosystems and sustainable management practices, this symposium was built on earlier redwood science symposia held in Arcata, CA in June, 1996 and in Santa Rosa, CA in March, 2004.

The first day of the symposium consisted of two simultaneous field tours, one to the North County and one to the South County. The North County tour focused on active redwood timber management on corporate ownerships operating under the unique policies that dictate decision making on the central coast, and Cal-Poly's forest management and research at its Swanton Pacific Ranch. It also included a brief tour of the Big Creek Lumber Company sawmill and a visit to areas burned in the more than 7,000 acre Lockheed Fire of 2009. The South County tour traversed the range of redwood forest conditions from the old growth of Henry Cowell State Park and the uncut 120 year old young growth of Nisene Marks State Park to uneven-aged young growth stands established by individual tree selection harvesting on non-industrial forestlands.

Opening remarks started the second day of the symposium and began the academic concurrent sessions. Local historian Sandy Lydon spoke about the special history of the redwoods in the region, recounting stories from his boyhood about roaming through the forests and giving a brief synopsis of the settlement of the area. Steve Sillett, Humboldt State University forestry professor, described his experiences climbing the redwoods and his discoveries in the redwood forest canopy ecosystems, as well as his findings on tree growth from dendrochronology measurements. Ruskin Hartley, Executive Director and

Secretary of Save the Redwoods League, called on the audience to set “audacious goals and collaborative actions.” He maintained that nature does not develop boundaries and that in moving forward, we should focus on a shared set of goals and that public and private land should progress simultaneously. Concluding the session, Ron Jarvis, Home Depot’s VP of sustainability talked candidly about the role of environmental sustainability practices and policies as part of the home improvement retailer’s business model. He noted that when he began in the sustainability department he undertook a two year long project to understand where every sliver of wood from over 9,000 products originated to ensure sustainable wood practices.

Over 75 concurrent oral presentations were showcased over two days, pertaining to the topics of: Ecology (15 presentations); Silviculture and Restoration (11 presentations); Watershed and Physical Processes (22 presentations); Wildlife, Fisheries, Aquatic Ecology (10 presentations); Forest Health (10 presentations); Economics and Policy (6 presentations); Monitoring (7presentations). In addition, almost 40 posters were displayed during the evening reception, ranging in topic from post-fire response, to long-term watershed research, and community forestry models. Held outside on the warm Santa Cruz evening, participants enjoyed a strolling dinner and networking with colleagues, making the reception a highlight of the symposium.

The symposium concluded with closing remarks about the future of research in the redwood region from John Helms, UC Berkeley and Mike Liquori, Sound Watershed. In addition, a panel including Dan Porter, the Nature Conservancy, Lowell Diller, Green Diamond, and Kevin O’Hara, UC Berkeley discussed the interface of research, management, and conservation. The overall discussion led to the conclusion that academic research and applied research should be made available to the field as a whole as findings progress and that more opportunities for networking and interactions should be made available to the forestry community.

Overall, the symposium fulfilled its purpose to identify key knowledge gaps, bring together multi-disciplinary teams, and help identify future opportunities for collaboration. Participants were pleased with the presenters and research shown. Many noted that a highlight of the symposium was being able to meet and interact with others whose works they had previously cited in their own research. Of the approximately one half of participants who completed the follow-up survey, 100% hoped to see more events like the 2011 Redwood Symposium.

Acknowledgments

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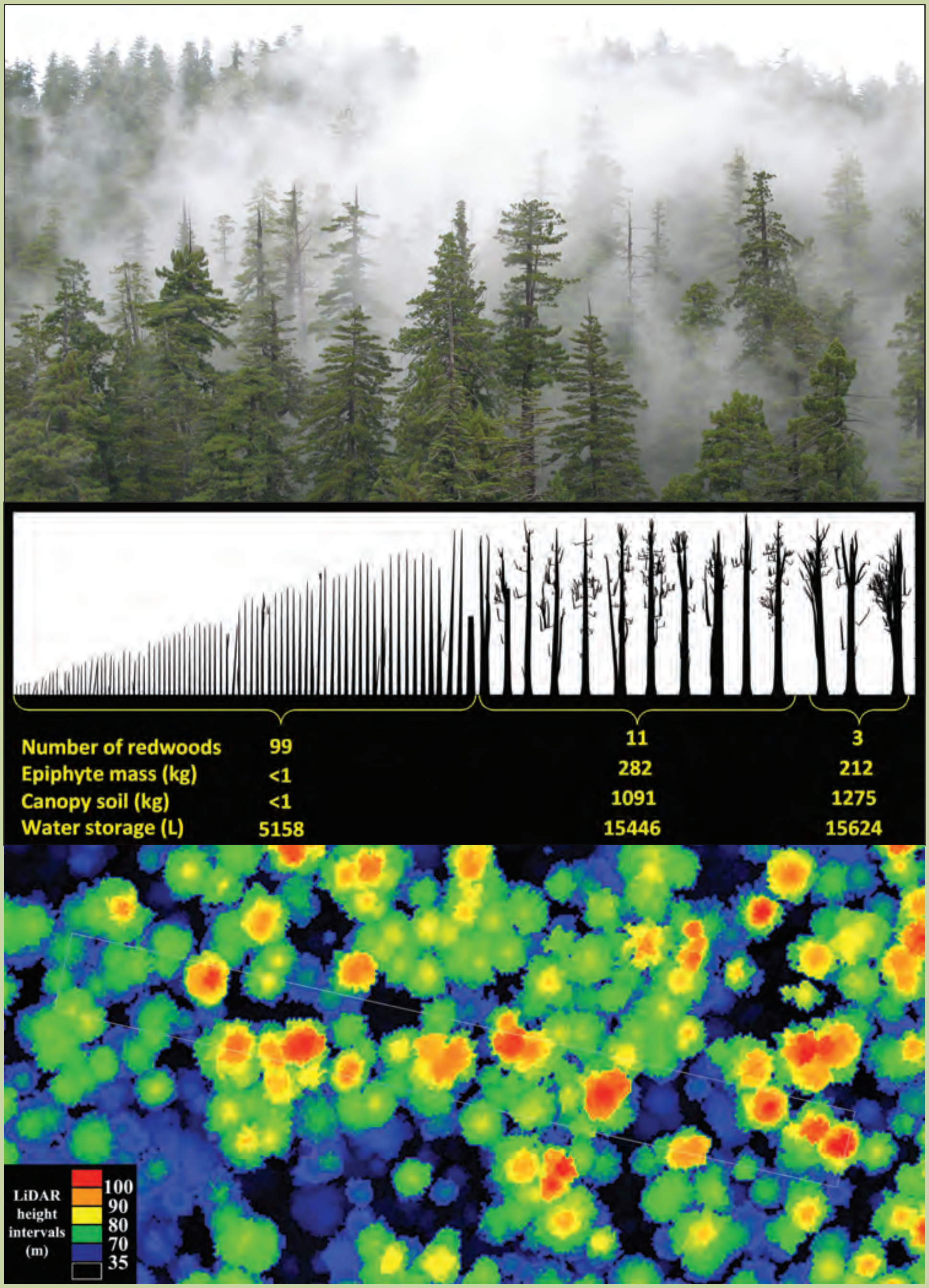

Road Surface Erosion on the Jackson Demonstration State Forest: Results of a Pilot Study[1]

Brian Barrett[2], Rosemary Kosaka[3], and David Tomberlin[4]

Abstract

This paper presents results of a 3 year pilot study of surface erosion on forest roads in the Jackson Demonstration State Forest in California's coastal redwood region. Ten road segments representing a range of surface, grade, and ditch conditions were selected for the study. At each segment, settling basins with tipping buckets were installed to measure coarse sediment and total runoff. Laboratory analysis of Total Suspended Solids (TSS) samples from the runoff provided estimates of fine sediment production.

Although the limited scope of our study prevented statistical analysis of site variables, several lessons emerged that may be useful to future studies. Total sediment production (fine and course) ranged from negligible (0.02 $kg/m^2/yr$) to more than 4.5 $kg/m^2/yr$, and varied greatly from year to year on most sites, in some cases by an order of magnitude. The share of TSS in yearly total sediment production also varied greatly from year to year. While site characteristics offer partial explanations of the results for each site, we found large variations among sites with similar characteristics. Lastly, the study required significant effort but was useful in providing site specific results; the variability in site characteristics adds to the challenge of understanding sediment production from forest roads.

Key words: Forest roads, surface erosion, coarse sediment, suspended sediment, runoff

Introduction

This paper describes instrument and study design, data collection, and empirical results from a 3 year pilot study of surface erosion associated with forest roads on the Jackson Demonstration State Forest (JDSF) in California's coast redwood region. The study was a joint effort by the California Department of Forestry and Fire Protection (CAL FIRE) and the National Marine Fisheries Service (NOAA Fisheries). Our goals were to assess the method and to collect data over a representative range of conditions.

Increasing our understanding of road surface erosion will help land managers and resource agencies improve watershed management. Watershed processes are complex and our pilot study is not definitive. While our study is too limited to support strong

[1] Results from the first year of this study were presented in Barrett and Tomberlin (2007).

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conclusions, a broader-scale application of the methods described here could provide a basis for assessing the predictive power of erosion simulation models and help to inform priorities for restoration funding.

Methods

Study area and site characteristics

The 10 road segments (sites) in this road surface erosion study are located on the JDSF in Mendocino County, California. The study area has a Mediterranean climate, with most of the precipitation (annual mean = 1.16 m) occurring as rainfall from November through April. The area is mountainous, with elevations ranging from sea level to 880 m (2,850 ft). Topography is generally steep and dissected as a result of rapid uplift rates. Underlying geologic materials are dominated by coastal belt Franciscan sandstone, and soils range from gravelly loam to fine-grained with a high clay content.

Study sites were chosen to represent a range of surface, grade, and ditch conditions typical of forest roads in the redwood region. Site selection was also influenced by operational considerations such as placement on the hillslope, travel distances, and risk of vandalism. Individual sites varied in topography, cut-bank height, ditch vegetation, overhead canopy, and year of construction or maintenance. Road surface conditions were also variable. Some native surface roads contained a fraction of native rock, while rocked roads had variations in the condition of applied rock. The road segments were selected from the existing network of crowned roads with inside ditches and ditch relief culverts, representative of many roads on the JDSF and in the region, but different from roads that utilize outsloping and rolling dips. This road type provided reasonable hydrologic isolation without re-construction or the construction of new roads. Road maintenance consisting of grading to clear ditches of soil and debris, smoothing the road surface, and removing brush for vehicle clearance, was implemented under normal scheduling on the existing road network and not controlled for the study. An example of a study site can be seen in *fig. 1*. Basic information on the sites is presented in *table 1*.

Surface flow on each road segment flowed to an inside ditch, from which a culvert directed it to devices that measured runoff and sediment production, as described below. Thus, the inside ditch was part of the road segment profile, although its relative contribution to sediment production is unknown. The catchment area for runoff on each site was measured from the base of the cut bank to the estimated crown of the road that served as a "water divide." Sporadic surface flow was observed from cut banks at four sites and varied with storm intensity. Because the actual catchment area for each segment could not be known precisely and likely varied to some extent with rainfall intensity, we have explored the sensitivity of our results to potential measurement error. However, in this paper we report results based on the single catchment area value deemed most probable for each segment.

Instrument design and calibration

Our method of estimating runoff and sediment production was based on a design by Black and Luce (2007). Each site had a settling basin that captured coarse sediment generated on the road segment, a tipping bucket with event logger that enabled estimation of total runoff, and a splash device that collected a subsample

Figure 1—Site 8, December HY07, showing rocked road surface following summer grading.

Table 1—*Site characteristics for HY05 to HY06. Minor changes in catchment area during the study period are not reflected here but were captured in calculation of results. Road maintenance did not occur for sites 3, 4, 5, 9, and 10 during the study period.*

Site (road segment)	Surface	Winter traffic	Ditch (percent vegetated)	Grade	Area (m^2)	Road maintenance (grading)
1 (1000-1)	Native	Light	10%	6%	1031	Prior to HY05, HY07
2 (240-1a)	Native	None	0%	4%	716	Prior to HY06
3 (90-1)	Native	None	10%	6%	634	
4 (210-2)	Native	None	10%	6%	778	
5 (210-1)	Native	None	10%	7%	560	
6 (240-1)	Native	None	0%	9%	399	Prior to HY06
7 (600-4)	Rocked	Light	75%	4%	757	Prior to HY07
8 (620-4)	Rocked	Light	30%	7%	452	Prior to HY07
9 (640-7)	Rocked	Light	30%	7%	723	
10 (640-1)	Rocked	Light	20%	4%	573	

of the runoff for analysis of suspended solids. A 5 ml subsample (circa 0.05 percent of tipping volume) was collected at each tip of the approximately 10 L bucket and flowed through a flexible tube into a closed 19 l (5 gal) bucket, which acted as a reservoir for composite post-storm sampling. An example can be seen in *fig. 2*.

Each tipping bucket was calibrated by providing flow at a known rate, using a flow meter, and recording the resulting duration between tips. Repeating this procedure with different flow rates enabled estimation of a calibration curve relating flow rate to duration between tips. Durations recorded by the data logger could then be used in conjunction with the calibration curves to estimate runoff on each segment, generating a hydrograph for each segment throughout the winter season.

Figure 2—Site 10 in operation with coarse sediment collection tank, tipping bucket, and suspended sediment sample bucket with tubing.

Data collection and analysis

Runoff and suspended solids data were collected throughout the rainy season. Regular visits, sometimes several per week, to the study sites were necessary to download tipping times from the event loggers. When the 19 l buckets containing splash samples were nearing capacity, or when a sample had not been taken for several weeks, a mechanically agitated composite sample was collected from the buckets for analysis of suspended solids at a commercial laboratory using the EPA 160.2 protocol (1999). We measured total suspended solids (TSS) rather than suspended sediment concentration (SSC) due to budget considerations and the finding in Gray and others (2000) that these measures differ little when the proportion of coarse sediment is small. In our study, because the settling basin captured most of the coarse sediment before TSS samples were taken, we did not expect TSS and SSC to differ markedly.

The coarse sediment captured by each settling basin was weighed at the end of the hydrologic year with a 2,000 kg (+/- 500 g) electronic dynamometer/crane scale. Each settling basin was topped off with water and the tank containing sediment and water was weighed and compared to the weight of the tank and water without sediment. These mass estimates were used to estimate the mass of dry sediment, as in Black and Luce (2007).

Results

Precipitation patterns and changes in site characteristics (largely due to road maintenance) are keys to interpreting our results. According to data from the U.S. Forest Service (USFS) Redwood Sciences Lab, precipitation at the South Fork Caspar Creek weir, located on the JDSF, totaled 1.69 m in HY06 (defined as 8/1/05 to 7/31/06), 0.86 m in HY07, and 1.00 m in HY08. Mean annual rainfall measured at the weir during 1963 to 2010 was 1.16 m (L. Keppler and J. Seehafer, USDA Forest Service, personal communication, 6/15/11).

Figure 3 shows total runoff (in kl/m^2) at each study site. Note that runoff data from sites 1 and 10 were not collected in HY08 due to constraints on staff time for travel to remote locations.

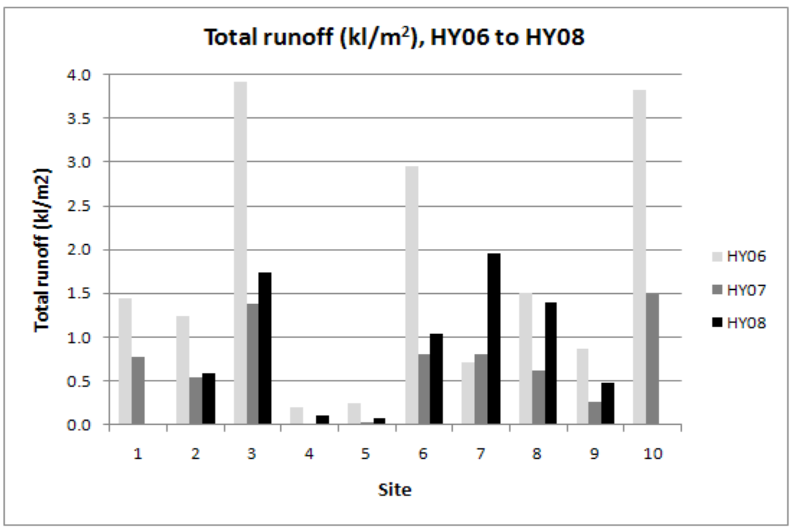

Figure 3—Total runoff (kl/m^2) for each site, HY06 to HY08. Data are normalized by area. Runoff was not collected for sites 1 and 10 in HY08.

Figure 4 shows annual sediment production per square meter on each road segment. Total sediment production ranged from negligible (0.02 kg/m^2/yr) to more than 4.5 kg/m^2/yr, and varied greatly from year to year on most sites, in some cases by an order of magnitude. While the highest-producing road segments were native surface segments, one native surface segment (site 4) produced less sediment on average than three of the four rocked surface segments.

Assuming a road surface bulk density of 1,600 kg/m^3 (Coe 2006), the sediment production rates given in *fig. 4* correspond to surface depth loss rates of 0.03 mm/yr to 2.85 mm/yr. The share of suspended solids in total sediment production generally ranged from 30 percent to 90 percent, with a mass-weighted mean of 52 percent.

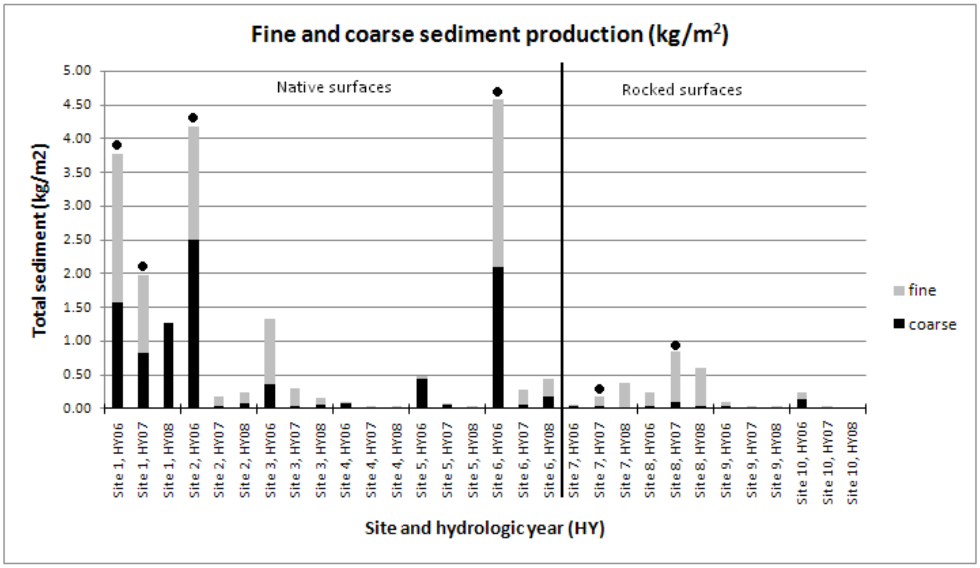

Figure 4—Fine and coarse sediment production (kg/m^2) for each site, HY06 to HY08. Data are normalized by area. Sites 1 through 6 are native surface roads while sites 7 through 10 are rocked surface roads. Fine sediment was not collected for sites 1 and 10 for HY08. Dots indicate sites where grading occurred in 1 or 2 previous hydrologic years (see *table 1*).

Discussion

Our results indicate that both total sediment production rates and the share of suspended sediment in total sediment differ greatly among sites and over time. Site 1 alone produced 45 percent of the total sediment produced on all the sites during the study period, despite the fact that fine sediment production data was not collected on this site in the final year. Site 4, in contrast, produced less than 1 percent of the total. Some of this variation can be attributed to known site characteristics, for example, site 1 has the greatest area, receives some light winter traffic, and was the only one graded prior to HY05. All four of the highest-producing segments in our study are native surface segments, but two native surface segments (sites 4 and 5) produced sediment at rates similar to the rocked segments.

As shown in *fig. 4*, the four highest annual sediment production totals were on road segments that had been recently graded to clear ditches of debris and smooth the road surface. This complicates the relationship between rocked and native surface roads because grading may have contributed more to sediment production than surface type. In the case of site 1, the first grading was prior to HY05 but the effects of grading appeared to be evident in HY06. Sediment production at sites without grading declined through the 3 year study period, except site 2 which increased slightly. Other factors, such as geology, topography, and ditch function, no doubt played a role as well, but we did not have sufficient data to support a statistical analysis of factors influencing variability in sediment production. Further, more data and analysis of erosion rates, rainfall intensities, and grading histories would be necessary to assess a cause and effect relationship between erosion and grading. For example, a simple conclusion that grading alone causes subsequent erosion does not consider the possibility that some road segments may have inherent design flaws leading to more frequent grading.

Two points are particularly worth noting in interpreting our results. First, HY06 was a very wet year (147 percent of average), and accounted for an estimated 70 percent of total sediment production on our study sites over the three-year period. Second, we have not attempted to control for the influence of traffic, since all our road segments are either closed in winter or are believed to be used by 10 or fewer light-duty vehicles (pickups and sedans) per week.

Our results are generally comparable to erosion measurements made in other forested areas of California. An application of the Water Erosion Prediction Project (WEPP) simulation model (Ish and Tomberlin 2007) generated a mean long-term surface erosion rate estimate of 4.14 kg/m^2 on native surface roads in our study area, which is similar to the higher sediment production rates shown in *fig. 4*. In studies from the interior portions of California, Coe (2006) reported a 16-fold difference in median sediment production rates between rocked and native road segments in the central Sierra Nevada, while Korte and MacDonald (2007) found that native and mixed surface roads (0.7 kg/m^2/yr) produced approximately three times the sediment as gravel surfaced roads (0.23 kg/m^2/yr) in the southern Sierra Nevada.

The results presented here are not definitive, as they represent data from only three years, and we have not yet fully explored the sensitivity of the results to uncertainties about sediment concentrations, catchment areas, and equipment function. A particularly important example of the latter was marked differences in the calibration coefficients for some tanks before and after the rainy season. Because we cannot know the rate at which the calibration coefficients changed during the season, the results reported here are based on a simple linear interpolation over time between the pre- and post-season calibration coefficients. Examining the sensitivity of our results to other possible patterns of change in the calibration coefficients—for example, such that the initial coefficients were operative until the last day of the season, or that the final coefficients were operative after the first day of the season—showed that the results were not very sensitive to this source of uncertainty.

Additionally, several known technical problems add to the uncertainty in our results. Because HY06 was a heavy rain year, 47 of 338 total data downloads (14 percent) indicated that the data logger had filled, resulting in some lost data. There was also some minor equipment damage due to site visitors, for example, on two occasions the tubing directing runoff subsamples to a collection reservoirs were removed. On site 7, an old buried culvert was found to be directing a significant amount of water from the study segment under the road and away from our instruments, which may explain the observed increase in runoff during the later, drier years of the study. We changed the filter size for TSS analysis from 1.5 to 1.0 microns after the first year, which increased by an unknown amount the estimated fine sediment production levels in HY07 and HY08 relative to what would have been captured with a 1.5 micron filter.

There are important questions related to road surface runoff and erosion that are beyond the scope of our study. We have not attempted to develop a statistical analysis of the factors contributing to road sediment production nor to assess delivery of sediment to the stream network. We did not do a comprehensive assessment of the share of organic material in our TSS samples, though in tests on a small (n=12) number of runoff samples the share of organics in TSS was found to be as high as 18 percent. This suggests that on some sites distinguishing organic from inorganic sediment would be important to estimating mineral sediment production.

The goals of this study were more limited: 1) to examine the feasibility of a particular approach to estimating road surface erosion in the redwood region, and 2) to generate estimates of sediment production on an existing network of crowned roads with inside ditches. The results presented here suggest that the method generates useful information, and clearly demonstrate that the application of generic rules of thumb to estimate sediment production at a given site may result in gross errors. The information, however, comes at a significant cost: instrumentation at each site cost approximately $1,800; project initiation and data collection during the first year required approximately one staff person per year; and maintenance, data collection, and analysis in subsequent years required approximately half of one staff person per year. However, it may be possible to lower costs and improve measurement accuracy by reducing the tank size on sites anticipated to have lower sediment production.

Conclusion

This pilot study of forest road surface erosion demonstrated remarkable variability among sites and years in the production of both fine and coarse sediment. Although the methodology was successful in providing site specific data on sediment production, the variability of site characteristics and small number of segments limits the ability to make conclusions. The results showed rocked road segments produced less sediment than most native surface segments, though two native surface segments produced amounts comparable to the rocked segments. The native surface roads with the highest sediment production had also been graded so it is unknown if the high production was the result of the surface type, grading, site characteristics, or a combination of factors. The share of fine sediment in total sediment was also highly variable, but generally more than 30 percent and in some cases almost 90 percent. These results, taken cumulatively, offer a cautionary tale about the perils of estimating sediment production based on rules of thumb.

The chief advantage of the method applied here is that it can provide useful, site-specific estimates of both fine and coarse sediment production. The disadvantages are that the method is expensive and labor-intensive, requiring significant staff time for data collection, analysis and equipment maintenance.

Acknowledgments

This project has been funded by the National Marine Fisheries Service (NOAA Fisheries) and the California Department of Forestry and Fire Protection (CAL FIRE). We thank C. Luce (USFS) for advice on instrument and study design. We also thank B. Baxter (CAL FIRE), P. Cafferata (CAL FIRE), J. Griffen (CAL FIRE), L. Keppeler (USFS), J. Lewis (USFS), and J. Seehafer (USFS) for helpful discussions and suggestions; R. Collins (CAL FIRE), B. Salee (CAL FIRE), J. Bawcom (CAL FIRE), R. Hundt (CAL FIRE), B. Perez (CAL FIRE), M. Lombardi (CAL FIRE), and A. D'Albert (CAL FIRE) for field work; S. Weir (CAL FIRE) and the Department of Corrections inmates at Parlin Fork Conservation Camp for manufacturing the tanks;

the CAL FIRE Fire Crews from Chamberlain Creek Conservation Camp for manually weighing the coarse sediment; A. Mamula (NOAA Fisheries), C. Thomson (NOAA Fisheries), and two anonymous reviewers for their comments and suggestions; and M. Jameson (CAL FIRE) and C. Thomson (NOAA Fisheries) for material support and for providing us time to work on this project.

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Total Maximum Daily Loads, Sediment Budgets, and Tracking Restoration Progress of the North Coast Watersheds[1]

Matthew S. Buffleben[2]

Abstract

One of the predominate water quality problems for northern coastal California watersheds is the impairment of salmonid habitat. Most of the North Coast watersheds are listed as "impaired" under section 303(d) of Clean Water Act. The Clean Water Act requires states to address impaired waters by developing Total Maximum Daily Loads (TMDLs) or implementing another program that will result in the attainment of water quality standards. TMDLs are an estimate of the maximum load necessary to meet water quality standards. In a general sense, a TMDL and its implementation plan is a water quality attainment strategy and provides a framework for assessing the watershed condition, evaluating the sources of pollution contributing to the water quality impairment, and developing a water quality restoration plan for the watershed.

Sediment budgets are useful tools to evaluate sediment impacts to water quality and channel morphology. As part of the TMDL development process, sediment budgets have been developed for twenty northern California watersheds. Anthropogenic activities such as logging and its associated road building, which commonly occur in this region, have dramatically increased sediment loading rates. TMDL studies estimate that is will take several decades or more for streams to transport the excess sediment out of the watersheds.

Tracking recovery of the watersheds from sediment impairments over time will be difficult. The Regional Water Board will use several tools to verify that progress is being made. These tools include sediment budgets, as well as monitoring hillslope and water quality conditions in the listed watersheds.

Key words: sediment budgets, targets, TMDLS, trend monitoring

Introduction

Over 40,000 water bodies in the United States do not meet the Clean Water Act goals of restoring and maintaining the chemical, physical, and biological integrity of the nation's waters (USEPA 2009). Excessive sediment has impaired the beneficial uses of water for nearly 6,500 waterbodies (USEPA 2009). In the North Coast Region of the California Regional Water Quality Control Board, several salmonid species are listed under the Endangered Species Act. In particular, the Central California Coast Coho populations in the southern portion of the range appear to be either extinct or nearly so, including those in the Gualala, Garcia, and Russian rivers (Good et al. 2005). Although there are several factors involved in the decline of

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salmonids, the destruction and modification of habitat are the primary reasons for decline in the western United States (National Marine Fisheries Service 2007).

Anthropogenic activities such as logging and its associated road building, which commonly occur in northern coastal California, can dramatically increase sediment loading rates (Gomi et al. 2005, Reid 1993). The increased sediment supply can negatively impact salmonid habitat in several ways. Excessive fine sediment can prevent adequate water flow through salmon redds, the nest in the stream substrate formed by a female salmonid in which eggs are laid, fertilized and incubated. Excessive fine sediment in the redd can cause a high level of mortality by limiting the oxygen supply to salmon eggs and preventing the removal of metabolic wastes (Chapman 1988). Increases in sediment supply can also decrease the pool depth and pool size (Lisle and Hilton 1999), which reduces rearing habitat for salmonids. Decreases in clarity due to suspended sediment can cause direct effects, such as mortality, and indirect effects like decreases in growth rates due to reduced food supply (Newcombe 2003).

Excessive sediment can also impair drinking water supplies, which is another concern in some North Coast watersheds. Also, excessive sediment can lead to changes in stream channel morphology. Aggradation, the filling in of a stream channel by sediment which raises the level of the streambed, may lead to decreased channel capacity, which can cause an increase in flooding frequency, magnitude and duration (Knighton 1998). This increase in flooding can cause property damage or result in nuisance conditions by limiting access for landowners.

Sediment budgets are useful tools to evaluate sediment impacts to water quality and channel morphology. A sediment budget is an “accounting of the sources and disposition of sediment as it travels from its point of origin to its eventual exit from a drainage basin” (Reid and Dunne 1996). A variety of tools and data sources are used to create the sediment budget, including monitoring information, aerial photography analysis, simple calculations, spreadsheet analysis, and computer models.

This paper reviews the sediment budgets created for 20 North Coast watersheds. The paper then discusses a monitoring strategy that can track watershed recovery and will help determine when watersheds have been restored.

Sediment budgets and total maximum daily loads

Under section 303(d) of the Clean Water Act, states are required to identify all water bodies that do not meet water quality standards. For those “impaired” water bodies, the states must develop and implement Total Maximum Daily Loads (TMDLs) or implement another program that will result in the attainment of water quality standards. A TMDL “shall be established at a level necessary to implement the applicable water quality standards with seasonal variations and a margin of safety which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality” (Clean Water Act, §303(d)(1)(C)). In a general sense, a TMDL and its implementation plan is a water quality attainment strategy, which provides a framework for assessing the watershed condition, evaluating the sources of pollution contributing to the water quality impairment, and developing a water quality restoration plan for the watershed. The North Coast Region of the California Regional Water Quality Control Board has 61 percent of the region’s area listed for sediment impairment under Clean Water Act Section 303(d) (CRWQCB 2008).

A key component of TMDLs is the source assessment. For sediment TMDLs, the source assessment typically takes the form of a sediment budget that identifies and estimates the hillslope sources of sediment. Due to the significant yearly variation in sediment loads in northern coastal California, which can range over several orders of magnitude, the sediment budgets rely on long-term estimates of sediment input. These estimates are often derived by using sequential aerial photographs to evaluate the occurrence of major sediment sources such as landslides. The sequential photographs often bracket significant storm events (for example, 1964, 1986, 1997 and 2003 storms). Therefore, the sediment budgets "average" the estimated sediment delivery over the air photo period.

While calculating the TMDL on a daily basis is a legal requirement, U.S. Environmental Protection Agency (USEPA) recognizes that it is impractical for land managers to measure sediment loads, or sediment discharges, on a daily basis. Therefore, sediment TMDLs are expressed as an average annual load which should be evaluated as a long-term (for example, 10 to 15 year) running average (USEPA 2007a). Furthermore, USEPA expects progress toward the TMDL to be evaluated by estimating the total sediment load relative to the natural load (USEPA 2007a), which is why the loading capacity is often expressed as a ratio relative to the natural loads, in addition to being provided as an absolute load. The underlying assumption is that while sediment delivery is very episodic, which makes the determinations of progress towards the TMDL very difficult, the ratio of total sediment to natural is not as sensitive to episodic events.

To date, 20 sediment TMDLs have been completed for North Coast Region watersheds. The estimated sediment loads and the TMDL, in other words the loading capacity, are shown in *table 1*. Comparisons between the sediment loads from TMDLs are difficult because different methods and categories were used to identify the sources and volumes of sediment reaching streams. However, some general conclusions can be reached. The road system is often the major source of sediment averaging 57 percent of the management-related sediment load. Logging sources of sediment averaged 24 percent of the management-related sediment load. Agriculture and grazing were very minor sources of sediment in these watersheds, while mining sources of sediment were important in some watersheds (for example, the Scott River watershed).

To provide a better understanding of sediment discharges from the 1970s through the 1990s from logging activities, Strauss (2002) reviewed the TMDL sediment budgets for several watersheds where the primary land use was logging. Seven watersheds were identified for this exercise: Noyo River, Ten Mile River, Albion River, and Big River, and portions of the Van Duzen River, South Fork Eel River, and Gualala River. The results indicated that 43 percent of the sediment delivered to streams was from natural or background sources, 1 percent was from specific sources other than silviculture and the remaining 56 percent was associated with timberland management. The amount attributed to logging was always substantial, ranging from 43 to 70 percent, and most of the logging-related sediment was associated with roads, but other processes like landslides from logging units were significant.

Table 1—*Sediment load estimates from northern California TMDLs.*

Watershed	**Size (km^2)**	**Natural (t/km^2)**	**Management (t/km^2)**	**Total (t/km^2)**	**Total (relative to natural)**	**TMDL (t/ km^2)**	**TMDL (relative to natural)**	**Reference**
Albion River	111	96	152	249	2.58	144	1.50	USEPA 2001a
Big River	469	110	110	220	2.00	138	1.25	USEPA 2001b
Eel River, North Fork	749	291	140	430	1.48	364	1.25	USEPA 2002
Eel River, Middle Fork	1950	201	29	230	1.14	211	1.05	USEPA 2003a
Eel River, South Fork	1785	378	331	708	1.88	472	1.25	USEPA 1999b
Eel River, Upper Main	1782	109	54	162	1.49	136	1.25	USEPA 2004
Eel River, Middle Main	1349	181	83	264	1.46	226	1.25	USEPA 2005
Eel River, Lower Main	774	251	272	523	2.08	315	1.25	USEPA 2007a
Garcia River	295	57	427	483	8.52	193	3.41	USEPA 1998a
Gualala River	774	133	294	427	3.21	166	1.25	USEPA 2001c
Mad River	1243	313	553	867	2.77	376	1.20	USEPA 2007b
Mattole River	767	1016	1786	2802	2.76	1261	1.24	USEPA 2003b
Navarro River	816	410	271	681	1.66	512	1.25	USEPA 2000a
Noyo River	293	130	74	204	1.57	165	1.27	USEPA 1999a
Redwood Creek	738	532	1131	1664	3.13	666	1.25	USEPA 1998c
Scott River	2106	157	105	262	1.67	196	1.25	CRWQCB 2005
Ten Mile River	311	109	111	220	2.02	137	1.25	USEPA 2000b
Trinity River	4978	379	197	575	1.52	474	1.25	USEPA 2001d
Trinity River, South Fork	2414	239	130	369	1.54	258	1.08	USEPA 1998b
Van Duzen River	1111	596	157	753	1.26	642	1.08	USEPA 1999c
Average	1241	284	320	605	2.29	353	1.34	
Median	795	220	155	429	1.77	242	1.25	

For most northern coastal TMDLs, the loading capacity has been set at or near 1.25 relative to the natural background sediment loads (*table 1*). Since estimates of current sediment loading in these northern coastal watersheds average 2.29 relative to natural background, significant reductions in sediment discharges are required to meet the TMDLs. On average, it's estimated that the management-related sediment load must be reduced by 69 percent to meet water quality standards.

These TMDLs and their associated sediment budgets are for large watersheds and many of the sediment budgets were the first attempt to analyze sediment source information at a watershed scale. It's expected that further research will increase the understanding of sediment sources and sediment loads.

Recent research has identified sediment sources that were not included in most TMDL sediment budgets. For example, Reid et al. (2010) found that gullies were a significant source of sediment following logging in the Caspar Creek watershed. Furthermore, Klein et al. (2011) recently compared turbidity, which is closely related to suspended sediment, for 28 watersheds in northern coastal California, some of which are on the 303(d) list for sediment. Watersheds with high harvest rates for the last fifteen years had turbidity levels approximately eight times greater than nearly pristine old-growth watersheds. This study indicates that logging activities continues to contribute sediment to impaired water quality conditions and that the impact of logging may have been underestimated in previous sediment budgets.

Tracking watershed recovery

Monitoring can have many different forms based on its objectives. Monitoring related to water quality regulation can be classified into the following categories: implementation monitoring, upslope effectiveness monitoring, instream effectiveness monitoring and compliance and trend monitoring (CRWQCB 2006). This paper is focused on compliance and trend monitoring, which describes monitoring that is intended to determine if water quality standards are being met. Specifically, this paper focuses on the parameters that can be used to determine the water quality conditions for cold water fish such as coho and to determine when a waterbody has recovered from sediment impacts. When monitoring data indicates that water quality standards are being met, the data can then be used for delisting from the 303(d) list.

The 303(d) Listing Policy adopted by the California State Water Resources Control Board (2004) establishes a standard methodology for assessing data and information for both listing and delisting purposes that utilizes a weight of evidence approach. While one of several approaches can be used, sediment delistings will likely use a situation-specific weight of evidence approach. This approach requires that delisting recommendations are supported by (1) data or information that affords a substantial basis in fact from which the decision can be reasonably inferred, (2) data and information demonstrates that the water quality standard is attained, and (3) a demonstration that the approach used is scientifically defensible and reproducible (CSWRCB 2004). Other approaches that can be used to delist waterbodies include demonstrating that numeric water quality objectives are not exceeded, nuisance conditions no longer exist or adverse biological response is no longer evident.

Water quality standards include the designated beneficial uses of water, water quality objectives to protect those designated uses, federal and state anti-degradation policies, and policies adopted by the State and Regional Water Boards. Water quality standards are generally contained in the Water Quality Control Plan for the North Coast Region (Basin Plan). The Basin Plan is one regulatory tool used by the Regional Water Board staff to implement the federal Clean Water Act and the state Porter-Cologne Water Quality Control Act to protect water quality. The cold freshwater habitat beneficial use appears to be the most sensitive beneficial use to excessive sediment. The water quality objectives related to sediment are narrative.

Regional Water Board staff reviewed the scientific literature related to sediment impacts on freshwater salmonid habitat to help interpret the narrative sediment-related water quality objectives. The Desired Conditions Report (CRWQCB 2006) identifies numeric targets which are directly measurable by known monitoring methods. The targets provide a means of assessing attainment, or recovery toward attainment, with the narrative water quality objectives for sediment in regards to the beneficial use of cold water fish, and specifically the freshwater habitat needs for salmonids. Non-attainment of these desired conditions, however, is not independently enforceable. The desired condition values would only be enforceable if they are specifically incorporated into a permit or if they are formally adopted as water quality objectives in the Basin Plan.

The desired conditions are intended to be used by the Regional Water Board and other agencies, organizations, and interested individuals to assess and monitor sediment impacts to water quality. Stakeholders, landowners, and other resource agencies are encouraged to monitor instream conditions and compare their data to these indices where applicable.

Since the relationship of water quality with sediment cannot be described by a single parameter, it is important to track a suite of parameters to assess watershed recovery and to determine when water quality standards are being met. There are five general categories of data and information needed to track the recovery of sediment impaired waterbodies (*table 2*). The three primary categories are channel substrate, channel morphology, and water column. Two other categories, hillslope and biological, can provide supporting evidence on watershed recovery.

The targets for the sediment monitoring parameters primarily come from the desired conditions report (CRWQCB 2006). However, other sources are also used for establishing appropriate targets, including approved TMDLs (for example, hillslope targets from USEPA 1999a) or Recovery Plans (for example, biological targets from NMFS 2010). The targets are based on current knowledge and may need to be modified depending on further research. Furthermore, some reports identify parameters whose targets are “increasing tends” (for example, increasing trend in variation along the thalweg elevation (CRWQCB 2006)). Further information on monitoring methods can be found in the referenced reports.

Table 2—*Trend monitoring categories and targets.*

Category	Parameter	Target	Reference
Channel Substrate	Substrate composition: Percent Fines < 0.85 mm	≤ 14 % of substrate is small fines	CRWQCB 2006
	Substrate composition: Percent Fines < 6.40 mm	≤ 30 % of substrate is small fines	CRWQCB 2006
	Embeddedness	≤ 25 % of gravels and cobbles are embedded	CRWQCB 2006
	Pool filling with fine sediment (V*)	≤ 20 % of the pool volume is filled with fines	CRWQCB 2006
Channel Morphology	Primary Pools	≥ 40% of reach is primary pools	CRWQCB 2006
	Large Woody Debris: small streams	> 11 key pieces/ 100 m	CRWQCB 2006
	Large Woody Debris: large streams	> 4 key pieces/ 100 m	CRWQCB 2006
Water Column	Suspended Sediment and Turbidity	Water Quality Objectives	CRWQCB 1994
Hillslope	Stream crossings with diversion potential	≤ 1 % of stream crossings have diversion potential	USEPA 1999a
	Stream crossings with failure potential	≤ 1 % of stream crossings have failure potential	USEPA 1999a
	Hydrologic connectivity of roads	≤ 1 % of the road network is connected to streams	USEPA 2003b
Biological	Spawning Adults (coho)	20 to 40 per km	NMFS 2010
	Juvenile Density (coho)	0.5 to 1.0 fish per m^2	NMFS 2010

Water column parameters in *table 2* already have water quality objectives in the Basin Plan. Regional Water Board staff does not propose to revise the standards or establish desired conditions, because further research is needed. However, several approaches appear promising, including examining turbidity exceedences (Klein et al. 2011) or using the Severity Index (Newcombe 2003) to determine water quality impacts. Also, it should be noted that suspended sediment and turbidity monitoring is helpful for monitoring trends and validating sediment budgets.

Conclusions

The sediment budgets developed for TMDLs confirm that many streams in the north coast region have suffered impacts from large inputs of sediment from anthropogenic activities. Much of the management source of sediment comes from logging and the roads that are used to access the timber. Substantial effort is needed to reduce sediment loads in these watersheds. Establishing trend monitoring programs is critical to measuring the progress of watershed recovery in these impaired watersheds. Since the relationship of water quality with sediment cannot be described by a single parameter, it is important to track a suite of parameters to assess watershed recovery and to determine when water quality standards are being met.

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Logging-Related Increases in Stream Density in a Northern California Watershed[1]

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Abstract
Although many sediment budgets estimate the effects of logging, few have considered the potential impact of timber harvesting on stream density. Failure to consider changes in stream density could lead to large errors in the sediment budget, particularly between the allocation of natural and anthropogenic sources of sediment.

This study conducted field surveys in randomly selected catchments in two managed and one old-growth watershed to determine the location of the channel's origins in the catchments. The drainage areas for identified channel heads were then delineated using a 1 m digital elevation model derived from laser altimetry. The two managed watersheds were heavily impacted by previous logging activities, particularly by tractor operations used to yard the timber out of the watersheds. The channel heads in the managed watersheds had smaller drainage areas than channels in a nearby old-growth watershed. The management activities led to a tripling of the drainage density in the managed watersheds.

Timber harvesting and the construction of skid trails used to transport timber to the road system led to increases in peak flow, ground water interception, soil compaction and drainage diversion, which reduced the drainage area necessary to initiate stream channels. Furthermore, it appears that recent ground-based yarding operations have further extended stream channels upslope, potentially creating additional sources of sediment for downstream receptors. Although these results may be unique to these watersheds, the changes in drainage density due to management activities found here emphasize the need to compare managed watersheds with undisturbed watersheds before using the current drainage network as a base-line for estimating chronic sources of sediment like bank erosion.

Key words: channel incision, drainage density, sediment budget

Introduction

Many watersheds in northern coastal California have been impaired by sediment discharges from non-point sources, particularly sediment sources related to logging activities. Efforts to assess the sediment impairment often include the construction of sediment budgets to create an "accounting of the sources and disposition of sediment as it travels from its point of origin to its eventual exit from a drainage basin" (Reid and Dunne 1996). Sediment budgets identify sediment sources and provide estimates of sediment delivery which can help prioritize erosion control efforts.

The extent of the stream network, or the drainage density, plays an important role in developing sediment budgets. Stream maps are needed to determine if discrete

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[1] This paper was presented at the redwood science symposium: June 21-23, 2011, Santa Cruz, California.
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features (for example, landslides) have delivered sediment to the network. The drainage density is also important for estimating sediment delivery from diffuse sediment-generating processes (for example, bank erosion). Topographic maps do not include the majority of headwater streams and is a particular problem in areas under forest canopy (Benda et al. 2005). Therefore, field surveys are conducted to determine the extent of the stream network in the watershed (Montgomery and Foufoula-Georgiou 1993). However, stream lengths may increase due to forest management activities, and estimated drainage densities based on only the current stream network could overestimate the natural drainage density. If the current stream distribution is used to estimate natural chronic sources of sediment and the stream network is more extensive than it had been prior to disturbance, the impacts of timber harvesting will be underestimated.

The point of transition from an unchanneled swale, also known as a zero-order basin (Dietrich et al. 1987), to a channel is referred to as the “channel head.” The channel head is the upstream limit of concentrated water and sediment transport between definable banks. Knighton (1998) describes five processes related to channel initiation: two by overland flow (Horton overland flow and saturation overland flow) and three by subsurface flow (seepage erosion, tunnel scour and shallow landsliding). These processes are not mutually exclusive, and all may be present even in a relatively homogenous landscape. However, landsliding is likely to predominate in steep areas, while overland flow and seepage erosion predominate in lower-gradient areas. The location of the channel head is affected by climate, with wetter regions needing smaller drainage areas (Montgomery and Dietrich 1988).

Hillslope gradient can also influence channel initiation. Montgomery and Dietrich (1988) reported inverse relationship between drainage area and valley gradient, especially where landslides initiated channels. Channel heads initiated by overland flow may also reflect a relationship between drainage area and gradient relationship (Montgomery and Foufoula-Georgiou 1993), as may those of gullied channels (Vandekerckhove et al. 2000). However, there are circumstances where there is no relationship between drainage area and slope (Jaeger et al. 2007, Wemple et al. 1996). Dietrich et al. (1987) noted no systematic drainage area-slope relationship at sites in Oregon where channel head locations were thought to be controlled by the flow paths through fractured bedrock.

Since the drainage area needed to initiate channels depends on climatic conditions, it seems reasonable to expect that management activities that increase runoff may also decrease the drainage area and hence increase the drainage density. Roads increase runoff because road surfaces have lower infiltration capacity than natural slopes. The drainage area needed to support a channel head is smaller for drainages receiving road runoff (Montgomery 1994, Wemple et al. 1996).

Logging is also likely to have an effect on channel head location. Prosser and Soufi (1998) observed gulley initiation during large rainfall events following forest clearing. Increases in peak flow due to a reduction in evapotranspiration (Lewis et al. 2001) likely plays a role in modifying channel head locations. Increases in peak flow could exceed the thresholds related to the channel initiation processes and decrease the drainage area for channel initiation.

In a comparison between cable-yarded clearcuts and old-growth forest, Pacific Watershed Associates (1999) found that valley catchments served as groundwater reservoirs in old-growth areas, with most runoff carried through a network of interconnected subsurface pipes. The incised channels or gullied swales within the old-growth areas are discontinuous, inactive, and located much farther downstream (in other words, have larger drainage areas) than those identified in the clearcut drainages of the harvested areas. Pacific Watershed Associates concluded the swales in logged areas had experienced gullying in response to first-cycle harvesting.

This study seeks to determine the effects of logging on stream network by comparing the stream density in two logged watersheds with a nearly pristine watershed. The field surveys also identified the channel initiation processes and the management features associated with channel heads. This information will be used to determine if the drainage density can be estimated from drainage area alone or from a drainage area-slope relationship.

Methods

In Elk River watershed, located near Eureka, California, three subwatersheds were surveyed to determine the catchment area needed for channel initiation and to examine the influence of valley gradient on the location of channel heads. These watersheds share similar bedrock, which primarily consists of the sedimentary rocks with a sheared and highly folded mudstone exposed in the deeper portions of the canyons of the watersheds. The three watersheds have average hillslope gradients of 23° to 24°. These watersheds experience a Mediterranean climate with dry summers and wet winters and with an average annual precipitation of 1650 mm. Forest stands in Elk River are dominated by redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*) (Buffleben 2009).

The primary difference between the three watersheds is their management history. South Branch North Fork Elk River (SBNFER) watershed was first logged in the 1970s, though small areas were harvested in the 1940s and 1960s. The western portion of the Corrigan Creek (CC) watershed was first logged in the 1950s and the eastern portion in the 1970s. These harvests were primarily clear-cut and tractor yarded on an extensive skid trail network. Measurements on air photos indicate the skid trail density is 32.9 and 31.4 km/km2 in SBNFER and CC respectively. Both of these watersheds experience ongoing logging entries beginning in the late 1980s, consisting of partial-cut and clear-cut harvests with tractor yarding. The portion of the Little South Fork Elk River (LSFER) surveyed in this study is primarily an old-growth redwood forest, although a 2.3 km road was constructed in the 1990s and decommission in 2003.

Since it is impractical to conduct surveys of the entire watershed for even these relatively small watersheds, the watersheds were divided into catchments from which a random selection of catchments was surveyed. Catchments within the three watersheds were delineated from a 1-m digital elevation map (DEM) derived from laser altimetry. 12 to 14 percent of the watersheds were surveyed between October 2005 and May 2006. Based on a nearby rain gauge located in Eureka, the inspections occurred during a wetter than average winter period (148 cm of rainfall, 58 percent greater than the average annual precipitation).

Field crews were provided large scale maps (typically 1:4000) of the catchments. Typically, field crews would hike up all swales and traverse other areas in the catchments to locate channels heads. Channel heads were defined as the farthest upslope location of a channel with well-defined banks (Montgomery and Dietrich 1988). Since stream channels typically begin as discontinuous segments and access to portions of the catchments was difficult due to thick vegetation and old logging debris, some subjectivity is introduced in identifying channel heads. The locations of the channel heads were recorded using Global Positions System (GPS), although if GPS reception was poor, a laser range finder was used to determine the distance to a known location (for example, a road or tributary junction). Along with the location, other attributes recorded included slope (as measured with a clinometer to a point approximately 5 m above the channel head), type (for example, spring, head cut), and management activities (for example, presence of roads, skid trails, yarding corridors, stand age). The drainage area for a channel head was defined as the upslope area draining into that feature as delineated on the 1 m DEM. Further details on the field methods and data analysis are included in Buffleben (2009).

Results

For the managed watersheds, SBNFER and CC, channel heads were found in most of the catchments and several catchments in these watersheds had multiple channel heads (*table 1*). Catchments without channel heads in the managed watersheds were small and didn’t have a major drainage axis or swale within their boundaries. It is likely that the limited drainage area in these catchments prevented erosion thresholds from being exceeded. Most channel heads in the managed watersheds were associated with some type of management feature, the most common of which are skid trails. This result is not unexpected considering the high skid trail density in these watersheds. Seepage erosion and saturation overland flow are important channel-forming processes along road and skid trail cutbanks. Tunnel scour is also commonly associated with skid trails. Landslides appear to be a minor process in channel-head formation in these watersheds.

Table 1—*Number (and percentage) of catchments with channel heads and number of channel heads with identified management associations.*

	South Branch North Fork Elk River	Corrigan Creek	Little South Fork Elk River
Catchments with channel heads	15 (94%)[a]	11 (65%)	6 (43%)
Number of channel heads	22	17	6
Road cutbank	2	1	2
Road landslide	1	-	-
Landing tunnel scour	1	-	-
Skid trail cutbank	5	4	-
Skid trail tunnel scour	8	2	-
Channel heads with identified management association	17 (77%)[b]	7 (41%)	2 (33%)

[a] Percent of catchments with channel heads.
[b] Percent of channel heads with identified management association.

The range in drainage area at the channel heads exceeds an order of magnitude (*fig. 1*). The average drainage size in these watersheds is 0.69 ha and 0.98 ha and the median is 0.42 and 0.72 ha for SBNFER and CC respectively. Catchments in LSFER are separated into two categories depending on whether or not the road passed through the catchments. Results for the five catchments that contain portions of the road are similar to those from the other managed watersheds. Three small catchments (averaging 0.6 ha in size) did not have channel heads. For the two catchments with channels, the channel heads were clearly associated with the road and the drainage areas for these channel heads reflect the road location in the catchment.

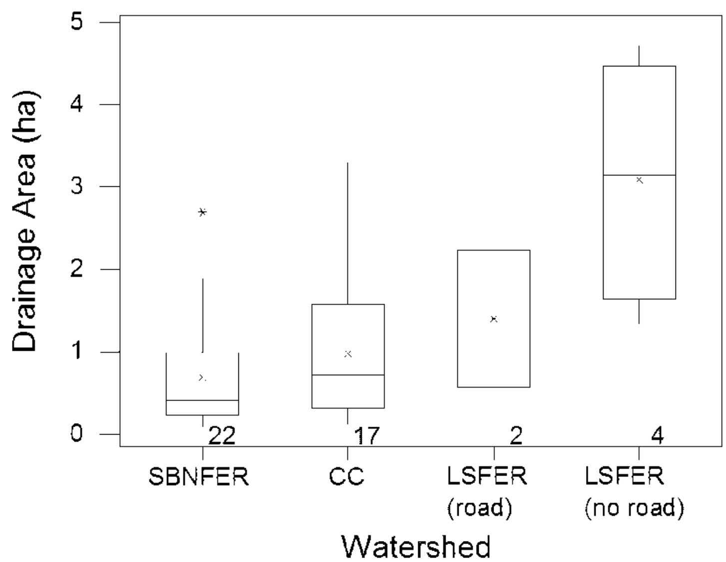

Figure 1—Box plot of drainage area of the channel heads. The number of channel heads in each group is shown above its name. The median (50th percentile) is marked by the center line within the box and the mean is shown as an X. The whiskers extend to the values that fall within 1.5 * IQR (interquartile range). Outliers are plotted with asterisks (*) when they fall outside of this range.

Nine catchments within LSFER were not affected by the road construction. Five of these catchments had no channels. While three of these catchments were small and did not have swales, two catchments were very large with drainage areas of 4.85 and 5.29 ha. These two large headwater catchments without channel heads exceeded the drainage area for the four catchments with identified channel heads, which had average and median drainage areas of 3.10 and 3.15 ha respectively. It appears that the area of the two large catchments without channel heads is below the erosion thresholds necessary to initiate a channel head. If so, it seems appropriate to include the area of these two catchments as a minimum value in determining the drainage area needed to initiate channels in the undisturbed portions of LSFER. Including these two catchment areas raises the average and median drainage area to 3.75 and 4.22 ha respectively.

The median drainage areas for channel heads in SBNFER and CC are significantly different than that for LSFER (Mann-Whitney test, $p = 0.0062$ and 0.0138 respectively). Furthermore, the p-values decrease when the two large catchments without channel heads were included in the LSFER. The drainage areas for the managed watersheds, SBNFER and CC, were combined and the median drainage area was used to construct estimated stream networks for managed conditions in the three watersheds. Likewise, the median drainage area for undisturbed catchments in LSFER, including the two large catchments where channel heads were not present, was used to construct stream networks for old-growth forested conditions in the three watersheds. Stream networks for forested and managed conditions were then compared to estimate the drainage density resulting from the timber management. The drainage density in the managed forests was to 2.7 to 3.1 times the natural drainage density.

To test for a relationship between slope and drainage area, regression analysis was conducted using the log-transformed drainage areas, since the drainage areas were not normally distributed (Anderson-Darling normality test, $p = 0.000$). Using all the channel head data in the regression analysis resulted in a poor, insignificant relationship, which indicates that slope is not a dominant factor in determining drainage density in these watersheds.

Discussion

Our surveys in the unaltered portions of the old-growth forest indicate that subterranean soil pipes play an important role in the transportation of stormflows, since infiltration rates are high and overland flow rarely occurs in undisturbed forested watersheds. We observed and measured several soil pipes at the channel heads (approximately 15 cm in diameter). It appears that soil pipes form a well-developed subterranean network and are stable enough to carry stormflows large distances downstream. Erosion thresholds are eventually overcome when several unchanneled swales merged.Timber management activities appear to have destabilized the soil pipe network and dramatically reduced the drainage area needed for channel initiation, thereby increasing the drainage density. Two aspects of management may have been particularly influential: the construction of roads (and skid trails) and the removal of vegetation. The increases in drainage density observed in these watersheds are greater than those found in previous studies (Montgomery 1994, Wemple et al. 1996). The large increases identified here may be due in part to the extremely high density of skid trails. Although only used briefly during a harvest cycle, skid trails have similar impacts as roads in that they intercept ground water, increase runoff due to ground compaction, and change drainage patterns. Skid trails were observed at many of the channel heads (*table 1*). Observations suggest that soil compaction on skid trails may play a role in tunnel scour and roof collapse in soil pipes.

The reduction in drainage area for the channel heads may have other contributing factors other than the presence and impacts of skid trails. Many of the channel heads in the logged watersheds were not associated with management features (*table 1*). Vegetation removal is likely to have reduced the drainage areas for channel heads through several mechanisms. Vegetation removal increases runoff due to reductions in transpiration and interception (Lewis et al. 2001). The increased runoff can destabilize the soil pipes and form gullies (Reid et al. 2010). Another factor that may contribute to destabilization of soil pipes is the reduction in root strength, which could decrease soil cohesion and resistance to erosion.

This study did not observe an inverse drainage area-slope relationship. One possible reason for the lack of this relationship may be the significant scatter in drainage area for a particular slope, thereby making it difficult to observe a trend (Jaeger et al. 2007). Also, it is possible that the range of slopes was too narrow to detect a slope-area relationship. Landslides, typically occurring on steep slopes, are present in these watersheds. However, only one channel head in these surveys was associated with a landslide. The lack of landsliding may indicate that these watersheds lacked significant portions of steep slopes, which would diminish the ability to detect a drainage area-slope relationship. However, a trend regarding the drainage area-slope relationship was observed. As the drainage areas have been reduced by management activities, channel heads have moved closer to ridgelines, where swale-axis slopes are steeper.

The increase in drainage density observed in these watersheds is important to consider during construction of sediment budgets. An increase of drainage density suggests greater peak flows which could add to channel erosion and sediment yields. Furthermore, if a sediment budget used the existing drainage density to estimate the sediment delivery from soil creep, it would be overestimate the sediment delivery from this natural process.

Furthermore, it is clear that in the past large amounts of sediment have been delivered to the stream network due to the shift in location of the channel head (Pacific Watershed Associates 1999). These relatively new channels, caused by management activities within the last 100 years, may still be unstable and are potentially chronic sediment sources due to continued headcutting and bank erosion occurring within the channels (Reid et al. 2010).

In several catchments in the Corrigan Creek watershed that had recent timber harvesting operations, we observed the upslope migration of channel heads. These channels appeared to be intercepting groundwater flow from skid trails used in the recent logging operations. However, these newer channel heads may only be temporary seeps that are due to the increased runoff associated with the harvest. Also, since our surveys took place during a wetter than average year, the new channels may not become permanently established or become chronic sources of sediment.

Given that water quality is impaired in the Elk River watershed and that it is extremely difficult to manage gully erosion once it has initiated, steps to prevent upslope migration of channel heads should be considered when developing plans to mitigate the impacts of future logging. Tractor operations and construction of new skid trails should be minimized along a swale axis. Furthermore, to reduce the increases in peak flows and loss of cohesion due to vegetation removal, partial-cuts should be considered instead of clear-cutting in well-defined swales.

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Physics-Based Simulations of the Impacts Forest Management Practices Have on Hydrologic Response

Adrianne Carr[1] and Keith Loague[2]

Abstract

The impacts of logging on near-surface hydrologic response at the catchment and watershed scales were examined quantitatively using numerical simulation. The simulations were conducted with the Integrated Hydrology Model (InHM) for the North Fork of Caspar Creek Experimental Watershed, located near Fort Bragg, California. InHM is a comprehensive physics-based hydrologic-response model. The North Fork watershed (including 11 tributary catchments) is the site of an ongoing study monitoring the impacts of forest practices. InHM was parameterized and calibrated using existing data and new field measurements of soil-hydraulic properties. Continuous long-term simulations were conducted for three wet seasons: before logging, after logging, and after a period of regrowth. Simulated increases in flow and peak discharges were considerably higher after clearcut harvesting. Concept-development simulations of cumulative watershed effects (CWEs) examined potential impacts of alternative timber harvest levels and methods relative to those that occurred in the North Fork watershed. Results from these simulations show that the increases in the simulated discharge after clearcutting were significant for the catchment and watershed scales and that relatively small changes in soil-hydraulic properties produced substantial changes in hydrologic response. The simulations in this study illustrate that timber harvesting can alter the streamflow generation mechanisms and patterns within a catchment.

Key words: cumulative watershed effects, forest hydrology, hydrologic-response, InHM

Introduction

The impacts of deforestation on the amount and timing of streamflow has been a key environmental concern for centuries (see Andreassian 2004). Removal of vegetation in a forested ecosystem decreases evapotranspiration and rainfall interception, leading to increased discharge and soil-water content (see reviews by Andreassian 2004, Bosch and Hewlet 1982, Brown et al. 2005, Jones and Grant 1996, Jones and Post 2004, Jones et al. 2001). Decreased interception and evapotranspiration caused by timber harvest can lead to higher peak flows, higher stormflow discharge depths, and higher subsurface pore pressures which can cause greater suspended sediment transport, increased downstream flooding, and greater likelihood of landsliding, respectively.

Paired watershed studies to date have mostly provided empirical analysis of the impacts of timber harvest on annual water discharge, peak flows, and sediment production. The studies of Andreassian (2004), Bosch and Hewlett (1982), Bowling et al. (2000), Brown et al. (2005), Burges (2003), Jones and Grant (1996), Jones et al. (2001), and Thomas and Megahan (1998) lead to the following generalizations: (i)

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streamflow responses to deforestation and regrowth are variable and dependent upon climate; (ii) smaller catchments have much more variability in response to vegetation change; (iii) if the average response is not captured (stationary relationship), then the monitored impacts may not be interpreted correctly; and (iv) observed data records can be quite poor. The observed changes in watershed response can be examined, but only through the limited perspective of the climate that occurs during the monitoring periods before and after logging. When the largest storm events are missing from the data set, the ability to forecast the impacts from extreme events is compromised. Simulation can provide answers to some questions, especially when the observed data are poor or site monitoring is limited (Alila and Beckers 2001). For example, physics-based simulation conducted in a what-if concept-development mode can provide useful insights for land managers concerned with the impact of future timber harvest/land management scenarios (Dunne, 2001, Dunne et al. 2001, Loague et al. 2006, Loague and Ebel in press).

This study examined the impacts of alternative forest management practices on near-surface hydrologic response using comprehensive physics-based simulation. The effort reported here employed the Integrated Hydrology Model (InHM) for simulations of the North Fork of Caspar Creek Experimental Watershed, located near Fort Bragg, California. The Integrated Hydrology Model (InHM) was developed by VanderKwaak (1999) in the spirit of the Freeze and Harlan (1969) blueprint. InHM was designed to quantitatively estimate in a fully-coupled approach, 3D variably saturated flow and solute transport in porous media, 3D variably saturated flow and solute transport in macropores /fractures, and 2D flow and solute transport over the surface and in open channels. The fundamental and innovative characteristics of the deterministic-conceptual InHM, including no *a priori* assumption for a specific hydrologic-response mechanism, are discussed by Vanderkwaak (1999) and Vanderkwaak and Loague (2001). Several successful applications of InHM have been reported in the last decade (for example Ebel et al. 2007, 2008; Heppner and Loague, 2008, Heppner et al. 2007, Mirus et al. 2007, Ran et al. 2011, and VanderKwaak and Loague 2001).

The Caspar Creek Experimental Watershed (see *fig. 1*) is the site of a long-term monitoring study of the impacts of timber harvest. In 1962 the California Department of Forestry and Fire Protection and the U.S. Department of Agriculture, Forest Service Pacific Southwest Research Station, Redwood Sciences Laboratory began a paired watershed study using two similarly sized basins within the headwaters of Caspar Creek watershed (see Ziemer 1998). The North and South Fork Basins are shown in *fig. 1*. The second phase of the Caspar Creek study, beginning in 1985, was a multiple paired-catchment study in the North Fork of Caspar Creek that evaluated the impacts of the then state-of-the art timber harvest practices (Lewis 1998). *Figure 2* shows a selection of the data collected during the study and the gaging stations on the 11 North Fork tributary catchments and on the main channel that monitored streamflow discharge from 1986 through 1995.

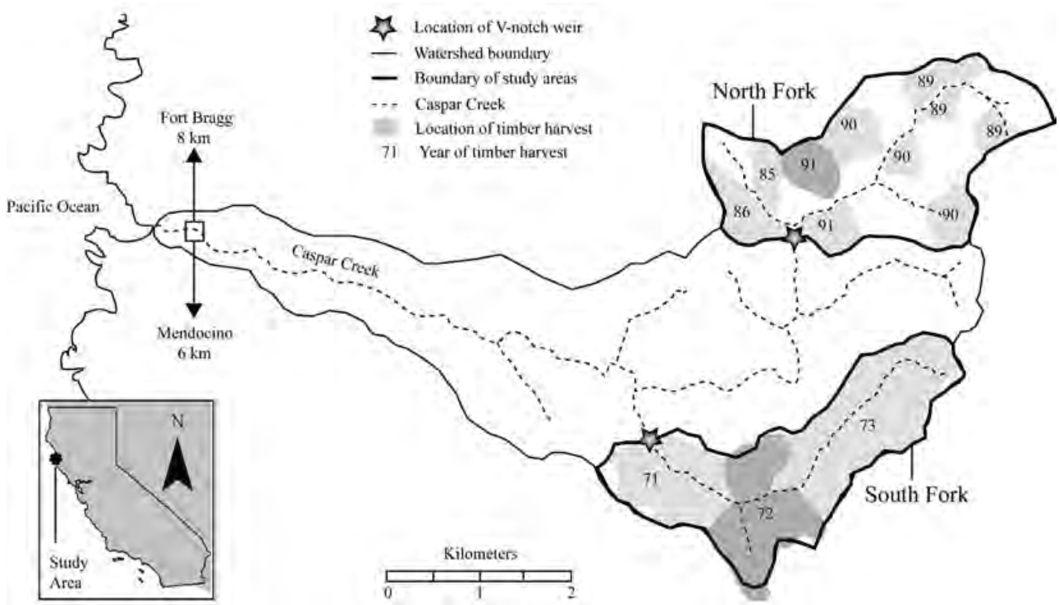

Figure 1—The Caspar Creek Experimental Watershed located within the Jackson State Demonstration Forest in coastal Mendocino County, California (after Ziemer 1998).

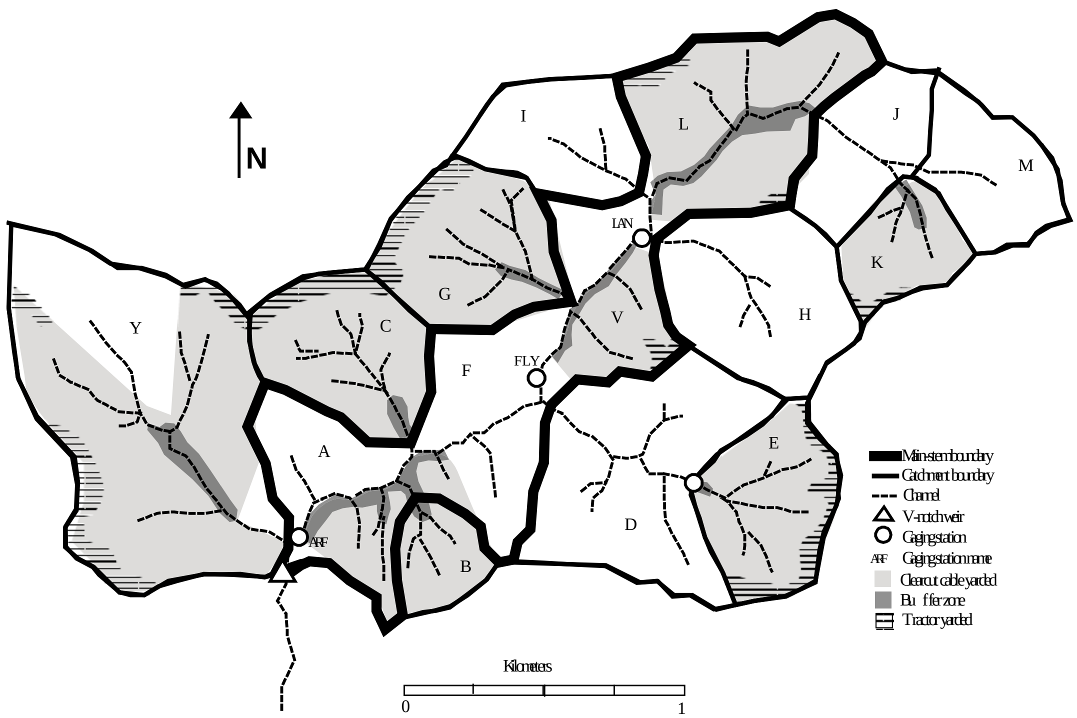

Figure 2—Outline of the main-channel boundary-value problem on a map of the North Fork of Caspar Creek. Management history, main-stem gaging stations, and North Fork catchments are identified.

Catchment- and watershed-scale simulations

Hydrologic-response simulations designed to investigate the impacts of timber harvest were conducted with InHM for the 11 North Fork Caspar Creek tributary catchments and subsequently for the entire North Fork watershed. Simulations were conducted for the wet seasons of 3 water years: (i) before logging – water year 1989 (WY89), (ii) after logging – water year 1992 (WY92), and (iii) after a period of regrowth – water year 1995 (WY95). The long-term hydrologic-response simulations were conducted for the 5 to 6 month rainy season that included the majority of the rainfall during the water year. The hydrologic response for each of the Caspar Creek catchments and the entire watershed was examined before and after logging. Changes in hydrologic response relative to forest management (for example, clearcutting) were assessed by comparison of the results from different simulation scenarios.

Boundary value problems

Thirty-three boundary-value problems (BVPs) were set up for catchment-scale simulations and three BVPs were set up for the watershed-scale simulations. Each BVP started with a 3D finite-element mesh for the area of interest. For the watershed-scale simulations, a main-channel finite-element mesh included all areas of the North Fork watershed not considered by the catchment-scale simulations. The mesh for the M catchment BVP is shown in *fig. 3*; the outline of the surface of the main-channel BVP is shown in *fig. 2*. For each BVP the bottom, upstream, and lateral catchment boundaries were impermeable. The surface boundary for each mesh was a specified flux (potential evapotranspiration and throughfall). The downstream boundaries for each BVP were a local head for the subsurface and a critical depth for the surface. The local head boundary condition is described by Heppner et al. (2007). For this study, the local head values were chosen to be water levels at downstream gaging stations for the catchment-scale simulations and the elevation of the confluence of the North Fork with the Middle Fork of Caspar Creek for the watershed-scale simulations. The x-y node spacing on the surface varied between 2 to 10 m at the channel and 24 to 60 m at the watershed boundaries with an average of 2,157 surface nodes. The surface of the main-channel mesh was made with an average x-y node spacing of 12 m for the channel and 40 to 55 m for the watershed boundaries, with a total of 3,522 surface nodes. For all meshes, the layer spacing in the vertical direction was 0.05 m for the near-surface layer, grading to 5 m at the bottom of the mesh.

Soil properties

To parameterize and calibrate InHM for the North Fork BVPs for this study, historical information was obtained and new field measurements of soil properties were made for the North Fork of the Caspar Creek watershed. The new information was needed to specify soil-hydraulic properties within the Caspar Creek flow system. The measurements made for this study included saturated hydraulic conductivity, soil texture, and soil-water retention. *Table 1* presents the range of soil hydraulic properties used in the simulations.

Climate and vegetation

The surface boundary conditions for the InHM simulations were estimated with BROOK90 (Federer 1995). BROOK90 is a process-based model designed to estimate evapotranspiration and soil-water movement in one dimension. BROOK90 solves the Shuttleworth and Wallace (1985) version of the Penman-Monteith equation to

Table 1—*Soil-hydraulic properties for the catchment boundary-value problems for the Caspar Creek watershed.*

		Range of values		Base case			
						van Genuchten parameters	
Depth (m)	**Nodal spacing (m)**	**Saturated hydraulic conductivity (ms^{-1})**	**Porosity (-)**	**Saturated hydraulic conductivity K_s (ms^{-1})**	**Porosity (-)**	**α (m^{-1})**	**n (-)**
0 - 0.15 a	0.05	3.5×10^{-2} - 1.1×10^{-1}	0.40 - 0.57	1.1×10^{-1}	0.55	30.00	1.57
0.15 - 0.3[b]	0.05	2.2×10^{-5} - 3.0×10^{-4}	0.45 - 0.47	2.2×10^{-5}	0.45	4.24	1.82
0.3 - 1.5 [c]	0.05 - 0.10	2.6×10^{-6} - 4.0×10^{-6}	0.42	2.6×10^{-6}	0.42	8.51	1.58
3.0 - 55.0 [d]	0.5 - 5.0	2.0×10^{-7}	0.10	2.0×10^{-7}	0.10	4.30	1.20

[a] soil layer 1; [b] soil layer 2; [c] soil layer 3; [d] bedrock

Table 2—*Vegetation properties for the BROOK90 simulations of the Caspar Creek watershed.*

Vegetation properties	**Second growth WY89**	**Clearcut WY92**	**Regrowth WY95**	**Source**
Leaf area index (-)	10	0	1	Reid and Lewis (2006)
Tree height (m)	60	0	1.5	Noss (2000)
Leaf width (m)	0.004	0	0.004	Fedderer (1995)
Maximum leaf conductance (ms^{-1})	0.0053	0	0.003	Korner et al. (1979), Korner (1994)
Maximum plant conductivity (ms^{-1})	30	0	5	Fedderer (1995)
Fraction of plant resistance in xylem (%)	25	0	7	Phillips et al. (2003)
Minimum plant leaf water potential (Mpa)	-2	0	-0.5	Koch et al. (2004), Woodruff et al. (2004)
Albedo (-)	0.14	0.15	0.15	Fedderer (1995)

determine the potential evapotranspiration from a single layer canopy and the soil surface based on climate and vegetation information from Caspar Creek. BROOK90 also estimates net throughfall, after rainfall interception and evaporation using a simple water balance method. The reader is referred to Federer (1995) for a complete description of BROOK90.

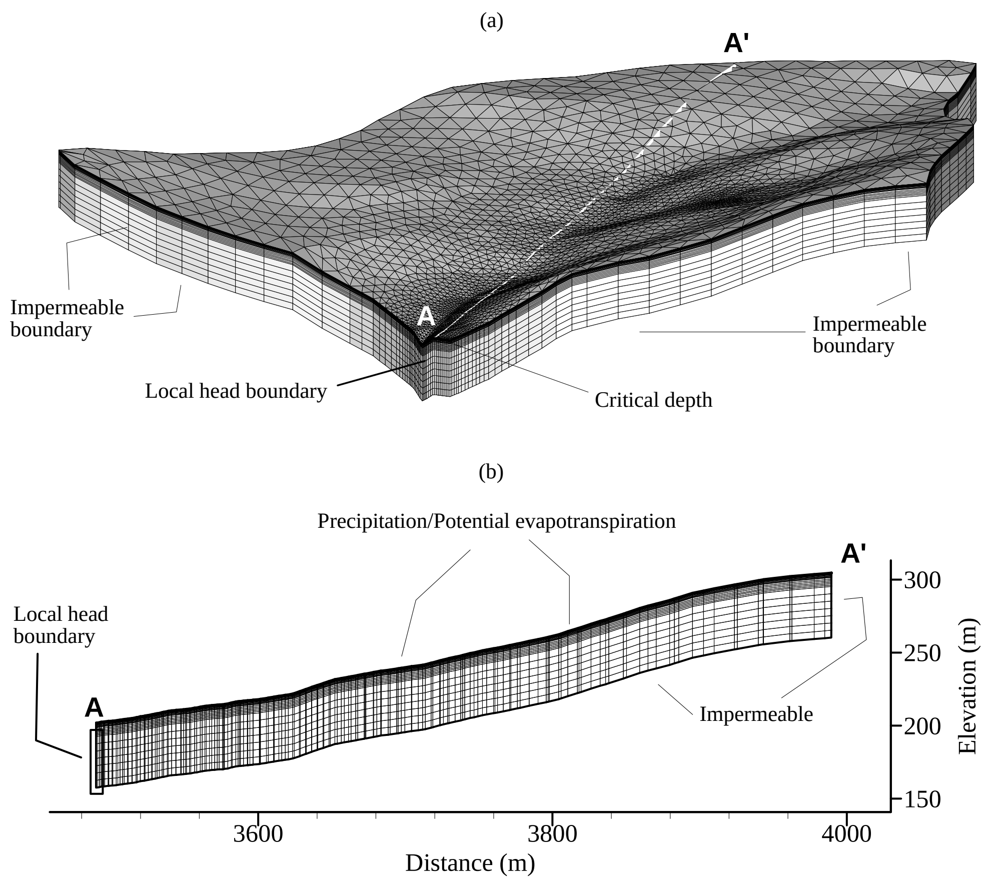

Figure 3—M catchment boundary-value problem used for simulations of hydrologic response and management impacts in the Caspar Creek watershed. (a) 3D mesh and boundary conditions. (b) Vertical cross-section taken from A-A' in (a) showing the mesh and boundary conditions with no vertical exaggeration.

The climate forcing data input to BROOK90 measured at Caspar Creek were (i) precipitation at five minute intervals, (ii) daily maximum and minimum air temperature, and (iii) daily solar radiation. Vapor pressure was approximated with BROOK90 using an algorithm developed by Murray (1967). Daily wind speed for the Caspar Creek watershed was derived from the NCEP/NCAR Reanalysis Data (Kalnay et al. 1996). Monthly average values of air temperature or solar radiation (calculated from 1986 to 1996) were substituted for missing measurements. When available, the vegetation properties were chosen based on site-specific information from Caspar Creek. *Table 2* provides the vegetation property values used in the BROOK90 simulations. Throughfall and potential evapotranspiration estimates from BROOK90, in 5 minute intervals, were used in the InHM simulations. The interception estimates from BROOK90 were calibrated against measurements from Caspar Creek (Reid and Lewis 2007).

Simulation procedure

The long-term simulations were conducted for the 5 to 6 month rainy season that included the majority of the rainfall during the entire water year. Initial conditions for each BVP were estimated (by simulation) before each rainy season. Then there was some calibration of InHM before the final hydrologic response simulations for the Caspar Creek catchments. For the watershed-scale simulations, the results from the long-term catchment-scale simulations were applied as input to the main-channel simulations for all 3 water years. The simulated hydrologic response results for the Caspar Creek catchments and the entire North Fork watershed were compared with observed data to (i) better understand hydrologic response related to the impacts of logging and (ii) evaluate the performance of InHM.

Model calibration

The M catchment had the most complete observed discharge record for the water years focused on in this study and was, therefore, used to parameterize and calibrate InHM. The M catchment simulation was calibrated to observed discharge data. The calibration was initially done for the first large storm (event 1) of WY89 and then for the entire WY89 simulation period. The InHM-simulated hydrologic response for the M catchment reproduced the transient observed discharge record to an acceptable level. Therefore, the M catchment calibration was used to parameterize all of the other catchments for this study. No calibration was done for the main-channel watershed-scale simulation. Further details of the calibration can be found in Carr (2006).

Management impact simulations

To simulate the impacts of forest harvesting at Caspar Creek, the BROOK90 vegetation properties were changed to represent clearcutting (WY92) and regrowth (WY95). The management impact simulations represented timber harvest patterns that occurred in the North Fork of Caspar Creek before WY92 (*fig. 1*). All of the BROOK90 parameter values for the three simulation periods are shown in *table 2*. After clearcutting, the leaf area index was set to zero in the catchments that were clearcut and was set to five for the main-channel simulation that was partially cut. As a result, there was no interception in the clearcut areas; and all of the rainfall was assumed to reach the ground as throughfall. Due to lack of field information, the soil properties were unchanged in the InHM BVPs after logging. It should be pointed out, however, that the cable yarding method of timber harvest is known to be fairly non-intrusive to soils (Laffan et al. 2001, Swank and Elliot 2001), so the assumption of minimal soil disturbance is a valid one.

Results

Model performance

Simulated streamflow was compared with observations for nine of the 11 catchments and for the watershed-scale results. Streamflow data are available in ten-minute intervals for all catchments, providing a good data set to compare to simulation results. However, upon close inspection one finds that much of this data was interpolated from sparse measurements. In some cases, actual streamflow measurements were much more infrequent than the ten-minute interval. In the early years, the data resolution was acceptable during most large storms but was sparse or absent during low-flow periods. The data quality improved in later years. Model performance was often poor during periods of poor data resolution. InHM performed best during the wetter periods for this study. Specifically, the model performed better for big storms and best for large events with high antecedent moisture content. It should be noted that WY89, WY92, and WY95 had average, below average, and above average annual rainfall. The performance of InHM was best for WY95, the wettest year and the year with the most complete observed data. A full discussion of model performance for this study can be found in Carr (2006).

Management impacts on hydrologic response

The increases in throughfall and decreases in potential evapotranspiration caused by timber harvesting had significant impacts on the simulated hydrologic response for the Caspar Creek watershed. Downstream measurement points captured the response in watersheds with a portion of the upstream area clearcut (partial clearcut) and the entire area clearcut (full clearcut). There was a large increase in simulated discharge depths during WY92 and WY95 after partial and full clearcut harvesting of the drainage area. The simulated increases in discharge depth after logging for WY92 and WY95, compared to the second growth response are shown in *table 3*. The most significant factor impacting the discharge of the catchments after logging was the increase in throughfall. Increases in throughfall result from reduced interception and the loss of evaporation from intercepted water.

Table 3—*Percent increases in simulated discharge depth when compared to the second growth response.*

	WY92		WY95	
	Catchment	**Watershed**	**Catchment**	**Watershed**
Partial clearcut	127	121	51	54
Full clearcut	204	NA	105	NA

Cumulative watershed effects simulations

The what-if concept-development simulations performed for this study examined the potential impacts of alternate timber harvest levels and methods relative to those that occurred between 1989 and 1991 at the North Fork of Caspar Creek watershed. Both the catchment- and watershed-scale hydrologic responses to different methods and magnitudes of timber harvesting were considered.

The CWEs simulations were carried out in three parts: (i) the hydrologic response changes resulting from a clearcut vegetation scenario were examined for the unharvested Caspar Creek catchments; (ii) the impacts of 100 percent clearcut timber harvest were examined for the entire North Fork watershed; and (iii) the impacts of soil compaction due to logging with skidders on seasonal hydrologic response and on the hydrologic response to a large rainfall event were examined for the nested catchment group K, M, and J. Using the same catchment and the same climate, thechanges in hydrologic response as a result of forest clearing and alternate forest practices were directly evaluated against the hydrologic response with a second growth vegetation scenario.

The simulated peak discharges for the largest storms increased approximately 40 percent at both the catchment and watershed scales as a result of 100 percent clearcut timber harvesting. In addition to the change in the water balance caused by a decrease in interception and evapotranspiration after logging, soil compaction directly impacted the overall hydrologic response. The simulated peak discharges for the largest storm increased an average of 63 percent for the K, M and J catchments (with changed soil hydraulic properties), illustrating this impact. The significant increases in simulated discharge in this study could cause considerable increases in the sediment carrying capacity.

This research represents a first step towards using comprehensive physics-based simulation to investigate the impacts of timber harvest. The work conducted for this study sets the stage for unraveling the complex processes that lead to hydrologically-driven cumulative watershed effects (CWEs) such as flooding, slope-stability, sediment transport, and their impacts on, for example, salmon habitat health. Changes in watershed function will most certainly accompany logging activities, but thresholds for adverse significant changes should, ideally, be decided upon before management actions so that adverse impacts can be prevented. By defining thresholds of, for example, maximum peak discharges, significant adverse CWEs can be avoided by proper management. The simulation-based approach presented here could/should be extended and employed to identify strategies for preventing/reducing adverse CWEs.

Acknowledgments

We thank the dedicated scientists and land managers at the USDA Forest Service Redwood Sciences Lab and California Department of Forestry at the Jackson State Demonstration Forest for their help throughout the years. The support of our long-term colleague Joel VanderKwaak is greatly appreciated. Thanks also to Brian Ebel, Ben Mirus, Chris Heppner, Laurie Benson, Adam Aables, Ben Kozlowicz, Kirsten White, Richard Edward Watts, Jon Sack, and Jennifer Aspittle for being excellent field assistants. This effort was supported by a US Environmental Protection Agency STAR fellowship, an ARCS of Northern California fellowship, a Stanford Graduate Fellowship, the now defunct Center for Earth Science Research (CESIR), and the Department of Geological and Environmental Sciences at Stanford University. We are grateful to the anonymous reviewers for their thoughtful comments.

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Erosion at Decommissioned Road-Stream Crossings: Case Studies from Three Northern California Watersheds

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Abstract

Post-treatment erosion was observed for 41 decommissioned road stream crossings in three northern California watersheds. Sites were purposefully selected in order to characterize the nature and range of post-treatment erosional responses. Sites with the highest visible erosion were selected in order to better understand the dominant process and incorporate any lessons learned into future projects. Sites were also intentionally selected where post-treatment erosion appeared to be negligible, or excavation techniques were judged be have been fully effective at removing erodible material. In these cases, our objectives and methods remained identical, but we wanted to examine the conditions that led to the apparent negligible erosion.

Results are consistent with other findings in the region. Erosion volumes ranged from 1.5 m^3 to 60 m^3, or, 0.1 percent to 4.5 percent of the initial volume excavated during treatment. Erosion averaged 11 m^3 per site or 0.4 percent of excavated volume in the Headwaters Forest Reserve and 21 m^3 per site or 2.4 percent of excavated volume in Lacks Creek. Repeat monitoring of a sub-set of sites in the Headwaters Forest Reserve over subsequent years, indicates that 99 percent of post-treatment erosion occurs in the first year following treatment. Channel incision is the dominant process of sediment production from treated sites, accounting for 80 percent of observed erosion. In response, woody debris has been incorporated into recently excavated crossings with the intent of providing armor and roughness elements to reduce channel incision. At those sites where post-treatment erosion is apparently minimal, incision remains the dominant process of sediment generation and can exceed 20 m^3. In some cases, excavation techniques were judged to be largely effective at removing erodible material, however, interstitial material stored between larger, more immobile clasts can produce surprisingly large erosion volumes as this material is winnowed away during higher flows.

Key words: erosion, road decommissioning, watershed restoration, sediment

Introduction

Removal of logging roads is an established method for reducing ongoing and potential sediment delivery to aquatic habitat sensitive to sediment inputs (for example, Madej 2001, PWA 2005). Across the redwood region of northern California, road removal programs implemented by a variety of landowners have resulted in substantial reductions in potential sediment delivery. For example, in the Redwood Creek watershed, for the period 1998 to 2000, road treatments have reduced the quantities of potentially deliverable road-related sediment by 58 percent

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in the lower portion of the watershed on National Park lands, and 18 percent in the upper two-thirds of the watershed[3].

Several efforts in the northern redwood region have documented erosion following road decommissioning at road-stream crossings (*table 1*). Most of these studies have shown that channel incision is the dominant form of erosion to occur, followed by side slope failures and surface erosion. All of these studies have documented post-treatment erosion, with the bulk of the erosion occurring in the first winter following treatment (*table 1*).

The intent of this study is to examine post-treatment erosion at road-stream crossings to inform managers, project supervisors and equipment operators on project effectiveness. Site selection was intended to develop a series of case studies to explore the range of responses experienced at decommissioned crossings. In this paper, our specific objectives are to:

1) Describe the principal erosional processes at work.
2) Quantify erosion volumes in the years following treatment.
3) Assess the temporal distribution of erosion.
4) Quantify erosion quantities at sites where best management practices appeared to be met, but erosion was still evident.
5) Contemplate measures to reduce erosion from future treatment sites.

We compare our findings from BLM-managed lands to studies conducted elsewhere in the northern redwood region and offer suggestions for future investigative work.

Study sites

The Bureau of Land Management (BLM) manages land in the Redwood Creek, Elk River and Salmon Creek watersheds. Redwood Creek and Elk River watersheds are listed as sediment impaired under the Clean Water Act 303(d) listing process. The areas contain an extensive network of older logging roads. In addition, all three watersheds provide habitat for federally listed salmon and steelhead populations.

Headwaters Forest Reserve

The Headwaters Forest Reserve (Reserve) is located in the headwaters of Salmon Creek and the South Fork Elk River, both tributaries to Humboldt Bay in Humboldt County, California. The Reserve was established in 1999 (BLM-CDFG 2004) and encompasses 30 km^2 of mixed redwood and Douglas fir forest. Approximately 12.5 km^2 of the Reserve is old growth redwood forest with little road construction. Approximately 85 km of roads were constructed throughout the Reserve from the 1950s through the 1990s. Many of these roads were constructed along streams or geologic contacts where a natural bench was present.

A previous road assessment identified approximately 50 km of road in the South Fork Elk River watershed (PWA 2004). The assessment identified nearly 83,000 m^3 of potentially deliverable road sediment. Of this, approximately 58,000 m^3 was identified from 242 stream crossings (PWA 2004).

[3] G. Bundros, Geologist, Redwood National and State Parks. June, 2011; personal communication.

Table 1—*Summary of studies investigating erosion at decommissioned road-stream crossings in northwest California.*

Author(s)	Location	# sites	Average erosion (m^3)	% of excavated volume[a]	Comments
Klein (1987)	Redwood Creek, Humboldt County	24	26.6	n/a	
Madej (2001)	Redwood Creek, Humboldt County	207	50	4.8	Incision and assoc. bank erosion most common
Klein (2003)	Upper Mattole River, Humboldt/Mendocino Counties	18	11.9	n/a	Channel incision 88% of observed erosion
PWA (2005)	Northern California	614	26	5	Incision greatest # of sites, surface erosion greatest volume
Cook and Dresser (2007)	Six Rivers National Forest	262	21	4.5	40% incision 60% bank failure
Keppeler and others (2007)	Caspar Creek, Mendocino County	25	24.6	4	83% of erosion after first winter
Maurin (2008)	Smith River, Del Norte County and Redwood Creek, Humboldt County	16	6.8	n/a	Noted need for further work on woody debris
PWA (2008)	Redwood Creek, Humboldt County	16	8.8	0.9	
Klein (2009)	Lost Man Creek, Humboldt County	30	14.7	0.4	15% of additional erosion following initial winter
Wilson (2008)	Headwaters Forest Reserve, Humboldt County	8	5.4	n/a	Incision 98% of observed erosion

[a] Expressed as total eroded volume as a proportion of excavated volume. See Klein (2003) for discussion on various means of displaying erosion results.

Road removal efforts began in the Reserve in 1999, focused on those road segments with the greatest potential to deliver sediment. Over the period 1999 through 2010, 34 km of road have been decommissioned in the Reserve encompassing both the Salmon Creek and South Fork Elk River watersheds.

Lacks Creek

Lacks Creek watershed is a 50 km^2 tributary to Redwood Creek in Humboldt County, California. The BLM manages approximately 60 percent of the watershed with the remainder in private ranch lands. Road assessment efforts have identified approximately 150 km of roads in the watershed with the potential to deliver approximately 150,000 m^3 of material (Bundros et al. 2004). Over the period 2007 through 2010, 13.6 km of road have been decommissioned and 26.3 km have been upgraded along the 101 km of road on BLM lands in the watershed.

Methods

Site selection

Sites were selected in order to provide a series of “case studies” to inform managers, project supervisors and equipment operators on the range of erosional conditions experienced following crossing removal. Site selection was based on our intent of capturing instances of larger erosional responses in the hopes of avoiding such occurrences in the future. Sites were visited following the first winter after treatment. In the Reserve, a sub-set of sites were re-occupied to assess erosion in subsequent years.

Erosional processes

We considered three types of erosion during our site visits: channel incision, bank failure, and rill/gully erosion along channel sideslopes. In practice, we recognized that some bank failures were likely triggered by channel incision. Regardless, we recorded these as bank failures. At each site, we recorded eroded volume. No attempt was made to quantify what proportion of the eroded volume was delivered to the channel network.

Tapes and marked depth sticks were used to record linear dimensions of eroded features. Where erosion patterns were relatively uniform, such as channel incision throughout the crossing, we measured width at three to six cross sections through the site and depth at three to seven points along each cross section. Values were averaged for each cross section and multiplied by the length of the feature. Where numerous or complex erosional features were present, we divided the site into two or more separate erosional features and calculated erosional volumes. Individual erosion features < 0.5 m^3 were not recorded.

Results

Forty two sites were assessed for erosion following treatment (*table 2*). At the Reserve sites, we revisited 19 sites from 2 to 10 years following our initial visit to determine the extent of any additional erosion.

Table 2—*Erosional responses from 41 decommissioned crossings on BLM-managed lands.*

Location	# sites	Average erosion (m^3)	% incision	% erosion occurring first year	% excavated volume[a]	Average drainage area (ha)	Average channel slope (%)
Lacks Creek	13	21	89	n/a	2.4	16.5	33.7
Headwaters Forest Reserve	28	10.8	68	99	0.4	20.7	25.5

[a] Expressed as total eroded volume as a proportion of excavated volume. See Klein (2003) for discussion on various means of displaying erosion results.

Discussion

Erosional volumes

Our results are consistent with others in the region. Average erosion across the 10 studies reviewed and summarized in *table 1* was 19.5 m^3 compared with the 21 m^3 and 11 m^3 observed in Lacks Creek and the Reserve, respectively. Expressed as a proportion of excavated volume, values also reflect the greater erosion in Lacks Creek. Readers are encouraged to refer to Klein (2003) for a discussion of various ways of expressing eroded volumes in context of either the site or watershed-scale sediment budgets. In short, we chose to express our erosion volumes in the context of excavated volumes due to reporting requirements and a lack of watershed-wide sediment budget data which would enable us to report the erosion volumes in the context of watershed-wide sediment sources.

We observed relatively large erosion volumes from sites where all appropriate treatment methods had been employed. Finished side slopes, channel slope and width through the site were appropriate for the setting. At one site in Lacks Creek, 54 m^3 of channel incision occurred. This was the largest site by drainage area (168 ha). The channel developed on this terrain flows over boulders in excess of two meters diameter. Winnowing of remnant fill material from amongst the boulders and consequent incision upstream was the source of the eroded sediment. However, in our judgment, further excavation was not possible given the equipment on site. Any further reductions in potentially erodible volume would have come with great time and expense to dissect the material out from the boulder insterstices.

We also note that a seemingly small incision notch over a long crossing can produce eroded volumes on the order of 2 to 6 m^3. While “under excavation” is often cited as a reason for channel incision, our findings suggest that some amount of channel incision is unavoidable at many sites.

Temporal distribution of erosion

In the Reserve, we revisited 19 sites a second time 2 to 5 years after treatment. Five of the sites were visited a third time. Only four sites showed evidence of subsequent erosion and the erosion observed was <1 m^3. Our results indicate that 99 percent of the post-treatment erosion occurs in the first winter following treatment. However, during our site visits in the Reserve, we noted that heavy re-growth in the years following treatment may have concealed some instances of subsequent erosion. However, we are confident that any larger erosional features (>5 m^3) would likely have been detected. This is consistent with others who noted the bulk of erosion occurring in the first winter season (*table 1*).

The role of woody debris in erosion rates

In response to our findings, we have begun placing woody debris in the channel to arrest, or at least minimize channel incision. The effectiveness of wood placement on post-treatment erosion has not been systematically studied. Maurin (2008), working in Lost Man Creek, concluded that more work was needed on woody debris placement strategies. His small sample size precluded any conclusions on the effectiveness of various placement strategies. Our initial observations suggest that wood placement may be least effective on areas of extensively sheared bedrock such as occurs in the Incoherent Unit of Coyote Creek. In these areas, channel incision leaves the wood ineffectively perched above the channel bed. In other areas, however, the placement of logs nearly perpendicular to the channel appears to create a series of storage compartments that effectively traps sediment eroded from upstream. We agree with Maurin (2008) that more work is needed in this area to determine the best strategies for incorporating woody debris into excavated stream crossings.

Conclusions

Our observations of erosion following the decommissioning of road-stream crossings add to a growing data set of regional observations. Both the volumes and style of erosion are consistent with others' observations in northwest California. Our results suggest that post-treatment erosion likely cannot be avoided and that channel incision is the dominant erosional response. Areas warranting further examination are treatment of crossings in areas of highly sheared bedrock and the best use of woody debris in these and other sites.

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Large Woody Debris Budgets in the Caspar Creek Experimental Watersheds[1]

Sue Hilton[2]

Abstract

Monitoring of large woody debris (LWD) in the two mainstem channels of the Caspar Creek Experimental Watersheds since 1998, combined with older data from other work in the watersheds, gives estimates of channel wood input rates, survival, and outputs in intermediate-sized channels in coastal redwood forests. Input rates from standing trees for the two reaches over a 15 year period varied from a high of 28 m^3/km/yr in the North Fork (in 1995) to a low of 0.12 m^3/km/yr in the South Fork (from 2000 to 2002). Rates in the South Fork, where a second-growth forest was selectively logged in the 1970s, have consistently been lower than those in the North Fork, where partial clearcutting in the 1990s left buffer strips along the channel. Since 2004, inputs in both reaches have been between 2 and 8 m^3/km/yr and are dominated by inputs from snags. More than 90 percent of the volume of conifer pieces that entered the channel in 1995 was still present in 2010; alder pieces from that period are mostly gone. Wood budgets demonstrate the contrast in wood volumes between the two channels and the importance of stored wood in the system.

Key words: in-stream wood, wood budgets, large woody debris, buffer strips

Introduction

An important aspect of sustainable forest management is maintaining adequate levels and types of large woody debris (LWD) in stream channels. Because LWD inputs are often episodic and pieces can persist for decades in the channel, an LWD budgeting approach can be a useful tool to estimate long-term effects of management activities, but such budgets are limited by the reliability of the estimates used for inputs, survival, and outputs. In the Caspar Creek Experimental Watersheds, an influx of LWD in the North Fork channel in 1995 created an opportunity to monitor piece persistence in that channel and to compare input and survival rates between the North and South Fork channels, which are adjacent to stands with contrasting forest management histories.

There is a large body of literature on LWD volumes in stream channels, but few studies have documented LWD dynamics over time (Wohl et. al. 2010). At Caspar Creek, O'Connor and Ziemer (1989) sampled LWD in the North Fork channel, and Surfleet and Ziemer (1989) sampled wood in both the North Fork and South Fork channels shortly after the North Fork logging. The current study differs from earlier Caspar studies by using monitoring of all large pieces within defined reaches to understand what causes changes in volume over time.

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[1] This paper was presented at the redwood science symposium: coast redwood forests in a changing California. June 21-23, 2011, University of California, Santa Cruz.

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Methods

The study area is in the Caspar Creek Experimental Watersheds, Jackson Demonstration State Forest, Mendocino County, California. Stream-adjacent stands are dominated by coast redwood (*Sequoia sempervirens*), Douglas-fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), and, in the South Fork, red alder (*Alnus rubra*). Both watersheds were clearcut, with channel clearing, in the 1860s through the early 1900s. Selective logging (with stream-adjacent roads) in the South Fork in the early 1970s removed about 2/3 of the standing volume. A series of small clearcuts with partially cut buffers along the mainstem channel removed approximately 45 percent of the volume in the North Fork watershed upstream of the XYZ tributary between 1989 and 1991 (Henry 1998). Some wood was removed from South Fork channels during logging. Buffer strips prevented as much wood from entering the North Fork channel during logging as had entered during South Fork logging, and any that did enter remained there rather than being yarded out. Some pieces, particularly in the North Fork, have been cut to maintain access trails for streamflow studies.

The North and South Fork watersheds have weir ponds at the downstream ends. The study reaches are along the mainstem channels upstream of these ponds, and, in the North Fork, upstream of the first major tributary. Reach slopes range from one to three percent, and mean bed widths vary from 6 m near the weir ponds to 2 m at the upstream ends of the study reaches. Characteristics of the two watersheds include:

Characteristics	North Fork	South Fork
Study reach length (m)	1822	1945
Watershed area, downstream end (ha)	391	393
Watershed area, upstream end (ha)	163	190

Field crews tagged and measured existing and new wood pieces in the study reaches starting in 1998. For each piece, median (or, for oddly-shaped pieces, average) diameter, length within a zone extending 1 m horizontal distance outside of the edge of the active channel, species, position, decay class, and distance from a benchmark were recorded. Any available evidence of input date, source distance from the bank, whether the source tree was a snag, and input mechanism was noted. Length in the active channel was estimated in the South Fork. Down live trees (leaning more than 45 degrees from vertical) and standing trees or stumps that interacted with the stream bed were included.

The original objective of the study was to monitor survival of pieces contributed by a series of windstorms in 1995 (Reid and Hilton 1998), so the 1998 tagging included all pieces > 0.1 m^3 from trees that entered the channel from 1995 to 1998 and pre-existing pieces >0.3 m diameter and >2.5 m long or >0.15 m diameter and >0.5 m^3. It quickly became obvious, however, that in order to identify new inputs (particularly from snags) all pieces larger than some minimum size needed to be tagged, so in 1999 all pieces >0.15 m diameter and >2.5 m long in the North Fork, and in 2000 in the South Fork all pieces >0.2 m diameter and >1.5 m long were tagged. In the North Fork, tagged pieces were mapped on existing channel maps that show all pieces >0.15 m diameter and >1 m long. The South Fork channel was surveyed in 2001 and wood pieces were mapped beginning in 2004; before that, pieces were located by distance upstream from the downstream end of the reach. In 2002, 2004, 2006, and 2008, existing pieces were checked and new, moved, broken, or changed pieces >0.2 m diameter and >2 m long in both channels were tagged and measured. In 2010, new hillslope inputs were tagged and the current status of 1995 inputs in the North Fork and 1995 to 2000 inputs in the South Fork was checked. The study is continuing; this paper presents preliminary results as of 2010.

I calculated channel input rates from standing hillslope trees from 1995 to 2010. For inputs from 1998 to 2010, I used new tree data from each year's survey. For 1995 inputs I used 1998 survey data and source information. In the North Fork, most trees from 1995 were flagged shortly after they fell, and most of those were checked in the 1998 survey, but because some pieces would have broken up or moved out of the channel by 1998 some volume is missing. In the South Fork, 1995 inputs are based on trees found in 1998 that appeared (from the condition of the bark, branches and twigs, and surrounding vegetation) to have been down for about 3 years. Pieces from snags were difficult to identify and age, so snag inputs are likely to be underestimated in both forks through 1998.

Wood volume budgets (inputs, changes, and net gain or loss) were calculated for pieces >0.2 m diameter and >2 m long for each survey interval. Because field crews did not tag all pieces larger than that size in 1998, I estimated volumes that were present in the channel in 1998 but not tagged. Many pieces were later tagged, and I used notes from the field crews, existing pool maps, and, in the North Fork, existing channel maps, to determine which pieces had been present since 1998. However, there were presumably some pieces >0.2 m diameter and >2 m long that were not tagged in 1998 and left the channel between 1998 and 2002 (or, in the South Fork, 2000), when tagging began on all pieces of that size or larger.

In the North Fork, the channel maps provided information needed for an estimate of this missing volume. For all pieces marked as "gone" in 1999 or 2002, I measured the mapped length within a zone defined by lines drawn 1 m outside of the mapped edge of the bank and summed those lengths by mapped diameter class. I calculated the diameter distribution within each class for pieces tagged in 1999 that had been previously mapped (these pieces were smaller than all pieces tagged in 1999, because the larger pieces had been tagged in 1998), and used the percent by diameter within each class to calculate a total volume gone for each period. In the South Fork, I estimated volumes of untagged pieces that had disappeared by 2000 but were shown in pool sketches from a 480 m subreach drawn in 1998 and 1999.That loss estimate was adjusted for logs not in pools (92 percent of logs tagged in 1998 in the pool subreach were in pools), and scaled back up to the full length of the study reach. Both estimates are approximate, and are likely to be accurate to ±50 percent.

Results and discussion

Hillslope input rates

Input rates from standing trees from 1995 to 2010 (t*able 1*) show the combined effects of episodic events and stand history. High input rates in the North Fork in 1995 and 1996 show the effects of the 1995 windstorms, particularly in buffer strips adjacent to clearcuts, while those same storms had much less effect in the smaller, younger stands in the South Fork. Rates in the North Fork decreased rapidly though the late 1990s, then increased as new fir snags began contributing wood. Snag inputs also increased in the South Fork, both from fir snags and from riparian alders. By 2010, input rates were less erratic, and South Fork rates were approaching North Fork rates. However, inputs to both channels are dominated by short-lived species (grand fir in the North Fork and grand fir and alder in the South Fork) and may diminish over time as those species disappear from the stands.

Table 1—*Volume of large wood contributed to stream channels from standing trees by year, species, and snag status. 1996 inputs entered between January and December 1995; later inputs entered between the previous survey and the listed year. Units are m^3/km/yr of pieces >2m long and >0.2 m diameter.*

North Fork	**1996**	**1998**	**1999**	**2002**	**2004**	**2006**	**2008**	**2010**
Redwood live	2.36	0.38	0.0	0.27	0.49	0.31	0.51	0.19
Redwood snag	0.0[a]	0.0[a]	0.0	0.0	0.0	0.0	0.0	0.03
Douglas-fir live	18.96	8.06	2.06	1.30	2.56	0.63	0.0	0.85
Douglas-fir snag	2.48[a]	3.56[a]	2.70	2.11	5.93	5.17	4.33	1.96
Grand Fir & other[b] live	3.81	0.97	0.0	0.02	0.06	0.03	0.53	0.0
Grand Fir & other[b]snag	0.56[a]	0.61[a]	0.0	0.45	1.18	1.11	0.62	4.17

South Fork	**1996**	**1998**	**2000**	**2002**	**2004**	**2006**	**2008**	**2010**
Redwood live	0.30	1.64	0.0	0.02	0.53	0.90	0.29	0.24
Redwood snag	0.0[a]	0.0[a]	0.0	0.0	0.0	0.0	0.0	0.0
Douglas-fir live	0.72	0.47	0.0	0.0	0.22	0.38	0.0	0.04
Douglas-fir snag	0.0[1]	0.0[a]	0.12	0.0	0.53	1.63	0.94	1.92
Grand fir & other[b] live	0.08	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Grand fir & other[b] snag	0.0[a]	0.0[a]	0.10	0.05	0.15	0.25	0.0	0.93
Alder live	1.98	0.36	0.55	0.0	0.19	1.14	0.64	1.56
Alder[c] snag	0.0[a]	0.0[a]	0.0	0.06	0.30	0.45	0.14	1.30

[a] Snag inputs are underestimated in 1996 and 1998, particularly in the South Fork.
[b] Includes western hemlock and "fir"—pieces that might be either grand fir or Douglas-fir.
[c] Includes a small amount of tanoak.

Piece survival

Almost all of the volume of large wood input to the North Fork channel during the 1995 windstorms that was present in 1998 was still in the channel in 2010. *Table 2* shows the percent of the 1998 volume that was still in the channel at the time of each survey, with survival noted first by species and next by whether it came from a standing snag or from a live or recently-live tree. Pieces from snags are slightly less stable than those from live trees, but still quite persistent. In the South Fork, most alder volume was gone by 2010, but some pieces were still there. Conifer pieces were generally stable in that channel also, but the small number of trees makes it difficult to determine if they were more or less stable than those in the North Fork.

Table 2—*Persistence of 1995 large wood inputs over time. Values are percent of the 1998 volume of pieces input in 1995 remaining at the listed time. Numbers in parentheses are the number of source trees in each category. Very small categories have been omitted. North Fork inputs are grouped by species and by whether the tree was a snag; all 1995 trees in the South Fork were identified as "live" or "probably live." Volume increases are caused by pieces moving further into the defined measurement area.*

North Fork	Percent of 1998 volume remaining					
Species	**1999**	**2002**	**2004**	**2006**	**2008**	**2010**
Redwood (15)	95	96	96	108	105	103
Douglas-fir (74)	100	100	97	94	94	92
Grand fir (17)	98	96	96	96	96	97
Status of source tree						
Live/probably live (76)	99	98	96	96	96	96
Snag /unknown (33)	100	103	96	90	91	80

South Fork						
Species	**2000**	**2002**	**2004**	**2006**	**2008**	**2010**
Redwood (3)	100	100	100	69	79	86
Douglas-fir (2)	100	100	100	100	95	95
Alder (4)	100	100	63	59	47	33

Part of the reason pieces are so persistent in these reaches is the length of the reaches—very few pieces move entirely out of the reach. I evaluated mobility within the reaches by comparing pieces that moved more than 5 m downstream between 2004 and 2006 in the two reaches. The winter of 2005 and 2006 included the highest flow during the study. Just under 10 percent of the pieces moved in each channel (16 of 205 pieces in the South Fork, 73 of 759 in the North Fork), but moved pieces were larger in the South Fork, with a mean moved piece volume of 0.53 m^3, compared to 0.32 m^3 in the North Fork. South Fork pieces also moved, on the average, about 20 m further. The percent of total volume moved was also slightly higher, 4.2 percent versus 3.8 percent in the North Fork. South Fork pieces appear to be more mobile in a given flow year, but it would be useful to compare movement in other high-flow years.

Wood budgets

Wood budgets for all intervals between surveys are summarized in *table 3a* for the North Fork and *table 3b* for the South Fork. A significant amount of wood entering the active channel in some years was previously-down pieces or standing stumps that had been outside of the active channel and entered the measurement area due to channel migration, or that had been buried in the bed or banks and became unburied, or that entered from adjacent hillsides or terraces by being knocked in by other trees, by sliding downslope, or by landslides (standing trees that entered from landslides are listed as "trees" in the budgets). These pieces are listed as inputs from "Off-channel." Pieces that left the channel in equivalent ways (by burial, changes in channel location, or transport onto inactive floodplains) are listed as "Off-channel" losses. Budgets start with the tagged volume present at the beginning of the period. Volume losses, in addition to "Off-channel," include pieces that were never found (Gone), or broke and became too small (Too small), or were transported downstream beyond the end of the reach, usually into the weir ponds (Downstream). Volumes added include new logs from standing trees, additions from “Off-channel,” unidentified transported pieces, and pieces that moved in from upstream.

Volumes of moved and broken pieces are subtracted from volumes after remeasurement to calculate a net change, which can be positive if pieces moved closer to the channel and thus had more volume within the measurement area. Trail work can also have a positive or negative net effect. Total changes are added to the original volume to calculate the volume at the start of the next interval. Budgets for 1998 to 1999 (or 2000 in the South Fork) and 1999 to 2002 have an additional category for untagged pieces >0.2 m diameter and >2 m long that broke or disappeared during those periods. These estimated volumes are, as described earlier, not very accurate. Since these numbers make up most of the calculated total losses in the years to which they apply, losses in those years could range from half to twice the listed value.

Table 3—*Wood budgets for the North and South Forks of Caspar Creek, 1998-2008. See text for explanation of budget categories. Units are* m^3*/km over the period listed. Estimated numbers are categorized as “untagged.”*

3a. North Fork

	1998 to 1999		**1999 to 2002**		**2002 to 2004**		**2004 to 2006**		**2006 to 2008**	
Tagged volume	311.99		314.53		332.98		349.00		357.26	
(untagged)	10.4		3.8							
Start volume		322.4		318.4		333.0		349.0		357.3
Losses:										
Gone	-0.42		-1.15		-2.70		-15.64		-5.08	
Too small	-0.00		-0.00		-1.36		-1.41		-0.47	
Off-channel	-0.70		-0.33		-1.44		-4.72		-1.74	
Downstream	-1.68		0.00		0.00		-0.06		-0.12	
(untagged)	-6.59		-3.84		0.00		0.00		0.00	
Total losses		-9.4		-5.3		-5.5		-21.8		-7.4
Moved/broken	-19.54		-17.72		-24.77		-57.51		-50.52	
Inputs from:										
Trees	4.76		12.45		20.45		14.48		11.97	
Off-channel	1.40		3.58		5.01		18.13		1.81	
Unknown	2.11		2.37		1.82		6.96		0.80	
Upstream	0.00		0.11		0.39		0.17		0.00	
Total input		8.3		18.5		27.7		39.7		14.6
Moved pieces	16.89		21.16		18.63		47.80		46.73	
Net moved		-2.6		3.4		-6.1		-9.7		-3.8
Net trail work		-0.3		-2.0		0.0		0.1		-1.5
Unchanged	289.37		291.55		301.70		267.31		296.70	

3b. South Fork

	1998 to 2000		**2000 to 2002**		**2002 to 2004**		**2004 to 2006**		**2006 to 2008**	
Tagged volume	104.62		106.31		105.52		104.16		112.07	
(untagged)	3.9									
Start volume		108.5		106.3		105.5		104.2		112.1
Losses:										
Gone	-0.29		-0.97		-3.96		-4.41		-1.70	
Too small	-0.00		-0.00		-1.00		-0.00		-0.92	
Off-channel	-0.09		-0.00		-1.44		-0.00		-0.00	
Transport out	-0.00		-0.00		-0.67		-1.86		-0.00	
(untagged)	-3.9									
Total losses		-4.3		-1.0		-7.1		-6.3		-2.6
Moved/broken	-3.49		-5.60		-7.48		-13.63		-16.98	
Inputs from:										
Trees	1.54		0.25		3.84		9.51		4.02	
Off-channel	0.00		0.06		2.38		3.60		0.75	
Unknown	0.67		0.27		0.65		1.88		0.87	
Upstream	0.00		0.00		0.00		0.07		0.00	
Total input		2.2		0.6		6.9		15.1		5.6
Moved pieces	3.35		5.21		6.40		12.74		14.10	
Net moved		-0.1		-0.4		-1.1		-0.9		-2.9
Unchanged	100.75		99.73		90.88		84.27		92.47	

Most wood in both channels was unchanged in any period, and the largest amount of change was accounted for by pieces that moved within the reaches, which generally did not result in much net volume change. Both reaches showed a net accumulation over the period, but net accumulation in the South Fork was much lower than in the North Fork and almost entirely a result of high inputs in 2006; it may not represent a long-term pattern. The North Fork accumulated volume in every interval, with a net accumulation rate over the budget period of 3.3 m^3/km/yr. South Fork inputs, outputs, and changes are consistently lower than those in the North Fork. Surfleet and Ziemer (1996) noted a strong contrast in volumes between the North and South Forks in 1994 (before the 1995 windthrow). They attributed this difference to stream clearing in the South Fork between 1967 and 1973. Data from this study indicates that the contrast has continued and increased, mainly due to differences in input rates to the two channels, exacerbated by differences in piece persistence (mainly due to species composition) and possibly by higher relative mobility in the South Fork.

Inputs from “off-channel” were highest in both forks in 2006, following the highest peakflow during the study. In the North Fork, these inputs were higher in 2006 than inputs from standing trees, and much of that volume was large pieces of old-growth redwood. Many pieces moving back and forth from “off-channel” are old redwood chunks, which persist in the channel for many years.

Conclusions

Wood inputs, outputs, moved volumes and total volumes are consistently lower on the South Fork than the North Fork, in some cases much lower. There are many factors that contribute to this difference, including differences in stand age and species composition, post-logging channel clearing in the South Fork, and post-logging blowdown in the North Fork. Current input rates in the South Fork are approaching those in the North Fork, but much of that input (26 to 48 percent since 2004) is alder, which does not tend to persist in the channel. Wood is continuing to accumulate in the North Fork, partly because more than 90 percent of the volume from the 1995 blowdown events is still in the channel.

Acknowledgments

Jen Feola, Dustin Idler, Elias Stienbuck, Gina Spratt, Sven Rosengreen, Dina Ederer, Alejandro Ramirez, Shauna Little, Megan Arnold, Forrest Phifer, Basilia Lopez, Ziza Bibic, and Alana Bartoli tagged and searched for and measured logs through the years, and many other people in Ft. Bragg and Arcata entered data, answered questions, and helped in many ways. Liz Keppeler provided her usual good advice and support, and Leslie Reid and Tom Lisle provided inspiration and insight. The study was partially funded by the California Department of Forestry and Fire protection.

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Sediment Production in a Coastal Watershed: Legacy, Land Use, Recovery, and Rehabilitation[1]

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Abstract

Sediment production has been measured for nearly half a century at the Caspar Creek Experimental Watersheds. Examination of this sediment record provides insights into the relative magnitudes and durations of sediment production from management practices including road construction, selection harvest and tractor skidding, and later road-decommissioning. The 424-ha South Fork was harvested under standards that applied before passage of the 1973 Forest Practice Act. Regression analysis of annual suspended sediment loads on peak flows indicates that sediment production roughly doubled, with a return to pre-treatment levels about 11 years after harvest ended. However, sediment production again increased in the 1990s as road crossings deteriorated in response to large storms. Road crossings decommissioned in 1998 eroded a volume equivalent to more than half of the total yield in 1999 and enlarged another 20 percent over the last decade. Suspended sediment yields since decommissioning were reduced only for small storms. Recent assessment of 1970's era roads and skid trails found 443 remaining stream and swale crossings. Stream crossing have eroded an average volume of 10 m^3. Stream diversions are common, and many sites have the potential for future diversion. Diversions along incised roads and skid trails contribute to episodic sediment inputs. Mainstem sediment loads are elevated relative to those at tributary gages located above the decommissioned riparian haul road, indicating that sediment yields at the weir are enhanced along the mainstem itself. Since turbidity monitoring began in 1996, South Fork mainstem turbidities have exceeded ecosystem thresholds of concern a higher percentage of time than those in the North Fork.

Key words: erosion, legacy, logging, sediment, roads, road decommissioning

Introduction and site description

Two centuries of logging in the redwood forests of northern California have transformed the region. Although the redwoods remain the defining natural characteristic of this "other California", the landscape has been altered in a myriad of ways, some obvious and some subtle. Management activities such as timber harvest, road construction and use, and site preparation have been shown to deliver sediment, nutrients, and other pollutants to streams. Temporal and spatial variability of management impacts continues to be a topic of major concern—one that long-term research is particularly suited to address (Anderson and Lockaby 2011). The Caspar Creek Experimental Watersheds are a source of such data for the region.

[1] This paper was presented at the redwood science symposium: coast redwood forests in a changing California. 21-23 June 2011, Santa Cruz, California.

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The Caspar Creek research watersheds are within the Jackson Demonstration State Forest, 5 km from the Mendocino coast. Streamflow and sediment data are collected from weirs constructed in 1962 on the 424 ha South Fork and 473 ha North Fork and from tributary gages installed between 1983 and 2001 (*fig. 1*). Over a 20-year period of rapidly evolving management practices, the 100-year-old 2nd-growth redwood and Douglas-fir forest of this coastal basin was selectively tractor-logged (South Fork, 1971 to 1973) and partially cable clear-cut (North Fork, 1985 to 1992). New roads were constructed in both watersheds in conjunction with timber harvest. A 4.7 km mainline riparian road was built in South Fork 4 years prior to harvest and decommissioned in 1998, 25 years post-harvest. These varied treatments influence hydrologic processes, recovery time-frames, and restoration responses.

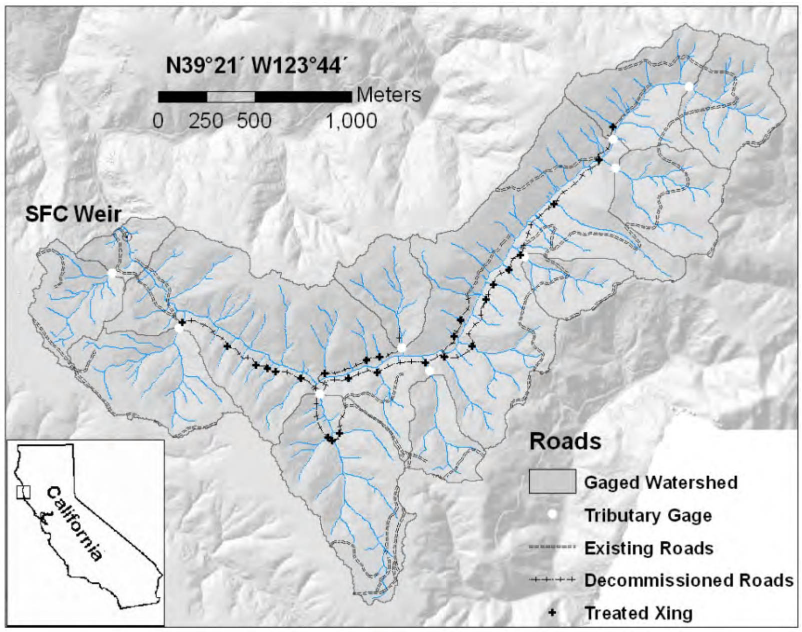

Figure 1—South Fork Caspar Creek watershed.

Several reports document enhanced sediment production in response to South Fork road construction, logging, and tractor-yarding (Lewis 1998, Rice et al. 1979, Tilley and Rice 1977). These prior reports rely on North Fork comparisons for estimates of South Fork sediment increases. Lewis (1998) found suspended sediment in the South Fork increased 335 percent the year after road construction, while yields for the next 3 years did not increase significantly. For the 6 years following logging, suspended sediment yields increased an average of 212 percent over expected values, or 331 percent when North Fork yields were adjusted to remove the effect of a large 1974 North Fork landslide. Suspended sediment yields recovered to background levels by 1979 and remained at background levels until 1985, when logging in the North Fork began. This report re-examines South Fork (SFC) suspended sediment trends through 2010 using analyses independent of North Fork (NFC) data and addresses the analytical problems presented by changing sampling protocols and episodic perturbations resulting from large landslides. Results are reviewed in light of erosion trends documented during 4 decades of field studies.

Methods

Sediment production estimates in the two watersheds are derived from water samples collected at the weirs (suspended loads) and annual bathymetric surveys of weir pond deposition. Pond deposition consists of approximately 40 percent suspended sediments and 60 percent bedload (Lewis 1998). Suspended sediment sampling protocols evolved over time. Initially, rising-limb data from fixed-stage siphon samplers were supplemented with manual DH-48 sampling during receding flows. Pumping samplers have been the primary means of suspended sediment sampling since 1976. Sediment concentrations, derived using standard gravimetric methods, were used to estimate loads using flow-duration sediment rating curves. Instream turbidimeters, deployed in 1996, are currently used in conjunction with pumped and DH-75 water samples to refine load estimates according to Turbidity Threshold Sampling protocols (Lewis and Eads 2009). Storm events are defined as the hydrograph rise to, and the ensuing recession from, a peak discharge exceeding 1.6 L/s per ha (0.17 yr recurrence interval).

Annual loads include an estimate of sediment flux between storm events and have been multiplied by a factor of 0.45 for years 1963 to 1975 to remove bias introduced by fixed-stage samplers. Lewis (1998) demonstrated the bias introduced by disproportionate sampling of rising hydrographs and the strong correlation between annual suspended loads and annual runoff. Using NFC calibration data, Lewis (personal communication, 2007) regressed annual loads on annual peaks and compared deviations to conclude that estimates prior to the 1976 change in sampling protocol are consistently higher than those of the subsequent period by a factor of 2.22 and proposed this same correction for both NFC and SFC annual loads.

To allow SFC sediment loads to be analyzed after 1985, when logging began in the North Fork control watershed, the present study employs a calibration relation between SFC annual suspended sediment loads and SFC annual maximum peakflows, using 1963 to 1967 and 1984 to 1992 data. The calibration relation is used to estimate expected annual loads under pretreatment (and post-recovery) conditions, and observed loads can then be compared with expected loads to calculate deviations associated with road construction, harvest, and later road decommissioning. A similar approach was taken using storm-based loads and peaks for the 1986 to 2009 data set to explore changes associated with road decommissioning.

The potential implications of altered sediment loads are evaluated using the 15 years of 10-min-interval turbidity now available for NFC and SFC. The number of days per year with turbidities exceeding the stream ecosystem stress thresholds proposed for Northern California watersheds (Klein and others 2008) were compared between sites using a paired t-test.

Erosion features larger than 7.6 m^3 were mapped in 1994, and subsequently after peak flows exceeding the 4 year return period (1997, 1998, and 1999). Beginning in 2000, this inventory was repeated on an annual basis. Slide dimensions and volumes were recorded.

Between 2004 and 2006, an inventory of legacy sediment sources was performed in the South Fork watershed based on protocols used for the Sinkyone Wilderness State Park Road Rehabilitation Project (Merrill 2003). Untreated roads and skid trails were evaluated in the field, using 2-m LiDAR and 1975 air photo imagery to help locate the features. Stream and swale crossings, roads, landings, gullies, stream diversions (existing and potential), and landslide locations were mapped. In addition, the field crew attempted to remeasure voids from historic slides. At stream and swale crossings, erosion voids were measured to an accuracy of 20 percent, and erosion potential, the amount expected to erode over the next 3 decades, was estimated based on drainage area, vegetation, and bank condition.

Erosion measurements along the haul road decommissioned in 1998 (Keppeler and others 2007) were repeated in 2011. The field crew evaluated 35 treatment sites for “sediment delivery potential” and gully “activity level” (Merrill 2003). At 10 sites, thalweg profiles and three to five cross-section transects were surveyed using control points established in 1999 to estimate erosion in the intervening period.

Results

Regression results confirmed the strong correlation between annual suspended sediment load and peakflow (r^2 = 0.90), allowing prediction of SFC loads independently of NFC measurements. SFC suspended sediment loads averaged 44 t/km^2 per year during the 1962 to 1967 calibration period while NFC averaged 56 t/km^2 per year (68 ± 40 (0.95 CI) for 22 pre-treatment years). This revised analysis, correcting for sampler bias, resulted in an estimate of excess suspended sediment produced post-logging of 928 t/km^2 (116 t/km^2 per year) through 1979 (*fig. 2*).

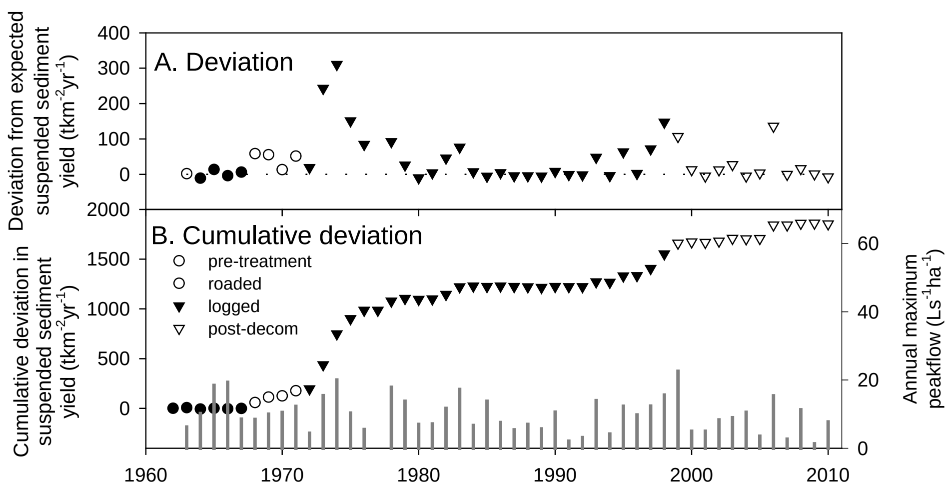

Figure 2—Excess suspended sediment production by year for (A) annual and (B) cumulative deviation from predicted load.

Beginning about 10 years after timber harvest, sediment production returned to pre-harvest levels, and this pattern persisted for another decade. Elevated sediment production was renewed in 1993 and was most evident during years with peak flows larger than the 4 year return interval flow (about 12 L/s per ha). Keppeler and Lewis (2007) detected SFC storm load increases of about 40 percent during 1998 to 2003, suggesting an episode of elevated sedimentation spanning the decommissioning treatment. Regression analyses of SFC storm loads on storm peaks for 1986 to 1998 and 2000 to 2009 show no significant difference ($p < 0.05$) after decommissioning for peaks larger than the 0.4 yr recurrence interval, but for smaller peaks, loads were significantly reduced (*fig. 3*).

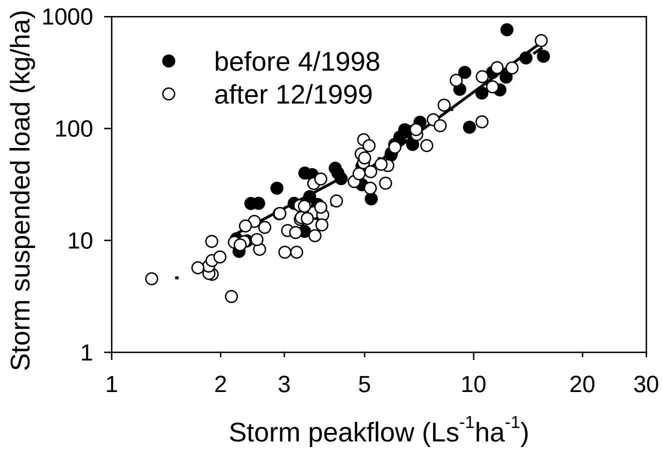

Figure 3—Sediment loads as a function of storm peak before and after decommissioning, divided into two peak discharge classes at the 0.4 yr return period.

Since implementation of continuous turbidity monitoring in 1996, SFC turbidities have exceeded ecosystem thresholds of concern more commonly than NFC turbidities. When turbidity records were tallied by stream ecosystem stress thresholds exceeding 25 (moderate), 50 (severe), 100 ntu (extreme) (Klein and others 2008) for each hydrologic year, SFC averages exceeded NFC's by 50, 60, and 120 percent ($p < 0.014$), respectively—a fairly consistent pattern even during years when NFC logging effects were evident. This result may reflect high inputs of fine sediments from road surfaces and bare ground in the South Fork, or may indicate that the physical properties of the sediment transported through the two weir ponds differ. The difference in ratios of average storm load above and below the two weirs supports the latter hypothesis. The weirs trap an estimated 20 to 30 percent of suspended sediment and virtually all bedload, thus the rationale for siting sampling stations ARF and QUE immediately upstream of the NFC and SFC weir ponds. Based on storm load data from 2001 to 2009, NFC transported just 70 percent of the load entering the pond, while SFC transported 80 percent.

Information from field-based sediment source assessments provides a context for interpreting the patterns of suspended sediment yield (*table 1*). The inventory of the entire South Fork watershed performed between 2004 and 2006 documented 443 remaining stream and swale crossings on untreated roads and skid trails. Of these, 325 had experienced partial failure. Swale crossings numbered 208, including 102 that had remained stable in the 3 decades since construction and only four that had eroded more than 10 m^3. In contrast, only 16 of 235 stream crossing had experienced no notable erosion. Eroded volume per stream crossing averaged 10 m^3, but most of this erosion did not appear to be recent. Fill material at risk of further erosion accounted for 16 percent of the estimated initial crossing fill. Stream crossings accounted for 82 percent of the "at risk" projection of erosion potential. Diversions onto roads and skid trails were common. Roughly 10 percent of stream crossings were active diversions while a similar number were "at risk" because a minor blockage could redirect flow onto the road surface. More than 300 m of entrenched diversions and active gullies were noted.

Table 1—*Field-based sediment source measurements (in-channel erosion from Reid and others 2010).*

Type	Years	m^3	m^3/yr	Method
Logging period				
landslides (re-assessed)	1971-76	7944	1324	inventory 2004-2006
Recovery period				
untreated crossing erosion	1971-06	2827	79	inventory 2004-2006
gully erosion	1971-06	932	26	inventory 2004-2006
mass-wasting > 7.6 m^3	1977-94	1705	95	inventory 1994
mass-wasting > 7.6 m^3	1977-06	2897	97	inventory 2004-2006
mass-wasting > 7.6 m^3	1995-99	4008	802	inventory, irregular
Post-decommission period				
mass-wasting > 7.6 m^3	2001-10	553	55	annual inventory
treated crossing erosion	1999	651	651	survey, 1999
treated crossing erosion	2000-01	108	54	survey, 2001-2002
treated crossing erosion	2002-11	150	15	survey, 2011
in-channel erosion	2001-08	na	291	survey, irregular

Landslide voids encountered during the recent inventory included 7944 m^3 (18.7 m^3/ha) from slides listed in the 1976 data set and 2897 m^3 (6.8 m^3/ha) in more recent features. However, half the slides in the 1976 data set were not relocated.

Soil displacement by mass-wasting for events larger than 7.6 m^3 declined since the 1970s and again in the most recent decade relative to the 1990s. Through 1994, the rate was 95 m^3/yr; from 1995 to 1999, 802 m^3/yr; and since 2000, 55 m^3/yr. Seven of 10 slides in the 1994 inventory were along haul roads and landings, but these accounted for less than 20 percent of the total volume. Only two failures along the mainline road were attributed to 1993. Between 1993 and 1999, 17 slides accounting for 70 percent of the mass-wasting volume occurred along the mainline road—15 resulted from culvert or fill failures. In the post-decommissioning data set, the largest feature was a 255 m^3 slide that originated along an untreated mid-slope road routing entrenched flow from a stream diversion to the fill slope. The slide traveled down the tributary channel as a debris torrent, depositing debris in the 1998 restored stream crossing void and continuing on to the South Fork mainstem.

Keppeler et al. (2007) report that the 1998 road decommissioning, followed by the largest peakflow of the 49 year SFC record, profoundly impacted sediment production in 1999. Re-evaluation of 35 treated crossings in 2011 indicated that erosion is ongoing at 31. Sediment transport potential was rated “high” or “extreme” at 12 of 37 sites. The predominant “Activity Class” (Merrill 2003) for the gullied crossings was “3” (indicating “good vegetative cover” with less than 50 percent of the incised area subject to erosion and transport). However, five were characterized as “1” (exhibiting “widespread transport and little or no vegetation”) and five were rated as “4” (supporting “good vegetative cover” and having gentle side-slopes).

The 2011 re-survey of channel cross-sections and profiles established at 10 sites in 1999 quantified additional scour and fill since 2001. Incision and widening were evident at 60 and 40 percent, respectively, of 42 cross-section transects, while 30 percent of transects showed neither. Mean profile elevation decreased by 0.09 m.

Headcut retreat was evident along all 10 surveyed profiles. In sum, these 10 sites enlarged by 80 m^3, or about 20 percent, since last assessed. Assuming a similar rate of erosion from the 25 sites that were not re-surveyed, total enlargement is estimated to have been about 150 m^3 over the last decade. A foot trail along the former road surface receives semi-regular (unsanctioned) bicycle and motorcycle use. The road has not fully revegetated and delivers an unknown amount of sediment to the stream.

Discussion

To relate erosion measurements to recent trends in suspended sediment yields at the weirs, it is helpful to compare unit-area sediment loads measured immediately upstream of the weirs with those measured at upstream tributary gages to identify areas contributing disproportionate amounts of sediment. Since 2000, when nine tributary gages were installed in the South Fork watershed, NFC event-based storm loads averaged 104.3 kg/ha based on contributions from the two gages delivering directly to the weir pond. Upstream gages produced similar loads, averaging 99.3 kg/ha. In contrast, South Fork event-based loads (the sum of mainstem loads measured 50 m upstream of the weir pond and the tributary gage load delivered directly to the weir pond) averaged 75.8 kg/ha. South Fork tributaries draining 52 percent of the watershed area produced 54.1 kg/ha per event, on average. Assuming the few small ungaged tributaries are not generating disproportionately high amounts of sediment, the SFC load is enriched below tributary gages and along the mainstem. Three of the tributary gages are sited within 42 m upstream of stream crossings restored during the 1998 decommissioning of the mainline road, and paired water samples were collected above and below the crossings during water years 2004 and 2006. These samples indicate suspended sediment concentrations were enriched below the decommissioned crossings ($p < 0.012$), reflecting the net contributions of erosion within the restored stream crossings and from the old road surface.

During the last decade, the volume of fine sediments in pools declined along the lower 490 m of mainstem channel above the SFC weir. However, pool depths decreased as well. Mainstem cross-sections, measured biennially since 2000 along 3100 m of channel, scoured 59 m^3 from 2000 to 2010[3]. It is likely that sediment inputs from both recent and historic mass-wasting, including materials eroded from failed and decommissioned crossings, were re-mobilized during high flows and augmented suspended sediment loads generated in upland portions of the South Fork watershed. In-channel erosion processes were also active in South Fork tributaries and account for a substantial portion of the suspended load at SFC (Reid et al. 2010).

Conclusions

Measured rates of suspended sediment yield in the South Fork Caspar Creek watershed were highest during the first decade after selection harvest and tractor-yarding was completed in 1973. A second episode of increased suspended sediment yield, coinciding with a notable increase in road-related mass-wasting, commenced 20 years post-harvest. Since decommissioning of the riparian road in 1998, reduced sediment yields have been detected only for small storms, though the incidence and

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magnitude of mass-wasting was lower during the last decade than in the 1990s. Mainstem channel measurements and tributary gage data suggest that sediment deposited prior to 2000 was mobilized and transported during the last decade. Erosion in upland areas, including diversion-induced mass-wasting, incision of crossings along untreated roads and skid trails, and in-channel gullying, remains active. Potential interactions with persisting legacy effects should be considered when modern forest management practices are superimposed on landscapes still responding to past disturbances.

Acknowledgments

This study is part of the cooperative Caspar Creek research program instituted in 1960 by the U.S. Department of Agriculture, Forest Service Pacific Southwest Research Station and the California Department of Forestry and Fire Protection. Leslie Reid and Jack Lewis provided critical components of this analysis.

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Declining Sediment Loads from Redwood Creek and the Klamath River, North Coastal California

Randy D. Klein[1] and Jeffrey K. Anderson[2]

Abstract

River basin sediment loads are affected by several factors, with flood magnitude and watershed erosional stability playing dominant and dynamic roles. Long-term average sediment loads for northern California river basins have been computed by several researchers by several methods. However, characterizing the dynamic nature of climate and watershed stability requires computation of annual loads. We computed annual suspended and bedload loads for the 1950s through 2009 for both Redwood Creek and the Klamath River. Results show high sediment loads coincident with a period of widespread logging by destructive practices and large storms in the 1950s through 1970s followed by a dramatic decline in sediment loads through the present. Analyses of annual departures from mean and time trend tests indicated the decline in loads is not due solely to the lack of very large storms. We infer it can also be explained by the partial recovery of watershed erosional stability from the 1980s through the present due to reduced logging rate, use of lower impact logging practices, and implementation of treatment programs for reducing erosional threats from logging roads.

Key words: bedload, suspended load, watershed recovery, Redwood Creek, Klamath River

Introduction

Sediment yield, or load, consists of the total mass of sediment particles transported by streamflow past a location along a stream or river over a given time period. Sediment is transported either suspended in the water column (suspended load) or along the channel bed (bedload). Quantification of sediment loads can be important for a variety of management needs, such as evaluating land use impacts, reservoir design, coastal sediment budgets, and others. In oceanographic studies, the long term total or average annual sediment yield may suffice for correlation with sediment accretion on the continental shelf (Farnsworth and Warrick 2007, Willis and Griggs 2003), but for assessing effects of changing watershed conditions, annual time series sediment load estimates are required. The objective of this analysis was to characterize fluctuations in sediment loads for the period of record for Redwood Creek and the Klamath River and to relate changes in annual loads to watershed conditions affecting loads.

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Study area

The Klamath River and Redwood Creek drain lands in northwestern California, with the Klamath basin extending into eastern Oregon (*fig. 1*). The Redwood Creek basin area of 717 km^2 lies entirely within the Coast Range of California and is underlain by metamorphic and sedimentary rocks of the Franciscan assemblage (Nolan et al. 1995), relatively weak rocks subject to high erosion rates. Redwood Creek's Mediterranean climate is dominated by rainstorms rather than snowmelt runoff, with most of the average annual rainfall depth of about 150 cm falling during winter months (November through March). Occasional intense rainfall events produce large stormflow peaks at the basin outlet near Orick, California. Harden (1995) reported peak discharges in the 1950s through 1970s ranging from about 1300 to 1400 m^3s^{-1}, with recurrence intervals of approximately 25 years.

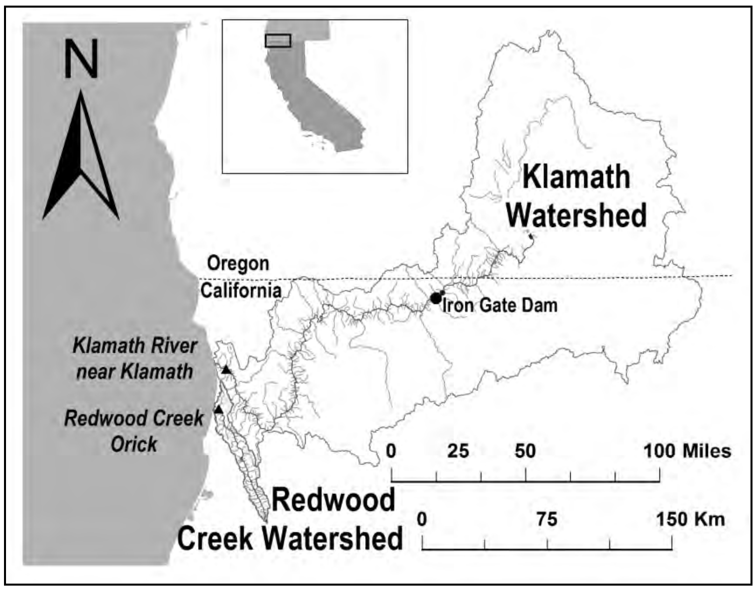

Figure 1—Location of the Redwood Creek and Klamath River watersheds, northern California and southern Oregon. Gaging stations near watershed outlets and location of Iron Gate Dam are also shown.

Due to its much larger size (40,720 km^2), the Klamath River basin contains broader ranges of both climates and lithologies than Redwood Creek. Ayers Associates (1999) report that mean annual precipitation ranges from about 280 cm in the upper reaches of tributaries near the coast to as little as 25 cm in the upper watershed near Tule Lake. Large floods are caused by both intense rainfall and rain on snow (Ayers Associates 1999) with several large storms having occurred over the past 50+ years. In the simplest terms, the basin can be subdivided geomorphically into the upper basin (above the Shasta River) and the lower basin based on bedrock geology and potential sediment delivery to streams. Hillslopes of the upper basin have generally lower sediment delivery potential due to more competent terrane and lower relief than the lower basin. Not far upstream of Cottonwood Creek, Iron Gate Dam traps much of the sediment load generated from the upper watershed. The Trinity River, the largest Klamath River tributary, is notable for its historically high sediment delivery to the lower river due, in part, to the presence of highly weathered granitic rocks (US Bureau of Land Management 1995). Elsewhere in the lower basin the Coast Range geology, similar to that of Redwood Creek, also creates high sediment delivery potential and sensitivity to land use disturbances.

The Redwood Creek watershed has been studied extensively since the 1970s, owing in part to the creation of Redwood National Park in 1968 and expansion of the park in 1978 due to concerns over erosion and sediment delivery originating on adjacent private timberlands (Nolan and Janda 1995). Stream gaging stations were established at key locations within the basin where water discharge and, for much of the period of record, suspended sediment and bedload transport rates were measured. Large increases in sediment yields due to widespread clearcut logging and tractor yarding were documented (Nolan and Janda 1995). The Klamath River has also had stream gaging stations operated on it for some time, several of which included periodic suspended sediment sampling. Together, water discharge and sediment transport samples at known discharges provide the information for making sediment load computations.

Best (1995) documents the post-World War II acceleration of logging in Redwood Creek when few regulations were in place to moderate effects on erosion and sediment delivery. Ayers Associates (1999) indicate that acceleration of logging began in the Klamath’s lower basin at about the same time as in Redwood Creek. With passage of the Z’berg-Nejedly Forest Practice Act of 1973 the size of individual harvest areas in California was reduced along with other rules to limit damage to hillslopes and streams. In Redwood Creek logging rates fell abruptly after park expansion in 1978 excluded the downstream one-third of the watershed from commercial timber harvest.

The response to harsh land use and large floods in Northern California’s redwood country in the 1960s through the 1970s caused persistent changes to hillslopes and channels. Madej and Ozaki (2009) document the export of channel-stored bed sediment in Redwood Creek from the 1970s through the present, with channel recovery process proceeding in a downstream direction. Recovery rate toward pre-aggradation channel bed elevations was dramatically slower in the gentler-gradient downstream reaches. Payne and Associates (1989) documented tributary delta growth in the lower Klamath River, indicative of elevated sediment delivery from these sub-basins reaching a maximum in the 1970s and persisting through the time of their analysis. Tributaries to the lower Klamath remain heavily aggraded, attesting to a high degree of prior land disturbance and persistent channel bed aggradation similar to that of Redwood Creek.

Methods

Annual sediment loads were computed for water years (WY) 1954 to 2009 for Redwood Creek at Orick (No. 11482500) and for WY1951 to 2009 for the Klamath River near Klamath (No. 11530500). The U.S. Geological Survey (USGS) has operated these stream gaging stations near the mouths of both rivers since the early 1950s and collected sediment samples for a portion of that time. Redwood Creek sediment data included both suspended sediment and bedload samples, but the Klamath dataset included only suspended sediment.

Suspended sediment

Redwood Creek suspended sediment sampling occurred from 1971 to 2009 during which 380 samples were collected. Klamath River suspended sediment sampling occurred from 1974 to 1995 with a total of 270 samples collected, of which 158 were used for estimating suspended load. To compute annual suspended sediment loads, we used the USGS program “LOADEST” (Runkel et al. 2004), a rating curve approach based on a multi-parameter log-linear model that accounts for the effects of discharge, time and seasonality on suspended sediment concentration (SSC). LOADEST contains a number of predefined log-linear models, and the best fit model based on the internally calculated Akaike Information Criterion is generally recommended. LOADEST automatically adjusts for log-space retransformation bias for determining average load estimates (Runkel et al. 2004). Review of log flow-adjusted SSC (as described later) showed a nonlinear trend with time for both rivers indicating potential SSC rating curve shifts. To overcome the nonlinearity we employed multiple log-linear models at obvious shifts.

Separate log-linear models were used on Redwood Creek for the periods of WY 1954 to 1970, 1971 to 1977 and 1978 to2009, and on the Klamath River for WY 1951 to 1964, 1965 to 1977 and 1978 to 2009. Load estimates for periods preceding data collection were based on a single-variable log-linear model of discharge using all available data. To improve the log-linear rating curves and better constrain Klamath River high flow load estimates for the pre-sampling (1951 to 1974) period, high flow data between 1957 and 1977 were used from two upstream stations composing 94 percent of the basin area draining to the lower Klamath: Klamath River at Orleans (No. 11523000) and Trinity River at Hoopa (No. 11530000). Mass balanced SSC values for the upstream stations and the corresponding mean daily flows for discharges greater than 1,800 $m^3 s^{-1}$ for the Klamath River near Klamath were then used in the Klamath River log-linear rating curves for the two periods. For the Klamath River, the selected multi-variable log-linear model was used beyond the data sampling period for the 1996 to 2009 load estimates. All selected log-linear models showed good fits to the data with coefficient of determination (R^2) values ranging from 89 to 99 percent.

Bedload

Annual bedload loads were determined for Redwood Creek and Klamath River using a recently published bedload transport formula (Recking 2010), which uses discharge, active channel width, slope, and surface particle size (D_{50} and D_{84}) as inputs. In general, loads were estimated by first developing log-linear rating curves using available streamflow measurement records, slope and surface particle sizes for each river with the Recking (2010) bedload formula. The rating curves were applied to the mean daily flow record and then summed by WY to produce annual bedload loads. We corrected for log-space bias by applying the correction factor exp ($s^2/2$) (Ferguson 1986) to estimated annual loads, where s is the standard error of a log-linear model of sampled bedload and discharge in Redwood Creek (correction factor = 1.29).

For Redwood Creek most of the necessary data were available for most of the period of record, with the exception of bed surface particle sizes prior to 2010. The USGS sampled bedload (150 samples) in Redwood Creek from 1972 to 2009, which included bedload particle size fractions. To estimate surface D_{50} and D_{84}, a relationship was developed between measured surface and bedload particle sizes. Adequate bedload particle size measurements were available to estimate changes in median surface D_{50} and D_{84} by decade. Although the ratios may have changed over smaller time scales, we felt this was a reasonable method for conditioning bed surface grain sizes through time, as suggested by comparison with USGS measured daily bedload rates for Redwood Creek: 96 percent of the predicted rates were within one order of magnitude of USGS measurements for the overlapping period (1971 to 1992).

Recking's (2010) model was also used to determine bedload annual loads for the Klamath River. Unfortunately, no data on bedload samples, channel bed material particle sizes or slope were available for the Klamath station, so field data were collected for this project in 2010. Particle sizes were determined by performing pebble counts (Wolman 1954) on a gravel bar near the gaging station and surveying the adjacent water surface slope. Lacking any prior data, we had to use the 2010 pebble count and slope data to estimate bedload loads for the entire period of record.

Time trends

Temporal trends in suspended sediment for the Klamath River and Redwood Creek were evaluated by two methods: 1) comparison of departures from period means of annual suspended sediment loads and peak flows, and 2) trends in SSC using the nonparametric Seasonal Kendall test (Helsel and Hirsh 2002). This test adjusts for seasonal variability by combining individual Mann-Kendall trend tests to seasonally grouped SSC. To increase the power of the Seasonal Kendall test, SSC data were adjusted for streamflow effects (flow-adjusted SSC) using a LOWESS smooth curve between SSC and streamflow (Helsel and Hirsh 2002). The residuals from the LOWESS curve were then used in the Seasonal Kendall test. Based on recommendations in Schertz et al. (1991) and the frequency of available SSC data, the Seasonal Kendall test was conducted on six seasons for the Klamath River, and three seasons for Redwood Creek. Monotonic trends in SSC are considered significant at the alpha < 0.05 probability level (p).

Results

The history of sediment loads delivered to the Pacific Ocean by Redwood Creek and the Klamath River is characterized by large inter-annual and decadal variability (*figs. 2A* and *2B*). Loads were highest during the largest storms of the 1950s through mid-1970s and have declined dramatically since.

As expected, annual average total suspended sediment loads for the periods were vastly different between the two rivers given the differences in watershed area. At 6,831,000 Mg yr^{-1}, the much larger Klamath River produced about seven times more suspended sediment over the long term (1950s to 2009) than Redwood Creek, which averaged 975,000 Mg yr^{-1}. However, the situation reverses when considered on an area-weighted basis: Redwood Creek produced about seven times more suspended sediment per unit area than the Klamath River.

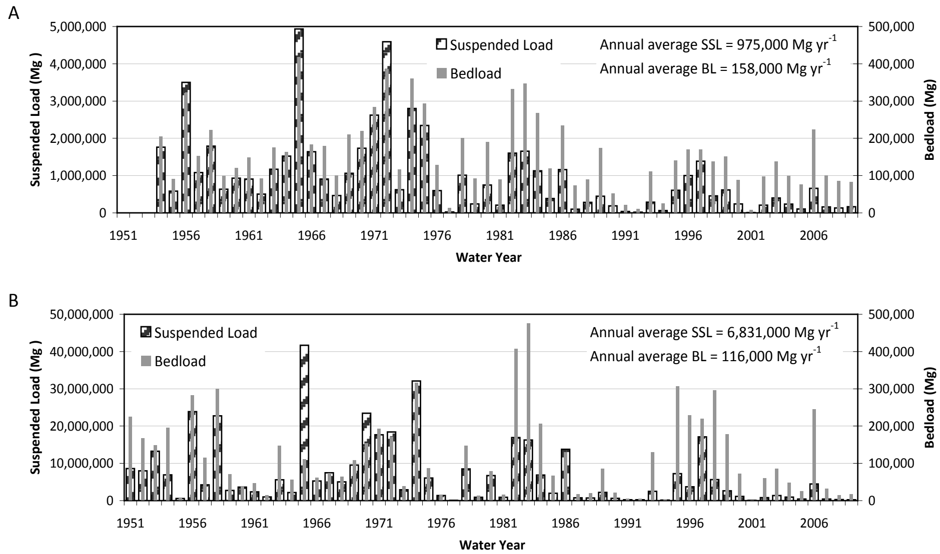

Figure 2—Annual suspended and bedload sediment loads for Redwood Creek (A) and Klamath River (B), 1951 to 2009. Note the difference in scales for suspended loads.

Unlike suspended loads, differences in bedload sediment loads were relatively small between the two rivers: 158,000 Mg yr^{-1} for Redwood Creek and 116,000 Mg yr^{-1} for the Klamath. Closure of Iron Gate Dam in 1961 on the Klamath River approximately 306 km upstream from the mouth (see *fig. 1*) reduced sediment fluxes from the upper one-third of the watershed. However, because area-weighted sediment delivery in the basin area upstream of the dam is much lower than that downstream (Ayers Associates 1999), reductions in basin yields to the Pacific Ocean would be less than the proportion of watershed area truncated by the dam. We note that bedload load estimates for the Klamath are based on a very limited data set for model inputs

Figures 3A and *3B* show departures in annual maximum instantaneous peak discharge and suspended load from their respective period means and trendlines for each. Large departures coincided with years of large sediment loads and floods. Declining trends since about 1975 characterize both variables for both rivers, but the trendline slopes of suspended load are steeper than those of peak discharge for both rivers, indicating loads have declined to a greater degree than peak flows. For Redwood Creek, the suspended sediment trendline is about three times steeper than that for peak discharge, and four times steeper for the Klamath River. For both rivers, the largest storms of recent decades (in WY1996 and WY1997 of roughly 10- and 15-years recurrence intervals, respectively) produced far less sediment than would be expected with similar sized stormflows before 1975.

Results from the Seasonal Kendall test (*fig. 4*) indicate that SSC significantly declined from 1975 to 1995 for the Klamath River ($p < 0.0001$), and from 1971 to 2009 for Redwood Creek ($p < 0.0001$). Trend results indicate a 5.9 mg L^{-1} yr^{-1} (0.7 percent yr^{-1}) decrease in SSC for the Klamath River, and a 3.5 mg L^{-1} yr^{-1} (4.4 percent yr^{-1}) decrease in SSC for Redwood Creek.

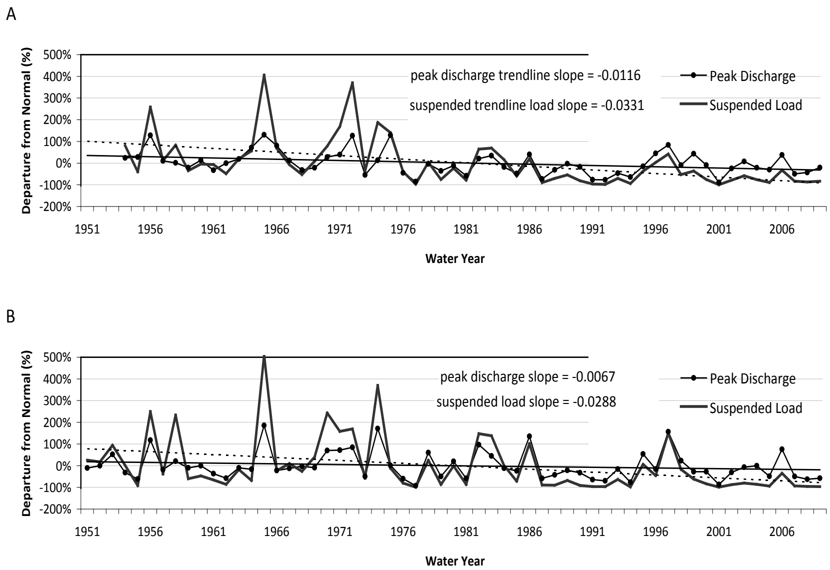

Figure 3—Suspended sediment load and annual peak discharge departures from normal for Redwood Creek (A) and Klamath River (B). Dashed line is trendline for suspended sediment load, solid line is trendline for peak discharge.

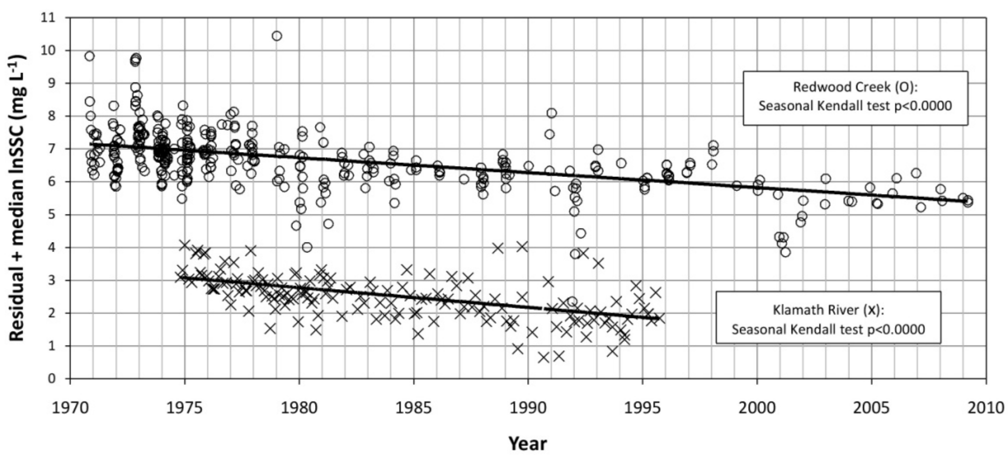

Figure 4—Suspended sediment concentration (SSC) residuals with flow effects removed and Seasonal Kendall test results.

Accompanying the reduction in loads, the proportion of bedload to suspended load has shifted in Redwood Creek and the Klamath River. Prior to 1976, the ratio of bedload to suspended load for Redwood Creek was 0.12, rising to over twice that (0.26) for the later period. The proportion of bedload to suspended load was much smaller for the Klamath River: the pre-1976 proportion was 0.012 and rose to 0.029 for the 1976 to 2009 period, although this should be viewed with caution considering that historical Klamath bedload loads were computed using model inputs collected only in 2010. This results in lower confidence in computed bedload loads with increasing time before present, although we believe this method to be superior to bedload computed as an assumed percentage of suspended or total load as others have done (e.g., Willis and Griggs 2003).

Discussion

Computing historic annual sediment loads necessitated the use of several methods and assumptions, however, use of a single sediment rating curve spanning decades, as done by other researchers (e.g., Farnsworth and Warrick 2007, Wheatcroft and Sommerfield 2005), cannot account for shifting relationships between water discharge and sediment transport that occur due to changing watershed conditions. Both the departures from normal analysis and the Seasonal Kendall trend tests indicated sediment loads are declining independent of lower peak flows. These declines highlight the importance of assessing temporal trends in sediment loads as opposed to average annual or period total loads. As watersheds become erosionally destabilized, sediment loads rise relative to flows, and as recovery to more stable conditions proceeds, loads decline relative to flows. Accordingly, sediment rating curves can be expected to shift upward or downward through time with the watershed disturbance regime.

Although no definitive causal link has been established, reduction in sediment loads since the mid-1970s probably can be attributed to natural watershed recovery processes, strengthened land use regulations, and watershed restoration programs. Watershed recovery processes such as vegetation re-growth and export of channel stored sediment as bedload have occurred throughout our study area even with ongoing timber harvest. Timber harvesting regulations that went into effect around 1980 in California, and have been continually refined since, reduced the allowable size of harvest units, elevated road construction standards, and extended riparian buffer requirements, all of which tend to reduce erosion and sediment delivery relative to earlier, more destructive practices.

In accordance Redwood National Park expansion legislation in 1978, a restoration program was initiated on Redwood Creek parklands focusing on the 700 km of logging roads that existed in 1978. To date, about 480 km (69 percent) of logging roads have been decommissioned (obliterated) such that their potential to deliver sediment has been greatly reduced. Road decommissioning has also occurred elsewhere in Redwood Creek on private timberlands, along with upgrading of logging roads to reduce their likelihood of sediment delivery. This type of work is also occurring in the lower Klamath River basin, but to a lesser degree than in Redwood Creek.

It remains to be seen how Redwood Creek and the Klamath River will respond to the inevitable recurrence of very large floods of magnitudes similar to those of 1950s through 1970s. However, our analyses suggest that the changing watershed conditions described above have provided the basins with a higher degree of resilience than that which led to the highly destructive erosion and sediment delivery events of the 1950s to 1970s.

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The VTAC Committee: Developing Guidance for an Alternative Regulatory Pathway to the Anadromous Salmonid Protection Rules[1]

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Abstract

In recent decades, riparian protection standards have been guided by generalized prescriptive rules. With the passage of the Anadromous Salmonid Protection rules in 2009, the Board of Forestry and Fire Protection (Board) established a regulatory pathway that provides an alternative approach for riparian protection based on site-specific criteria (14 CCR § 916.9 [936.9, 956.9](v)). This new pathway seeks to promote more immediate (short-term) responses to active riparian management practices that might not otherwise occur under the more prescriptive rule protocols. This approach requires consideration of both watershed-scale limiting factors (in other words, context assessment) and site-based factors to lead to a modified riparian management design that provides benefits to the aquatic environment. It is the Board's intent that allowing site-specific plans will create an economic incentive for landowners to engage in active management and restoration activities in riparian areas. The implementation of this new approach is being overseen by the Anadromous Salmonid Protection Rule Section V Technical Advisory Committee (VTAC), composed of members from academia, the timber industry, professional consulting firms, and the public. The California Department of Forestry and Fire Protection (CAL FIRE) appointed this committee in October 2010. The VTAC is seeking to establish principles, guidelines, and procedures to guide landowners in the use of this new rule section. The VTAC is focusing on: (a) broadening incentives, and (b) developing permitting efficiencies that properly balance the risks of negative impacts with the potential benefits to listed salmonid species. In short, one goal of the VTAC is to reduce the regulatory barriers that might otherwise prevent landowners from engaging in active management and restorative actions in riparian areas.

The VTAC will use multiple pilot projects identified by landowners in both the Coast Ranges of California and the interior part of the state to demonstrate active riparian management, with potential implementation in 2011. These pilot projects include a range of desired objectives, including increasing large wood loading, promoting increased biotic diversity, reducing catastrophic wildfire risk, and accelerating conifer tree growth.

Key words: anadromous, buffer, riparian, restoration, salmonid

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[1] This paper was presented at the redwood science symposium: coast redwood forests in a changing California. June 21-23, 2011. Santa Cruz, California.

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Introduction

Over the past 4 decades, there have been several refinements and additions to riparian protection rules originally established by the Z'Berg-Nejedly Forest Practice Act of 1973. Similar processes have occurred in Oregon, Washington, Alaska, and on federal timberlands (Everest and Reeves 2007). Common to most of these riparian protection standards is the use of prescriptive rules for longitudinally continuous buffer strips. These buffer strips require mandated widths and restricted harvesting practices, such as a minimum basal area or canopy cover, to remain following harvest for specific classes of stream. Such standards typically result in relatively passive, continuous, uniform, one-size-fits-all streamside buffers.

Recently in California, the California State Board of Forestry and Fire Protection (Board) developed Anadromous Salmonid Protection (ASP) rules that seek to protect, maintain, and improve riparian habitats for state and federally listed anadromous salmonids. The Board adopted an option (14 CCR § 916.9 [936.9, 956.9] Section (v)), referenced here as the Section V Rule, that would support more site-specific decision making in the design of riparian prescriptions that could be applied during the time of adjacent timber harvest. This approach offers an alternative to prescriptive uniform buffers and may be more protective of ecological functions (Liquori and others 2008) (*fig. 1*). The intention of this new rule is to promote a more explicit riparian design process that addresses specific ecological and geomorphic functions and processes specific to the project site. In theory, more site-based riparian treatments could better maintain, improve, restore and/or expedite those ecological

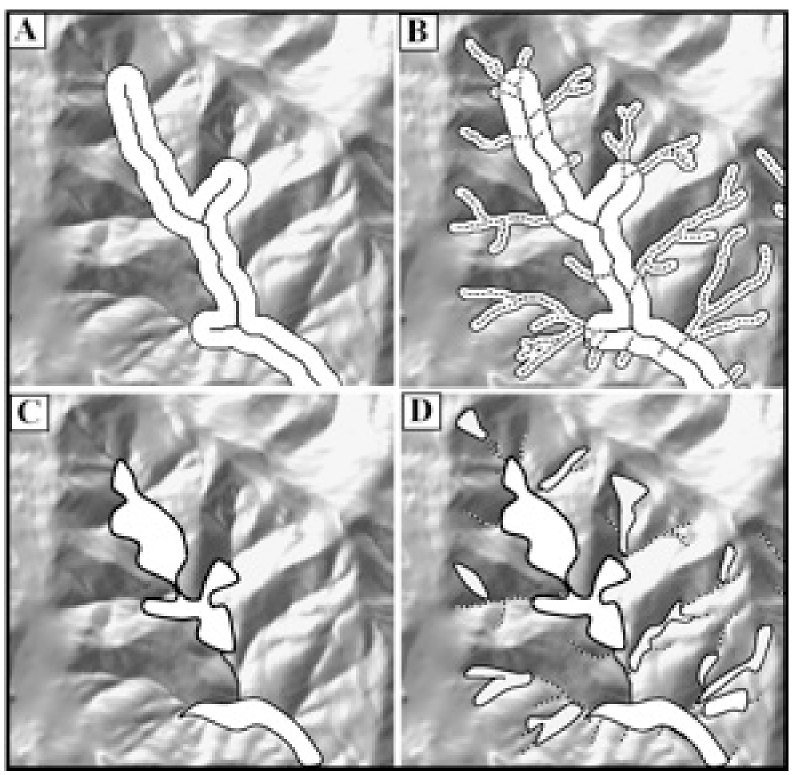

Figure 1—Uniform stream buffer protection measures (A and B) v. spatially variable prescriptions (C and D) (image from Liquori et al. (2008) (used with permission).

processes most important to sustaining resilient salmonid habitat conditions throughout forested landscapes (Benda et al. in press, Liquori et al. 2008, Ryan and Calhoun 2010).

The Section V Rule allows alternative treatments to the riparian zone with the caveat that the effects of the proposed practices on riparian functions must be at least equal to those expected from the revised prescriptive standards that were adopted. This rule seeks to promote both immediate (short-term) responses to active riparian management practices that might not otherwise occur under the more prescriptive rule protocols, as well as longer-term habitat improvements (Liquori et al. 2008). This approach requires consideration of both watershed-scale limiting factors (in other words, context assessment) and site-based factors to lead to a modified riparian management design that provides benefits to the aquatic environment. It was also the Board's intent that allowing site-specific plans will create an economic incentive for landowners to engage in active management and restoration activities in riparian areas. Thus, it will allow for economic incentives to encourage restoration.

The Section V Technical Advisory Committee (VTAC) was established by the Board to develop a guidance document that will allow broad application of the site-specific approach for riparian management and implementation of at least two pilot projects. The VTAC is composed of members from academia, the timber industry, professional consulting firms, the public, and includes state and federal agency representatives. The VTAC is seeking to establish principles, guidelines, and procedures to guide landowners in the use of this new rule section. It is focusing on: (a) broadening incentives, and (b) developing permitting efficiencies for measures that are necessary for recovery of listed salmonid species. One important goal of the VTAC is to reduce the regulatory uncertainty that might otherwise prevent landowners from engaging in active management and restorative actions in riparian areas.

VTAC outreach survey genesis and summary

The VTAC developed an online survey to improve acceptance of the adaptive rule described by the ASP Rule 916.9 Section V and to gain a better perspective on regulatory requirements that the landowner may face when trying to implement a Section V plan. The survey was distributed to landowners, Registered Professional Foresters (RPFs), agency personnel, and the public in spring 2011 (see: http://calfirevtac.weebly.com). A brief introductory video, as well as background information on the rule section, was provided on this website. Information from the survey responses is helping the VTAC develop approaches that will be used for the guidance document being produced for this new approach to riparian forest management.

The VTAC received 123 responses to the survey. Landowners and land managers supplied 39 percent of the sample, while 32 percent were agency staff, 19 percent were consultants, 6 percent were from the general public, and 4 percent represented advocacy groups. Approximately half of the respondents are California RPFs. Roughly 70 percent of respondents had either detailed or moderate knowledge of the ASP Section V rule prior to taking the survey and 87 percent of the respondents were either very or somewhat knowledgeable of the California Forest Practice Rules.

Eighty percent of respondents said they favor more flexible site-based riparian protection zone treatments that are professionally designed over a broadly prescriptive rule. Seventy-three percent agreed that treatments should be well-grounded in science. Nearly 80 percent responded that they are likely to support a landowner's ability to apply site-based riparian treatment through Section V rules if it is technically justified.

Respondents informed the VTAC that their primary concern regarding the Section V rules is "too much uncertainty/inconsistency in interpretation." Approximately 50 percent felt that well-documented examples of success with regard to navigating the various regulatory agencies' requirements would improve their comfort level in the use of Section V projects. Thirty-seven percent of respondents said they are highly or moderately likely to submit a Section V project (15 percent did not reply to this question). Respondents stated that the primary reason why a landowner would be unwilling to submit a Section V project is "too much uncertainty in the process." Approximately 30 percent plan to submit or may submit a Timber Harvesting Plan (THP) in the near future with a Section V project.

The primary take-home messages from the survey are the following: (1) there is widespread agreement that site-based riparian management can be used where it is justified; (2) an increased level of certainty is required for extensive use of the Section V process; and (3) successful pilot projects are needed to demonstrate to landowners that this approach can work.

Potential types of section V site-specific projects

Site-specific riparian management projects that are justified will vary depending on current watershed and riparian conditions, geographic location, and geomorphic characteristics of the site. Desired objectives for riparian management include increasing large wood loading, promoting increased biotic diversity, increasing nutrient cycling and biotic qualities of the salmonid food-base, and reducing catastrophic wildfire risk. The following sections provide brief descriptions of selected references related to these topics, but is not meant to be a comprehensive review of the riparian science literature.

Large wood

Most of the anadromous fish-bearing watersheds located in the Coast Ranges of California are currently deficient in large wood loading due to removal of wood from the 1960s through the 1980s (NMFS 2010, Wooster and Hilton 2004) and removal of much of the riparian forest in harvests prior to 1973. In smaller coastal streams, large wood is usually the dominant structural agent, important in pool formation and a critical component for habitat complexity. Generating adequate volumes of large wood following removal is a slow process in second-growth coastal stands due to lack of adequate material and low levels of mortality in these long-lived species, typically requiring 75 to 150 years to reach acceptable levels (Wooster and Hilton 2004), and was exacerbated by excessive removal of large conifer trees in riparian zones in past decades.

Felling and/or excavating selected large conifer trees from the riparian zone near the fish-bearing streams may be appropriate to rapidly create deep pools in channels with the appropriate gradient and bankfull width that are currently lacking wood. Past monitoring work has shown that large wood placement projects can lead to higher densities of juvenile coho and other salmonids (Roni and Quinn 2001, Whiteway et al. 2010). Unanchored large wood has the highest likelihood of relative stability in smaller streams (for example, third order) when the bole length is at least two times bankfull channel width and the rootwad remains attached (Collins 2000, WFPB 2001).

Spence et al. (1996) state that for second-growth stands, thinning may be appropriate in order to facilitate recovery and protection of key functions, particularly in coastal forests. Thinning can accelerate riparian conifer tree growth in areas where uniformly young stands exist. In these stands, there are limited large conifers for future recruitment to the channel. Additionally, the stream ecosystem is likely to be temperature limited and lack large tree habitat characteristics. Very high tree densities can preclude the development of large trees for decades in coast redwood stands (Thornburgh et al. 2000). Ligon et al. (1999) concluded that to grow and maintain larger diameter conifer trees in riparian areas similar to the stand structure for those elements found in late-successional stands, "it may be necessary to manage these zones through thinnings and selection harvests to promote the growth of the larger trees present that have the best opportunity to maximize diameter and height growth."

In general, thinning has been found to be an effective means of enhancing old forest development in coast redwood forests by accelerating tree growth, modifying species composition, and increasing stand-level variability (O'Hara et al. 2010). Silvicultural approaches to improve tree growth include low thinning (in other words, thinning from below), commercial thinning (crown thinning), and variable density thinning (VDT) (Teraoka and Keyes 2011). Cafferata et al. (2005) reported that thinning from below produced more large trees in a coastal riparian flood prone area over a 60-year modeling period than standard single tree selection.[7] Low thinning in a coast redwood/Douglas fir stand has recently been reported as accelerating conifer growth but not promoting redwood dominance. This treatment allowed Douglas-fir to remain competitive in the upper canopy (Teraoka 2010, Teraoka and Keyes 2011).

Biotic diversity and nutrients

Increased biotic diversity and nutrient input into anadromous salmonid watersheds is increasingly being recognized as a key function produced from the riparian forests. Past studies have shown that a 30 m buffer with no tree removal reduces impacts to a stream that are similar to a no-harvest level (Newbold et al. 1980). However, completely excluding vegetation management in the buffer strip can limit opportunities to increase fish growth rate and biomass (CSBOF 2008). Several studies suggest selective thinning of the riparian canopy can result in an increase in aquatic macroinvertebrate production, thus raising the food availability for salmonids (e.g., Wilzbach et al. 2005). Conifer verses hardwood-dominated riparian stand types can produce differing stream temperatures and channel morphology due to differences in canopy conditions and root density (Liquori and Jackson 2001 (*fig. 2*). Past research suggests that at appropriate locations, active riparian management that promotes an appropriate mixture of conifers and hardwoods can enhance primary

[7] Thinning from below was defined as harvesting intermediates and co-dominants only and specifying that the quadratic mean diameter (QMD) of the stand must increase after harvest.

productivity and produce temperature regimes that promote fish production. The riparian stand characteristics most likely to achieve these functions include: (1) a sufficient number of nitrogen-fixing deciduous trees distributed at key locations within the stream network, and (2) a sufficient number of riparian canopy gaps that allow for sunlight to support macroinvertebrate production while balancing effects on other riparian functions (CSBOF 2008).

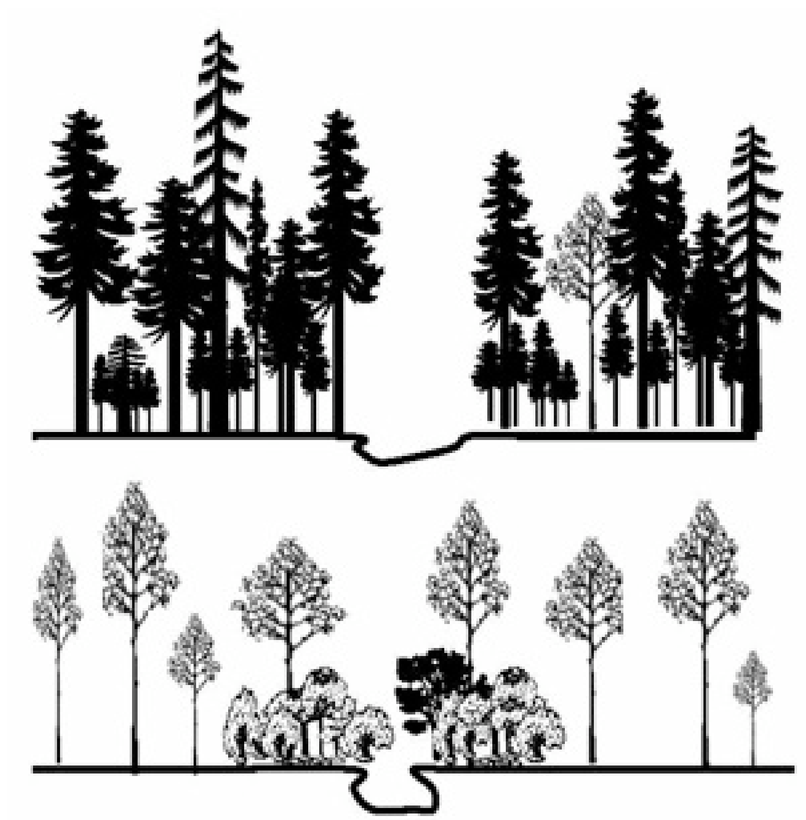

Figure 2—Diagram of a unevenaged conifer-dominated riparian stand (above) v. a hardwood-shrub riparian stand (below) from the east slope of the Cascade Mountains in Washington (image from Liquori and Jackson 2001). Note differences in stream shading and channel morphology for each stand type.

Fuel hazard reduction

Research has shown that fuel hazard reduction projects can reduce the risk of catastrophic crown wildfire (Martinson and Omi 2003, Omi and Martinson 2004). This concept directly applies to riparian zones, where active management can prevent dense stands of trees from contributing to rapid fire spread upslope, particularly in the interior parts of California with hotter and drier climates. Balancing the potential benefits of fuel hazard reduction with the possible risks to habitat recovery, however, will be complex and require careful consideration by both the project proponent and the reviewing agencies.

Fire behavior models have been used to show that some parts of the landscape are highly prone to catastrophic wildfire (in other words, a rapidly moving crown fire). This information can be used to justify prescriptions to reduce surface fuels, intermediate fuels, and co-dominate fuels (in other words, “ladder fuels”) in the riparian zone that differ from the standards in the ASP rules. Treatment could occur both in the Class I (fish-bearing) watercourse and lake protection zone (WLPZ) core, inner, and outer zones, as well on the hillslopes beyond the WLPZ, creating a landscape level fire hazard reduction project. Recent modeling work by Van de Water and North (in press) shows that Sierra Nevada riparian forests are significantly more fire prone under current management regimes with excluded harvest that allow for a build-up of fuels when compared to riparian areas that have been exposed to an active-fire regime. Additionally, under current conditions, riparian forests were found to be significantly more fire prone than upland forests. The 2007 Angora Fire at South Lake Tahoe provides an example of this situation. Murphy et al. (2007) reported that dense stands of trees in the Angora SEZ (Stream Environment Zone) likely contributed to the rapid fire spread to Angora Ridge by supporting crown fire runs upslope.

Framework for riparian design

Site-based riparian design should: (1) identify the relative importance of riparian function inputs (wood, heat, sediment, litter/invertebrates) for forming aquatic habitat and to affect water quality in the adjacent stream reach, and (2) assess the potential connectivity (in other words, transport) of riparian inputs to downstream channels. The sensitivity or response of aquatic resource condition to riparian inputs and the potential downstream connectivity are indicators of riparian function importance.

Channel response potential and downstream connectivity are strongly influenced by channel geomorphology (gradient, confinement, bed composition), stream size/flow, and position in the stream network. Although channel morphology and flow vary along the stream network, a stream may be subdivided into reaches with distinct channel forms that are reflective of watershed topography and channel forming processes (Montgomery and Buffington 1997, Paustian et al. 1992). Such geomorphic stream classification provides a qualitative tool for assessing response potential to changes in riparian function inputs (WFPB 1997). We adopt this approach by apply a typing system that combines stream size (based on channel width or regionally calibrated basin areas) with the Montgomery-Buffington classification (*table 1*). More rigorous evaluation of response potential can be facilitated by quantitative modeling of channel conditions and watershed processes (Benda et al. 2007).

VTAC guideline document and pilot projects

The VTAC is developing guidance documents that will: (1) identify qualifying criteria for suitability of sites for various treatment options, (2) help identify site-based objectives, (3) outline treatment option alternatives that are consistent with those objectives, (4) establish guidance for the data necessary to justify proposed actions, and (5) clarify administrative procedures for obtaining agency approvals. In preliminary “interim” guidance, the VTAC has established a simplified procedure for prioritizing among four objectives: wood loading, thermal regulation, nutrient cycling, and sediment.

VTAC pilot projects guidelines documents will allow RPFs to determine if site-specific riparian management is appropriate for a given location. Both a “default design process” using a structured classification system, and a more flexible

Table 1—*Preliminary channel sensitivity/response potential to changes in exchange function inputs in relation to stream size and channel type.*

		Function input (channel response metric)			
Channel type	**Stream size**	**Large wood (pool formation)**	**Shade (temperature)**	**Sediment (grain size)**	**Litter (retention)**
colluvial	small	M	H	M	H
bedrock	all	L	H	L	L
cascade	all	L	M	M	L
step pool	all	M	H	M	M
plane bed	all	H	H	H	L
pool riffle	small	H	H	H	H
pool riffle	medium	H	M	H	H
pool riffle	large	H	L	H	M
dune ripple	small	M	H	L	M
dune ripple	medium	M	M	L	L
dune ripple	large	L	L	L	L
alluvial fan	all	H	M	H	H

“customized design process” requiring more data and expertise are available in the draft document. Major steps using the default design process include: (1) evaluating existing site conditions, (2) identifying functional objectives, and (3) developing site prescriptions.

While still under development, key concepts to be included in this guidance document are: (1) methods for assessing watershed-scale limiting/constraining factors, (2) information on how to conduct the assessment at the appropriate spatial scale, and (3) riparian stand modeling methodologies for watershed-scale projects (for example, the RAIS model for LWD and shade; Welty et al. 2002).

The initial version of the guidelines will be used for implementing Section V pilot projects that are undertaken in the summers of 2011 and 2012. Potential pilot project locations range from Santa Cruz County to Humboldt County in the Coast Ranges, as well as in the Klamath Mountains and the Sierra Nevada. The list of potential pilot projects include the full range of desired objectives, including increasing large wood loading, promoting increased biotic diversity, reducing catastrophic wildfire risk, and accelerating conifer tree growth.

The vision for the final document is broad and includes sections on project pre-consultation guidance with the state and federal agencies, context assessment at the watershed scale, project/site evaluation tools using a refined version of the modified Washington watershed analysis approach discussed above, and monitoring guidance. The final document will use feedback from the preliminary documents produced by the group.

Conclusions

The VTAC is optimistic that the pilot projects will be successful and demonstrate to landowners in California that site-specific riparian management is both economically viable and ecologically valuable. It is clear that rapidly accelerated habitat improvement for listed anadromous fish species such as coho salmon is needed if these species are to recover in this state (NMFS 2010). Hope remains high since all surveyed stakeholder groups agree that site-based riparian management is appropriate where it is justified and necessary for recovery. Most types of projects seem likely to be accepted by the state and federal reviewing agencies in appropriate situations. It does appear, however, that riparian management that results in impacts only on hillslopes will be initially easier to permit as part of a THP in California than those that cause impacts in the stream channel (for example, large wood placement projects). These projects may require a federal permit before they can be completed and difficulties currently exist with addressing compliance with the Federal Endangered Species Act (ESA).

The status of the pilot projects and guidance document that will allow for broad application of the site-specific approach for riparian management will be presented to the Board in the second half of 2011. The final VTAC report is expected to be completed by the summer or early fall of 2012.

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Assessing Effects of Changing Land Use Practices on Sediment Loads in Panther Creek, North Coastal California[1]

Mary Ann Madej,[2] Greg Bundros,[3] and Randy Klein[3]

Abstract

Revisions to the California Forest Practice Rules since 1974 were intended to increase protection of water quality in streams draining timber harvest areas. The effects of improved timber harvesting methods and road designs on sediment loading are assessed for the Panther Creek basin, a 15.4 km^2 watershed in Humboldt County, north coastal California. We compute land use statistics, analyze suspended sediment discharge rating curves, and compare sediment yields in Panther Creek to a control (unlogged) stream, Little Lost Man Creek. From 1978 to 2008, 8.2 km^2 (over half the watershed) was clearcut and other timber management activities (thinning, selection cuts, and so forth) affected an additional 5.9 km^2. Since 1984, 40.7 km of streams in harvest units received riparian buffer strip protection. Between 2000 and 2009, 22 km of roads were upgraded and 9.7 km were decommissioned, reducing potential sediment production by an estimated 40,000 m^3. Road density is currently 3.1 km/km^2. Sediment rating curves from 2005 to 2010 indicate a decrease in suspended sediment concentrations when compared to the pre-1996 period, although Panther Creek still has a higher sediment yield on a per unit area basis than the control stream.

Key words: suspended sediment rating curves, timber harvest, watershed assessment, sediment yield

Introduction

Historically, timber harvest and related road construction have caused accelerated hillslope erosion and sediment delivery to streams, impacting salmonid habitat in California and the Pacific Northwest (Meehan 1991, Murphy 1995). In recognition of these problems, several regulatory efforts have been implemented to ameliorate the effects of land use activities on streams. For example, revisions to the modern California Forest Practice Rules, implemented beginning in 1974, were intended to increase protection of water quality in streams draining timber harvest areas. Protective measures included use of improved riparian buffers, where timber harvest operations are limited, and improved standards for newly constructed and upgraded roads. In July 2000, the California Forest Practice Rules changed watercourse crossing sizing requirements for new or reconstructed permanent stream crossings from a 50-year storm to a 100-year storm (CDF 2001), thus gradually reducing the risks of culvert failure and resultant gullies and landslides as new larger culverts are installed. Forest Practice Rule changes in 1983 changed how streams were classified and increased tree retention along Class I (fish bearing) and Class II (habitat present

[1] This paper was presented at the redwood science symposium: coast redwood forests in a changing California. June 21-23, 2011. Santa Cruz, California.

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for non-fish aquatic species) streams. Class III streams contain no aquatic life and receive the least protection.

In addition, Section 303(d) of the federal Clean Water Act of 1972 requires states to identify water bodies that do not meet water quality standards and are not supporting their beneficial uses. Redwood Creek in Humboldt County, north coastal California, was 303(d) listed as sediment impaired in 1996, and a Total Maximum Daily Load (TMDL) was developed for the watershed in 1998 (CRWQCB 1998a, 1998b). The Redwood Creek Sediment TMDL identified major sediment sources to Redwood Creek as streamside landslides, gully erosion, and cutbank and fillslope failures on unpaved logging roads. Although state forest practice rules do not require long-term road inspection and maintenance, the TMDL included targets for desired watershed conditions with respect to road design, location, inspection, and maintenance along with timber harvest practices in sensitive streamside areas. In recent years landowners in the basin have conducted several erosion control projects on their lands to address erosion concerns. Madej et al. (2007) evaluated sediment loads in the mainstem of Redwood Creek in the context of the TMDL, but trends in sediment yields in tributary basins of Redwood Creek have not yet been analyzed to that extent.

In Panther Creek, a tributary of Redwood Creek, timber harvest has been the primary land use since the 1930s. Green Diamond Resource Company (GDRCo), the current landowner, began implementing an Aquatic Habitat Conservation Plan (AHCP) in 2007, which includes the Panther Creek basin. During the past decade, GDRCo has completed several road assessments, road upgrades and road decommissioning projects in the watershed. Since 1980, the U.S. Geological Survey (USGS) and Redwood National Park (RNP) have operated a stream gaging station at the mouth of Panther Creek, where stream discharge and suspended sediment are measured. Consequently, it is an opportune time to evaluate sediment yields measured at the gaging station to determine if sediment reduction goals have been met. The objective of this paper is to evaluate possible changes in sediment loading in Panther Creek due to improvements in timber harvest methods and road designs and maintenance by analyzing shifts in suspended sediment discharge rating curves, and comparing sediment yields in Panther Creek to a nearby control (unlogged) stream.

Field area

Panther Creek has a drainage area of 15.4 km^2 (*fig. 1*). Total basin relief is 600 m. The bedrock is a quartz-mica schist of the Franciscan assemblage. Aerial photographs taken in 1936 show that about half of the basin was covered with old-growth redwood and Douglas-fir forests, but upper hillslopes had been clearcut logged and yarded using steam donkeys (steam-powered winches that dragged logs across the hillslopes) and railroad equipment. Harvest operations accelerated in the 1950s with increased demand for timber products following World War II. Beginning in the late 1940s and continuing into the 1980s ground based equipment, such as crawler tractors, were used for skidding. By the early 1980s nearly all of the old-growth forest had been harvested. Second-growth harvest activities began in the 1980s utilizing commercial thinning and rotational harvests as the timber reached commercial size and age. The majority of roads in Panther Creek were constructed before implementation of the Z'berg-Nejedly Forest Practice Act beginning in 1974, which provided for more stable road designs in subsequent years. In addition, the early old-growth harvesting retained very minimal riparian protection along streams.

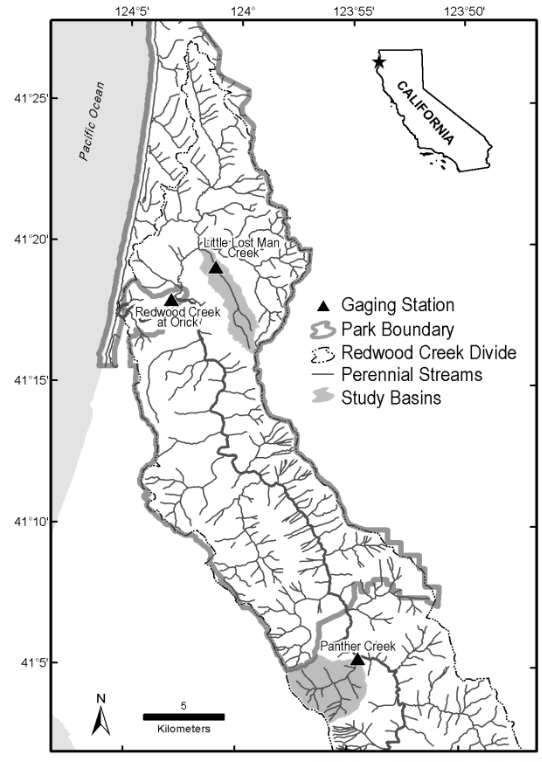

Figure 1—Location map.

In 1980 the USGS, in cooperation with Simpson Timber Company (later renamed Green Diamond Resource Company), established a gaging station on Panther Creek (Station #11482125). In addition, the USGS established a gaging station on an unlogged tributary in the lower Redwood Creek basin, Little Lost Man Creek (Station #11482468) in 1975 (*fig. 1*). Little Lost Man Creek (9.1 km^2) is underlain by sandstones and mudstones of the Franciscan assemblage. Basin relief, at 610 m, is similar to Panther Creek, and both basins exhibit similar erosion processes (primarily shallow debris slides and debris flows). Continuous streamflow and storm flow suspended sediment samples were obtained at both stations. RNP assumed operations of the gaging stations in 1993. In 2006 RNP installed turbidity sensors at the gaging stations to supplement the suspended sediment sampling program.

The Panther Creek basin receives an average of about 2,000 mm of precipitation annually, most of which falls as rain between October and March. Annual mean flow is 0.68 cms. The highest peak flow of the 30-year record, 57 cms, occurred on January 1, 1997, although based on records from Redwood Creek, peak flows in the 1960s and 1970s were probably larger (*fig. 2*). The storm in January, 1997 initiated 13 landslides > 400 m^2 in area in the Panther Creek basin, which mobilized about 24,000 m^3 of material. About 80 percent of this volume was generated by landslides associated with roads and landings, including a 9,000 m^3 debris flow that severely scoured the channel (Curren 2007). Several smaller landslides occurred the previous year as well. Annual rainfall in Little Lost Man Creek basin is 1,700 mm and mean annual flow is 0.34 cms. No landslides were detected here on 1:6,000 air photos following the 1997 storm.

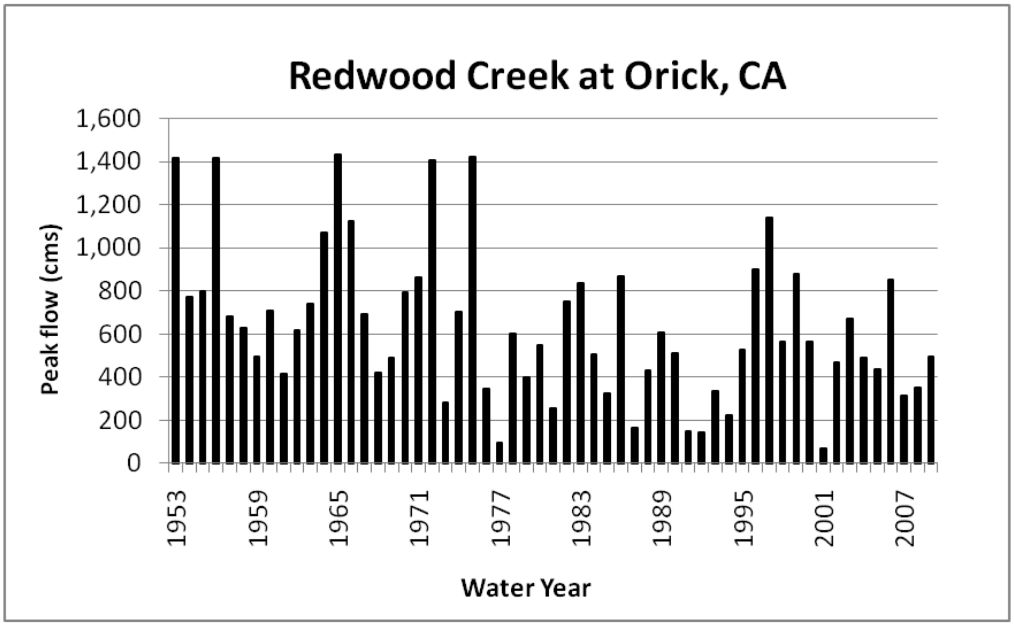

Figure 2—Peak flows at Redwood Creek at Orick, CA.

Methods

We used timber harvesting plans approved by California Department of Forestry and Fire Protection (CAL FIRE), aerial photographs, and field-based road inventories to compile land use statistics for the Panther Creek watershed. Because work proposed in a timber harvesting plan may be implemented any time within three years of plan approval, exact dates of some timber harvest and road work are not known. Attributes included the timing and size of timber harvest, silvicultural and yarding methods, and level of streamside protection (RNP, unpublished data). Little Lost Man Creek is almost completely unlogged, with no change in land use during the study period.

Gaging stations in Panther Creek and Little Lost Man Creek record water stage at 10-minute intervals. Stage data are converted to discharge values using individual stage-discharge rating curves for each station, based on occasional discharge measurements that relate recorded stage to water discharge. Since 2006, electronic turbidity sensors mounted on articulating booms have measured turbidity in the water column at 10 minute intervals. Before 1992 suspended sediment samples were collected manually at high flows, whereas after 1992 an electronic pumping water sampler collected samples according to changes in turbidity or stage (Klein 2003). Water samples were processed in USGS or RNP laboratories to obtain suspended sediment concentrations. Accuracy of suspended sediment load computations most likely improved greatly with the addition of automated pumping samplers and turbidity-controlled sampling (Lewis 1996).

We constructed suspended sediment rating curves by plotting sediment concentration (SSC, mg/l) against instantaneous discharge (cms) on a logarithmic scale for five time periods, based on groupings of water years that were wetter (w) or drier (d) than normal (n): pre-1986 (w), 1987 to 1995 (d), 1996 to 1998 (w), 1999 to 2005 (d), and 2006 to 2010 (n). (A water year (WY) extends from October 1 to September 30). We ran an ANCOVA model on the sediment data, using log(discharge) as the independent covariate, log(suspended sediment concentration) as the dependent variable, and time period as a categorical factor. An interaction variable of [time period * log(discharge)] was included to assess possible differences in intercepts and slopes of the sediment rating curves.

Annual suspended sediment loads were available for most years for WY1980 to 2010 for Panther Creek from either computations by RNP (WY1992 and later) or from the USGS (before WY1992). Annual loads also exist for most of this period for Little Lost Man Creek, with the exception of two periods (WY1983 to 1984 and WY1990 to 1992). Prior to the use of continuous recorded turbidity as a surrogate for SSC, the sediment rating curve approach was used for computing loads.

Results

Based on CAL FIRE-approved harvesting plans, the cutting rate in Panther Creek from 1978 to 2008 ranged from 0 to about 7 percent of the watershed annually, and averaged about 3 percent per year. Even-aged management (clearcut, rehabilitation, and so forth) was applied to more than 60 percent of the watershed. Non-even-aged management (thinning, selection, and so forth) was applied to an additional 30 percent during this period. For the same period, about 60 percent of the harvested area was yarded by tractors, 30 percent by cable systems and 10 percent by a combination of helicopters and feller-bunchers. In 2006 GDRCo stopped all use of tractors for ground-based yarding operations in favor of feller-bunchers, which appear to minimize ground disturbance.

Road density in the Panther Creek basin is currently 3.1 km/km^2, down from 4.2 km/km^2 in 2000. Since 2000, 24.5 km of road have been upgraded and 15.7 km have been decommissioned. The potential sediment savings associated with this work is about 40,000 m^3 (Bundros and Short 2011). Based upon road assessment data from 2004, one third of all roads assessed in Panther Creek were being maintained. In 2009, reassessment showed that three-fourths of the assessed roads were being maintained (or were maintenance-free because they have been decommissioned). GDRCo’s AHCP contains standards that require road inspections and maintenance of mainline roads every year and secondary roads every 3 years for the 50-year life of the permit (GDRCo 2006), which are more stringent standards than the state forest practice rules.

Watercourse and Lake Protection Zone (WLPZ) rules were adopted by State Board of Forestry and Fire Protection and implemented in 1974, following the passage of the Z'berg-Nejedly Forest Practice Act of 1973 by the State Legislature. The rules and classification criteria for streams have changed several times, with significant changes occurring in 1983 (Berbach 2001). Since 1984, 57.6 km of streams have been included in timber harvest plans in the Panther Creek basin, of which 7.3 km were classified as Class I. Of the remaining stream length, for the period of 1984 to 1993, 44 percent (16.5 km) were classified as Class II, and 9.3 km classified as Class III. For the period 1994 to 2008, 26.2 km of stream were classified as Class II (78 percent) and only 7.6 km classified as Class III. Consequently a higher percentage of stream length received increased riparian tree retention and protection in recent years.

Sediment rating curves were constructed for five time periods (*fig. 3*). The sediment records exhibit high variability, as is typical of mountainous, forested watersheds (Sadeghi et al. 2008). Nevertheless, we can discern some statistically significant trends in the results. The results of the ANCOVA analysis show that time period is a significant variable in explaining the trends ($p < 0.001$). The earliest period (WY1980 to 1986, #1) had the steepest slope, which can indicate that new sediment sources became available when discharge increased (Asselman 2000).

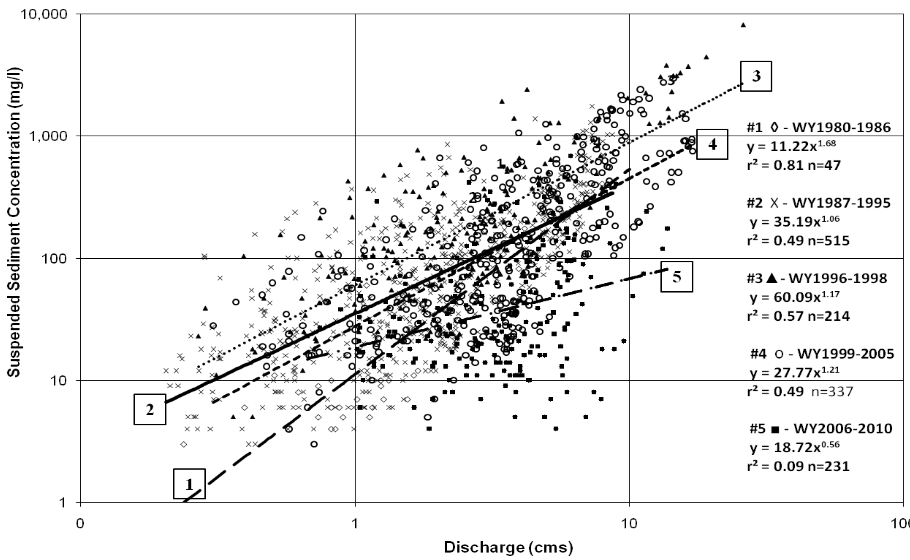

Figure 3—Sediment rating curves for Panther Creek for five time periods.

However, because of logistic difficulties in manual sampling, this early period also had far fewer samples (47) than other periods. The slope of the rating curve decreased in the following period (WY1987 to 1995, #2), showing that sediment concentrations were lower at high discharges than in the earlier period. The period with high landslide activity (WY1996 – 1998, #3) had a significant upward shift in the sediment rating curve, with higher sediment concentrations across the range of flows. These elevated levels did not persist long, however, and the curve for the period WY1999 to 2005 (#4) lowered to the WY1987 to 1995 level (intercepts were not significantly different). The most recent period (WY2006 to 2010, #5) has the lowest rating curve. Even though the relationship between suspended sediment concentration and discharge is weak for this time period (r2 of only 9.3 percent), it is still significant at the 95 percent confidence level (p-value of < 0.001).

Annual suspended sediment loads differed substantially between Panther Creek and the control stream, Little Lost Man Creek, during the common years of operation. The annual suspended sediment load for Panther Creek ranged from a low of 23 Mg to a high of 36,000 Mg. Average sediment load for Panther Creek was 220

Mg/km^2 versus 56 Mg/km^2 for Little Lost Man Creek. We compared the ratio of sediment loads for each year (Panther load divided by Little Lost Man load, *fig. 4*). The average ratio was 4, with a maximum of 20 during the high flow year of WY1997. In WY1981, 1999 and 2001 the normalized sediment load in Panther Creek was less than that of Little Lost Man Creek. In WY1999, several inner gorge landslides temporarily elevated loads in Little Lost Man Creek, thus reducing the ratio for that year. WY2001 had the lowest annual flow and the lowest sediment load values in the Panther Creek record, so the small differences between Panther and Little Lost Man creeks in low flow years (0.3 Mg/km^2) probably are not meaningful.

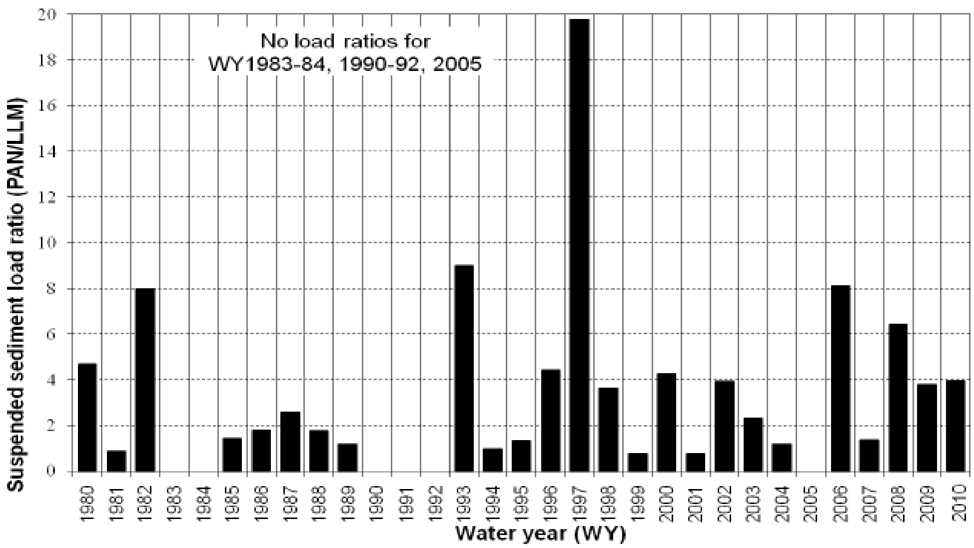

Figure 4—Comparison of annual suspended sediment yields between Panther Creek (PAN) and a control stream, Little Lost Man Creek (LLM).

Discussion

Sediment supply to rivers in forested mountainous terrain is commonly episodic in nature through such processes as mass movement and gullying. Large storms are usually needed to trigger severe erosion events, and the landslide-producing storm in Panther Creek in 1997 is an example of such an event. In the Redwood Creek basin, once material is delivered to steep low-order channels, much of it is routed downstream within a few years (Pitlick 1995). In recent years land managers have implemented erosion control work on the hillslopes of the Panther Creek basin to decrease sediment input to streams and provide more protection to sensitive streamside areas. Nevertheless, most of these treatments were implemented post-1997 and have not been "tested" by a large storm (20-year return period or greater). We would not expect reductions in erosion to be detected at the mouth of Panther Creek until such a test occurs.

Sediment rating curves shifted upward during and immediately after a period of high landslide activity in 1996 to 1998. Since then, suspended sediment concentrations at moderate to high discharges have decreased. Even so, the sediment records exhibit high variability. This variability problem may be addressed by further investigations, such as separating data by storms or hydrograph limbs. Sediment loads are still higher in Panther Creek than in the control stream, but are less than half that of the Redwood Creek basin average (600 Mg/km^2 measured at the Redwood Creek at Orick gaging station).

Assessing the influence of land use practices and hillslope improvement work on sediment loads is complicated by factors such as the lag time between implementing erosion control work and sediment transport events, the effect of legacy features from past timber harvest, and the occurrence of extreme events. The high volume of material mobilized from road-related landslides in the 1997 storm attests to ongoing sediment production from remaining legacy roads. On the other hand, current policies such as increasing riparian protection, decreasing timber harvest on inner gorge slopes, and reducing controllable sediment sources along roads may reduce sediment loads in future storms. Continued stream and hillslope monitoring will help elucidate such linkages.

Conclusions

The Panther Creek basin is an actively managed watershed with an average timber harvest rate of three percent of the watershed per year. Improvements in timber harvest techniques, streamside protection, road design and maintenance implemented in the Panther Creek basin in recent years appear to have caused a downward shift in the sediment rating curve and reduced sediment loads relative to those of decades earlier. Results from sediment sampling indicate that sediment loads were highest during a period of high landslide activity in 1996 to 1998; most of the sediment volume was contributed by landslides associated with roads and landings. Consequently, the effect of untreated legacy erosion problems in a basin can obscure the beneficial effect of improved erosion control work and improved timber harvest practices elsewhere in a watershed. Since 1998 suspended sediment rating curves for Panther Creek have shifted downward, indicating a general decrease in sediment concentrations. Annual suspended sediment yields remain about four times higher than in a control (unlogged) watershed. Hydrologic stresses during a large storm with a discharge having greater than a 20-year recurrence interval are needed to test the effectiveness of recent erosion control work and improved timber harvest methods on the hillslopes and to realize the time-delayed benefits. Adaptive land management involves monitoring the effects of management activities, and modifying land management approaches and techniques based on what is found to be effective. The data set for Panther Creek provides an excellent baseline for future assessments of improved management actions.

Acknowledgments

We are grateful for the field and technical support from Tom Marquette, and the meticulous laboratory analysis of Carrie Jones, both of Redwood National Park. Green Diamond Resource Co. provided access to the Panther Creek gaging station. Julie Yee and Philip van Mantgem provided statistical guidance, and reviews by Darci Short, Matt House, and two anonymous reviewers improved the manuscript.

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Fine Sediment Sources in Coastal Watersheds with Uplifted Marine Terraces in Northwest Humboldt County, California

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Abstract
Erosion in the Mill and Luffenholtz Creek watersheds in Humboldt County, California, with their extensive clay soils, can lead to high turbidity levels in receiving bodies of water, increasing the costs of treating water for domestic water supplies. Detailed road and erosion surveys and monitoring of suspended sediment, discharge, and turbidity levels in Mill Creek (3.11 km^2) and Luffenholtz Creek (12.95 km^2) were completed to determine the sources of turbidity. Watershed physiographic and land use characteristics were compared for those basin areas upstream from the eight monitoring sites and a statistical analysis was completed to determine whether significant differences existed between turbidity levels in the watersheds. The Luffenholtz Creek watershed, particularly Grassy and 18 Creeks, and the main stem of Luffenholtz, just downstream of the 21 Rock Quarry, were the sub-watershed areas with the highest turbidities. They were also the sub-watershed areas with the highest density of roads, the most miles of roads adjacent to streams, and the highest concentration of identified erosion-prone sites. Although Mill Creek was the watershed with the highest rate of timber harvesting during the past 10 years, the steepest stream profile, and a history of more extensive fires, its turbidities were nearly always lower than Luffenholtz Creek.

Key words: sediment budget, turbidity, forest road, water quality, coastal California

Introduction

Turbidity in Luffenholtz Creek, north coastal California, has increased treatment costs for the City of Trinidad. Excessive turbidity, especially the kind caused by erosion of colloidal clay, can be expensive and difficult to treat. Even with effective use of filters and chemical alars (which floc the clay particles together allowing filtration) there is still the potential for creation of a dangerous by-product, chloro-tri-halomethanes that can remain in the treated water. This by-product is a known cancer-causing agent [2]. High turbidities can lead to water shortage emergencies and can create safety hazards for the community; storage shortages can occur from brief firefighting efforts or broken pipes during a freeze. Storage tank recharge can be difficult during periods of high turbidity.

Creeks draining the Trinidad area terraces were considered by Boyd (1978) to be possible sources of water quality degradation in the Area of Special Biological

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Significance (ASBS). This ASBS is a specifically designated sensitive coastal area in and around Trinidad Bay in the area near where Mill and Luffenholtz Creeks drain into the ocean (*fig. 1*).

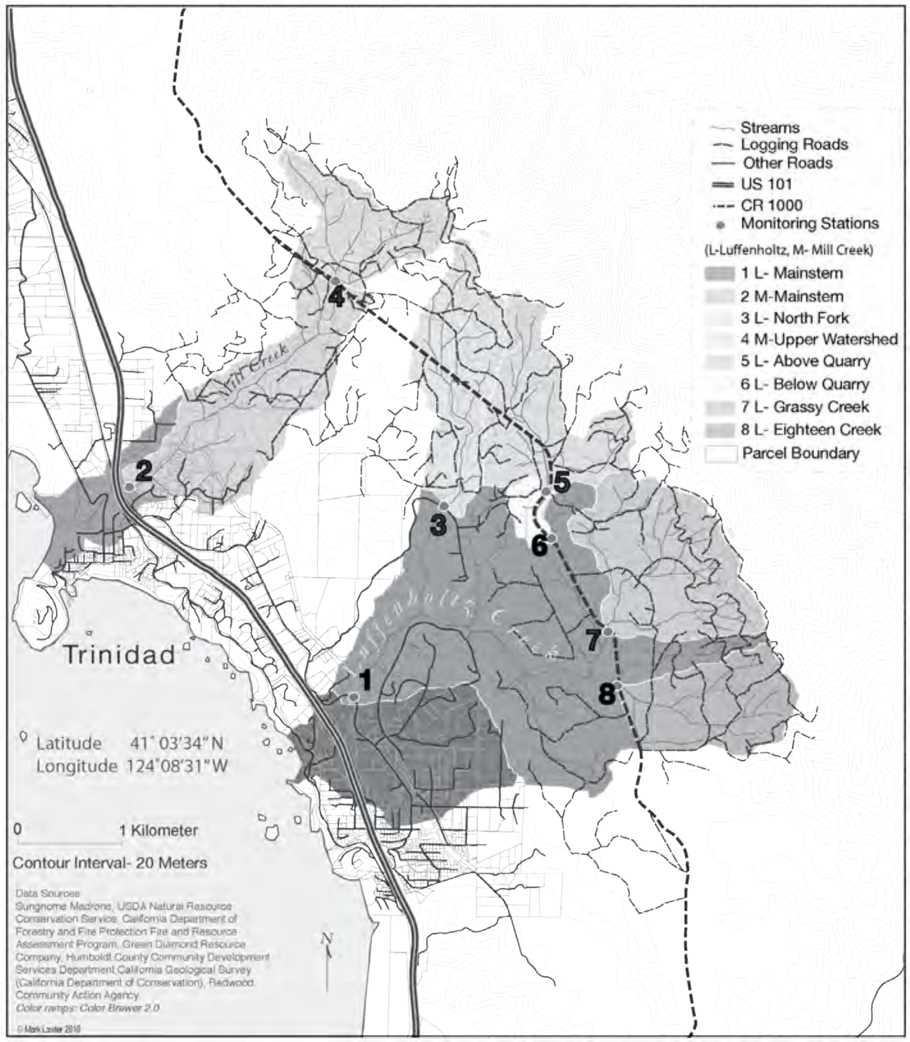

Figure 1—Study watersheds.

The objective of this study is to locate and prioritize for treatment the sources of fine sediment in these coastal watersheds dominated by uplifted marine terraces. This study will help inform the efforts to improve water quality in these streams for humans and aquatic species. For this investigation we formed two hypotheses. The first is that road density and adjacency have a greater effect on turbidity than other physiographic and land use characteristics on these uplifted marine terraces, and that because of this Luffenholtz Creek will have higher turbidities than Mill Creek. The second hypothesis is that the 0.8 km section of the CR 1000 Road immediately below the 21 Rock Quarry is a significant contributor of turbidity to Luffenholtz Creek due to its immediate proximity to the stream.

Study site

Luffenholtz and Mill Creeks were chosen for study because they represent unique, and not well studied, topography for this region. They provide habitat for endangered species and water supplies for domestic use. Study sites are shown in *fig. 1*. Both creeks support populations of anadromous steelhead trout (*Oncorhynchus mykiss*), an Endangered Species Act Listed species, and cutthroat trout (*Oncorhynchus clarki*). The Mill Creek watershed is 404 ha in size, and lies on the north side of the city. It was the original water supply and is now the back up or alternative water supply. The Luffenholtz Creek watershed is 1527 ha in size and is the primary source of water for the citizens of the city of Trinidad, population 311, and other watershed residents. As the Mill and Luffenholtz Creek watersheds are adjacent and have similar soils, rainfall, and geological formations, they were expected to have similar particle characteristics and turbidity levels.

Comprehensive sediment source assessments have been completed recently for most of the watersheds that drain the Trinidad Terraces area, including Mill and Luffenholtz Creeks (Allan and Ledwith 2008). Water quality sampling on the two creeks was done in spring of 2007 by Cinowalt and Van Matre (unpublished). Sampling at the mouths of all the main local creeks, draining this uplifted terrace area, was carried out in 2006 through 2009, by Madrone and Allan (unpublished). My project builds on these recent studies. The grab samples taken over a 4 year period increased from 1 per year in 2006 to 42 samples in Water Year 2009. This increased sampling was also focused completely on Mill and Luffenholtz Creeks for WY 2009. The results of this sampling were used to compare with watershed characteristics and land use variables. These variables included the results from detailed on-the ground roads and creek surveys in order to determine what and how big the sources of sediment were, whether there was delivery potential, and how they affected turbidity. *Table 1* provides the detailed watershed characteristics and land use variables that make up the different sub-basins of the Luffenholtz and Mill Creek watersheds. These details include geology, fire history, drainage area and basin relief, road surfacing, densities and lengths, timber harvest history, harvesting types and techniques, past and potential soil loss. The following discussion highlights the table information.

The geologic map for the Trinidad area shows that Mill Creek drains predominantly Franciscan rocks. Luffenholtz also has significant areas of Franciscan geology, mostly in the inner canyon and on slopes between terraces. The remainder of both watersheds is underlain by uplifted marine terrace soils. The Franciscan soil types have higher clay content (23 to 40 percent) in the A and B horizons than the Marine Terrace soil types (13 to 30 percent). Both soil types have high clay contents (*fig. 2*).

Table 1—*Site/sub-basin characteristics.*

Item	Site 2 Main Stem Mill Creek	Site 4 Upper Mill Creek	Site 1 Main Stem Luffenholtz Creek	Site 3 North Fork Luffenholtz Creek	Site 5 Luffenholtz Creek Above 21 Quarry	Site 6 Luffenholtz Creek Below 21 Quarry	Site 7 Grassy Creek	Site 8 Eighteen Creek
GPS Locations	N41°03'56" W124°08'24.7"	N41°04'59" W124°07'1.2"	N41°02'50.5" W124°06'47.5"	N41°03'49.7" W124°06'10.2"	N41°03'54.5" W124°05'33.1"	N41°03'39.9" W124°05'31.9"	N41°03'12.2" W124°05'6.4"	N41°02'55.2" W124°05'2.7"
Watershed Characteristics								
Drainage Area (Hectares)	311	127	1295	129	153	171	231	153
Hydrologic Network	Linear	Dendritic	Dendritic	Linear	Dendritic	Dendritic	Dendritic	Dendritic
Basin Relief (ft)	1164	608	1039	606	561	601	580	713
Basin Relief (m)	355	185	317	185	171	183	177	217
Relief Ratio	17	30	15	16	21	18	21	23
Geology % Franciscan	70	60	62	66	67	67	80	60
Geology % Marine Seds.	30	40	38	34	33	33	20	40
Land Use Variables								
Kilometers road per Kilometer^2	3	2	3	5	4	4	5	7
Kilometer road	9	3	40	6	6	7	11	11
Kilometers road on native surface	6	1	26	4	4	4	8	5
Kilometers road on rocked surface	3	1	15	2	2	2	3	6
Past soil loss (20 years) (m^3)	450	390	3680	220	230	1530	180	1520
# potential erosion sites	23	15	131	22	18	24	23	20
# sites per square kilometer	7	12	10	17	12	14	10	13
Potential soil loss (m^3)	1040	10	5270	960	270	650	970	1040
Potential soil loss (m^3/kilometer^2)	330	10	410	740	180	380	420	580
% harvested 2000-2010	44	25	20	50	30	40	20	50
% harvested 1990-1999	7	0	40	10	30	30	60	10
% harvested 1979-1989	< 5	0	40	30	20	10	10	30
% area burned 1906 fires	0	0	0	0	0	0	0	0
% area burned 1936 fires	50	100	30	67	100	100	30	0
% area burned 1945 fires	60	100	30	75	80	76	40	0

Methods

We used the methods listed below to evaluate sediment sources for two small coastal watersheds in northwest California, USA. Seven sources of data are utilized in this analysis including road erosion and sedimentation assessments completed in 2007, by Madrone, D. Allan, and T. Ledwith of the Redwood Community Action Agency for all of Mill and Luffenholtz Creeks. This included an aerial photo analysis as well as a 100 percent survey of all roads and stream crossing sites. Grab samples for turbidity were collected near the mouths of Mill and Luffenholtz Creeks, as well as other local creeks that drain into the ASBS area or the bay, in 2006 to 2009 by Allan and Madrone. Grab samples for turbidity and SSC were collected in spring 2007 by Cinnowalt and Van Matre at Site 1 on Luffenholtz Creek and downstream 0.5 km from Site 2 on Mill Creek. Stream bank and landslide erosion inventories and assessments were completed in spring 2008 by Allan and Madrone, for the inner canyon areas on Mill and Luffenholtz Creeks. GIS mapping was completed by M. Lawler and included 2009 data collected by the author on timber harvest history of silvicultural and yarding methods, and rate of harvest, parcel boundaries, soils,

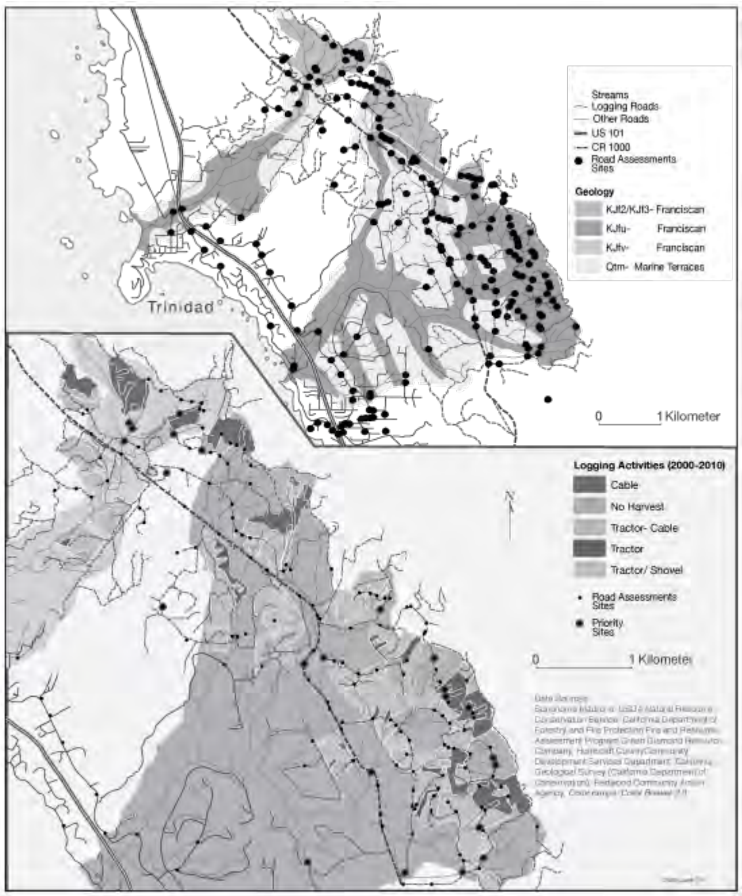

Figure 2—Road assessment sites, geology, timber harvesting (2000 to 2010).

geology, fire history, and locations of roads and streams. Grab samples for turbidity and flow measurements were collected in WY 2009 by Madrone at Site 1 on Luffenholtz Creek, at Site 2 on Mill Creek, and at six tributary sites upstream in Mill and Luffenholtz Creeks. Continuous probe sensors for turbidity and stage were maintained at Site 1 on Luffenholtz Creek and at Site 2 on Mill Creek in the spring of 2009. The relationship between turbidity and SSC in these 2007 samples was used to generate SSC for the 42 turbidity grab samples taken at Sites 1 and 2 in water year (WY) 2009. Suspended sediment loads were computed for three late season storms in WY2009 using automated turbidity data as a surrogate for suspended sediment concentration (SSC, mg/l) (Lewis and Eads, 2001). The storm dates (5/1 to 5/4/09, 5/4 to 5/9/09, and 5/13 to 5/17/09) were chosen because these were the only storms for which both sites had reliable continuous turbidity data.

Results

As compared to Luffenholtz Creek, Mill Creek is smaller, steeper, has more unstable ground and has burned more completely and more often in the past 100 years. Mill Creek has had a higher rate of timber harvest in the past 10 years than Luffenholtz Creek, but in the harvested areas buffers were left along the creeks and the logging was completed on less erodible ground. Mill Creek was also harvested with less disturbing shovel-logging excavators with a loader head. Notably, Mill Creek does have a lower road density and less road adjacency to streams and therefore less delivery potential (*table 2*). Swift (1988) also found that roads and skid trails were a major source of sediment from forestry-related activities.

Table 2—*Watershed road densities*.

Creek	Luffenholtz Creek at Westhaven Drive	Joland Creek at Westhaven Drive	Mill Creek at Stagecoach Road	South Fork Parker Creek at Scenic Drive	North Fork Parker Creek at Scenic Drive	McConnahas Mill Creek at Scenic Drive	Dead Man's Gulch at Scenic Drive	North Fork Two Creeks at 2nd Avenue	South Fork Two Creeks at Westhaven Drive	Two Creeks at Scenic Drive
Site #	Site 1	Site A	Site B	Site C	Site D	Site E	Site F	Site G	Site H	Site I
Kilometer road	40.0	3.4	9.0	N/A	2.3	4.1	0.6	N/A	N/A	4.3
Drainage area (km^2)	13.0	0.5	3.1	N/A	0.3	2.7	0.3	N/A	N/A	0.9
Road density (km/km^2)	3.0	6.8	3.0	N/A	6.8	1.5	2.0	N/A	N/A	5.0

Luffenholtz Creek has a legacy of riparian roads with close proximity and thereby delivery, as well as a high road density, due in part to the dendritic nature of the channel network. It has also had a high rate of annual timber harvesting over the past 40 years, including along its streams and in the inner canyon. Harvest rates in the 1980s and 90s averaged 4 percent per year including clear cutting with tractor logging. The watershed with the highest rate of harvest on steep ground and the highest density of roads also has the higher turbidities.

Mill Creek is nearly always less turbid than main stem Luffenholtz Creek (*fig. 5*). Suspended sediment load calculations for three storms (5/1 to 5/4, 5/4 to 5/9, and 5/13 to 5/17) in early May 2009 were 0.02 metric tons (6 x 10^{-5} t per ha) for Mill Creek and 21.29 metric tons (0.16 t per ha) for Luffenholtz Creek (*figs. 3* and *4*). A plot of normalized flows for the two main stems based on the 42 grab samples in WY 2009 showed that flows are similar (cms per km^2) for Luffenholtz and Mill Creeks when normalized, but the North Fork is not similar (*fig. 5*). Starting with the late December 2008 storms, the North Fork normalized discharge regularly exceeded that of Luffenholtz and Mill Creeks.

The rainy season for WY 2009 can be characterized as being 75 percent of normal (112.7 cm compared to the 17 year average of 150.9 cm per year at Allan’s gage). Rainfall was about 21 percent higher in WY 2009 at the Luffenholtz Gage for a total of 143.4 centimeters. There were numerous dry periods between storms, with the only significant rainfall events (greater than 5 cm in 24 hours) happening on 12/28/08, 3/16/09, and 5/4/09. These events match up with the highest turbidities and stages recorded. Turbidities at Site 5 upstream of the 21 Rock Quarry were substantially lower during the first part of the season (from October 2008 through mid December 2008) than at the downstream Site 6 on Luffenholtz Creek. As the season progressed and after a large rainfall event (140 mm in 5 days) peaking on 12/28/08 the difference in turbidities between Site 5 and 6 was decreased. The rainfall events of 3/16 and 5/4 did not raise turbidities at Site 6 over Site 5. During the rainfall event of 54 mm in 6 hours on May 4, while lower Luffenholtz, the North Fork, and Mill Creeks had a jump in turbidities, upper watershed Sites 4 through 8 showed little increase in turbidity. This suggests that the upper tributary retention basins had begun functioning again after a dry period in April.

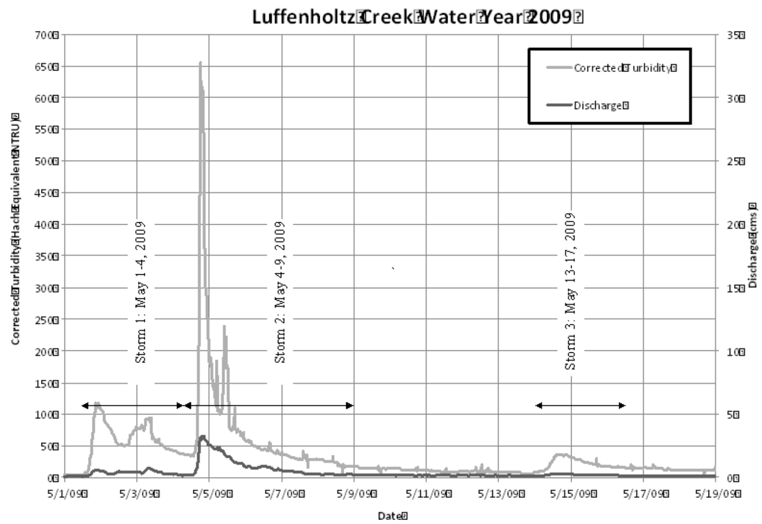

Figure 3—Hydrograph and turbidigraph for three storm periods, Luffenholtz Creek, Humboldt County, California.

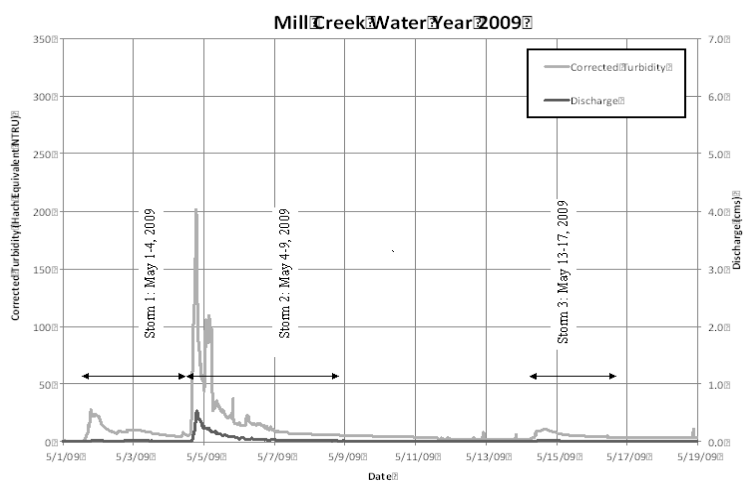

Figure 4—Hydrograph and turbidigraph for three storm periods Mill Creek, Humboldt County, California.

Statistical analysis of turbidities from WY 2009 grab samples was completed for several of the sites using non-parametric tests because the turbidity data did not pass normality. The P-values for all sites were less than 0.05 and because the data did not pass normality the median value is the parameter for all tests. The analysis is nonparametric in that actual values were not used, just their relative rank, or position toward one another. Luffenholtz Creek (Site 1) was found to have significantly higher turbidity than Mill Creek (Site 2) and Site 5 upstream of the 21 Quarry had significantly higher turbidity than the downstream Site 6.

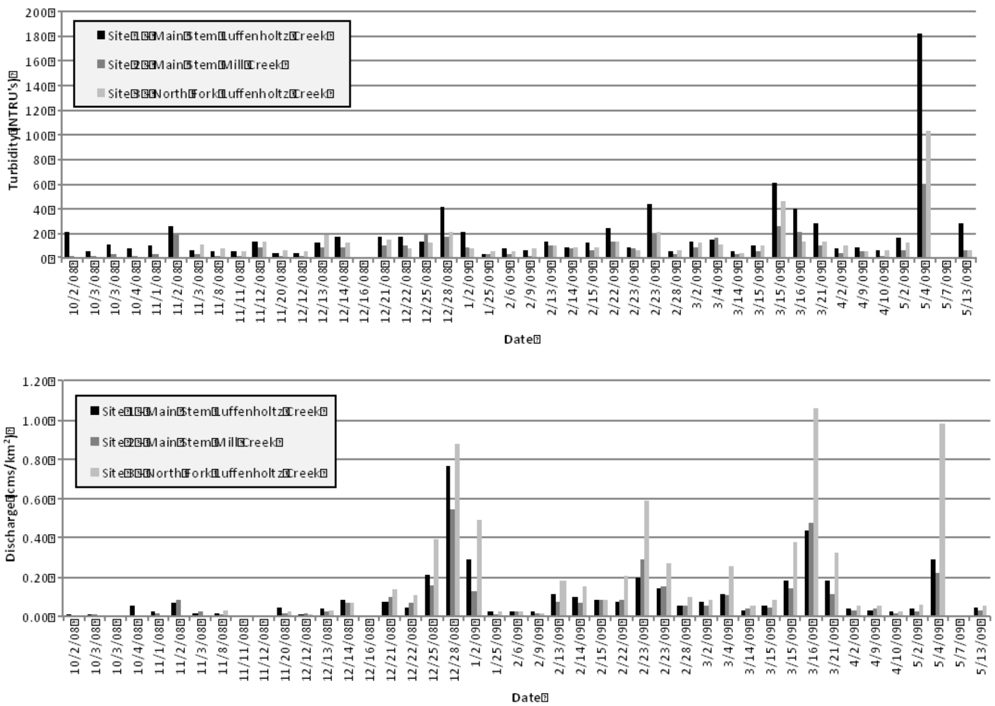

Figure 5—WY 2009 Grab sample turbidity and discharge data for Main Stem Luffenholtz Creek vs. Main Stem Mill Creek vs. North Fork Luffenholtz Crek, Humboldt County, California.

Discussion

Joland Creek (South Fork Luffenholtz Creek) had the highest turbidities of all of the Luffenholtz Creek tributaries monitored under this study. The North Fork of Parker Creek and both forks of Two Creeks also had very high turbidities. These high turbidities may be associated with the high road densities of dirt and gravel roads coupled with year-round land and home site clearing activities. The results are from grab samples taken between 2006 and 2009. None of these tributaries flow into Luffenholtz Creek or Mill Creek above the city's water intakes.

Mill Creek has steeper average slopes and greater basin relief than Luffenholtz Creek watershed and it has a greater percentage of its watershed in unstable geology. It has a higher rate of harvest during the past 10 years (44 percent for Mill vs. 20 percent for Luffenholtz), and has been burned more severely and more often, and yet it has lower turbidities than Luffenholtz Creek. This may be because it has a slightly lower density of roads per square kilometer (2.9 km per km^2 for Mill to 3.1 km km^2 for Luffenholtz), has fewer total kilometers of roads (9 for Mill and 40.2 for Luffenholtz), has fewer kilometers of native surface roads (5.8 to 25.7), and far fewer erosion sites (23 to 131) and less actual past soil loss (450 m^3 for Mill Creek and 3680 m^3 for Luffenholtz Creek) (*table 1*).

Recent timber yarding in Mill Creek was done with shovels (excavators), cable, and helicopters. These operations left substantial wildlife, vegetative, and canopy buffers along all perennial streams and have stayed off of the steep less stable ground. Riparian buffer strips provide an efficient and widely accepted way to help protect aquatic ecosystems and downstream values from the effects of upslope land-use activities (Reid and Hilton 1998).

Luffenholtz Creek watershed on the other hand is four times the size of Mill Creek and 10 times the size of the North Fork of Luffenholtz Creek (*fig. 1*). It had a higher rate of harvest than Mill Creek in the 1980s and 1990s, some of which was clear-cutting on steep slopes in the inner canyon and the headwaters areas, and was done with crawler tractors. This has left a legacy of stored sediments along floodplain areas in the inner canyon. These stored sediments are mobilized at bankfull events, which is when elevated turbidities occur. Luffenholtz Creek has several roads immediately adjacent to the streams. It also has more overall kilometers of roads, more native roads, and more kilometers of high-use rocked roads. Therefore, Luffenholtz Creek has much more past erosion (3680 m^3 for Luffenholtz Creek and 450 m^3 for Mill Creek) and potential erosion sites and potential erosion (131 sites for Luffenholz Creek and 23 at Mill Creek, and 5270 m^3 for Luffenholz Creek compared to 1040 m^3 at Mill Creek) (*table 1*). Rice and Sherbin (1977) noted that when considering erosion hazard in northwest California the most important considerations probably are the yarding method used, the amount of roads, and the magnitude of storms actually experienced following the harvest.

In conclusion, the data collected and analyzed for this study suggest that higher turbidities in Luffenholtz Creek, as compared to Mill Creek, are related to differences in timber harvest practices, higher road densities, more erosion-prone sites, and greater delivery potential. Data collected showing higher turbidities in the fall at Site 6 as compared to Site 5 suggest that this is due to the close proximity of the road to the creek and that a decrease in the turbidity difference later in winter and spring was due to exhaustion of fines from the road.

Data collected suggest that the lower turbidities in Mill Creek are due to lower road densities, low delivery potential for sediment, and improved timber harvesting techniques that keep heavy equipment out of riparian areas and retain substantial forested buffers for sediment filtering.

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Comparison of Estimated and Measured Sediment Yield in the Gualala River[1]

Matthew O'Connor[2], Jack Lewis[3], and Robert Pennington[2]

Abstract

This study compares quantitative erosion rate estimates developed at different spatial and temporal scales. It is motivated by the need to assess potential water quality impacts of a proposed vineyard development project in the Gualala River watershed. Previous erosion rate estimates were developed using sediment source assessment techniques by the North Coast Regional Water Quality Control Board and by the California Geological Survey. Reported in this study is the estimated sediment yield under existing conditions in Soda Springs Creek (1.5 mi^2) based on similar, but modified, techniques. The modified technique incorporated elements of both prior approaches, substituting detailed data for road erosion and adding site specific data for landslides not previously available. This study also reports sediment yields determined by a suspended sediment monitoring program utilizing Turbidity-Threshold Sampling techniques. The monitoring program produced data relating stream turbidity, suspended sediment concentration, and stream discharge during Hydrologic Years (HY) 2006 and 2007. Turbidity and streamflow data collected for HY 2008 to 2011 were used in conjunction with the relationship between turbidity and suspended sediment concentration from HY 2006 to 07 to estimate sediment yields over an additional 4 year period. Measured sediment yields are substantially lower than predicted by sediment source assessment techniques. Variation in geomorphic processes over time and space and methodological problems of sediment source assessments may be responsible for these apparent discrepancies.

Key words: suspended sediment, erosion, water quality, mass wasting, turbidity threshold sampling

Introduction

Water quality in many northern California Coast Range watersheds is designated as "impaired" because of excessive sediment loads. Sediment source assessment (SSA) techniques customarily used to estimate sediment yield in this region require identification of erosion processes and their distribution in the watershed based on field observations and remote sensing data. Erosion rate and sediment delivery to streams are estimated by measuring soil voids such as eroded gullies and landslide scars and by using model algorithms for surface erosion processes and soil creep rates into streams. This approach has been used to estimate both current (impaired) and natural background erosion rates and provide the basis for determining erosion rates expected in a restored watershed. Erosion rates are equated with sediment yield on the assumption that over the long-term they should be in equilibrium. However, time spans required for such equilibrium are unknown and may be many decades.

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[1] An abbreviated version of this paper was presented at the redwood science symposium, UC Santa Cruz, June 21-23, 2011.

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Validation of these estimates can be accomplished by direct measurements of sediment yield.

The Gualala River is listed as having sediment-impaired water quality per provisions of the Clean Water Act Section 303(d).The Gualala River watershed (*fig. 1*) is underlain by Jurassic age sedimentary and meta-sedimentary rocks of the Franciscan assemblage. The San Andreas Fault passes through the western edge of watershed. A substantial portion of the watershed is mantled by active and dormant rock slides and earthflows. The Coast Range is geologically young, and rapid rates of uplift are believed to have contributed to high erosion rates. Watershed studies by the State of California (Klamt et al. 2002) found that historic landscape disturbance caused by logging was severe, and that a trend toward watershed recovery was evident.

Four sub-watersheds located in northwest Sonoma County (*fig. 1*) have been investigated using SSA techniques and the turbidity threshold sampling (TTS) program to measure sediment yield (OEI 2008). Findings for Soda Springs Creek are representative of the three other sites, which are excluded from this report for brevity.

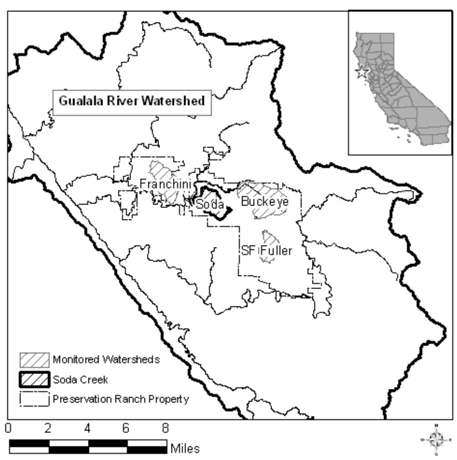

Figure 1—Location map.

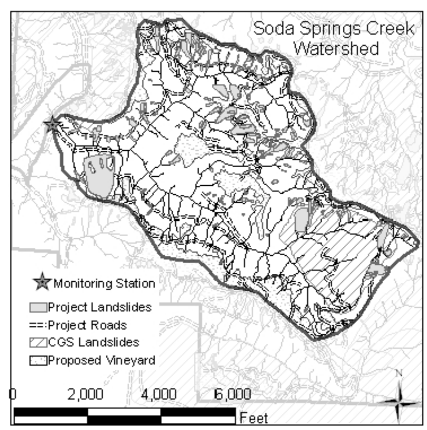

Figure 2—Study area attributes. “Project” and “CGS” refer to source for landslide data.

Methods

Prior sediment yield estimates were developed using SSA techniques for drainage areas of about 50 mi^2 by the North Coast Regional Water Quality Control Board (NCRWQCB 2001) in the Gualala River Watershed Technical Support Document for Sediment (TSD) and by the California Geological Survey (CGS) for an area of 298 mi^2 (Fuller and Custis 2002). The TSD provided estimates of sediment delivery rates to streams from both natural and anthropogenic sources. The CGS study focused on estimating background (“natural”) erosion rates associated with accelerated soil creep rates to streams adjacent to large rock slides and earthflows. Techniques in this analysis incorporate aspects of the NCRWQCB and CGS methods to estimate sediment yield under existing conditions in four small watersheds (1.2 to 3.1 mi^2, *fig.1*).We substituted detailed local data on road erosion from field surveys, utilized a more detailed map of the stream network developed from field observations, and we added site specific data for landslides collected during field surveys of the project area.

Suspended sediment monitoring using the TTS system (Eads and Lewis 2002) was implemented in late 2005 with assistance from RiverMetrics LLC (Rand Eads, formerly of USDA Forest Service Redwood Sciences Lab).This program was conducted to validate sediment yield estimates based on sediment source assessment techniques and to establish baseline water quality and streamflow conditions. The initial analysis of TTS data for HY 2006 to 07 estimated annual suspended sediment yield (SSY) based on the regression relationships between turbidity-suspended sediment concentration (SSC) data pooled for each year (Eads 2007). A supplemental analysis of these data was based on regression relationships between turbidity and SSC on a storm-by-storm basis (Lewis 2009). In this study, we utilized a regression relationship between turbidity and SSC for pooled data from Soda Springs Creek for HY 2006 and 2007 to estimate SSY for HY 2008 through HY 2011 using turbidity and discharge data.

Sediment source assessment

Sediment sources were classified as background (natural) sources or anthropogenic (management-caused) sources. Background sources were comprised of three elements: rockslide and earthflow soil creep, bank erosion, and “natural” landslides (*table 1*). Anthropogenic sources were comprised of two elements: road-related erosion (including landslides) and erosion from skid-trails and log landings.

Table 1—*Summary of sediment source assessment data sources.*

Erosion Process	Principal Data Source
Rockslide and Earthflow Creep	• CGS landslide maps (Fuller and Custis 2002) • Project area landslide maps (Kleinfelder 2008) • Channel network maps for project area
Bank Erosion	Channel network maps for project area
Natural Landslides	TSD (NCRWQCB 2001)
Road-related Erosion (including landslides)	• Project area erosion rate estimates (PWA 2005) • Road network maps for project area • TSD road erosion rate estimates
Skid Trails & Landings	TSD

Rock slide and earthflow soil creep rates were estimated based on interpretation of literature data (Fuller and Custis 2002). They presented low- and high-range creep rate estimates. These soil creep rates to stream channels were differentiated based on the style of mass wasting and the activity class of the mass wasting feature (*table 2*). Additional data regarding the location and type of landslides (Kleinfelder 2008) supplemented existing data (Fuller and Custis 2002). Sediment delivery from soil creep was calculated as the product of two (representing two stream banks), average bank height of 3.3 ft (consistent with field observations), and stream length intersecting active and dormant mass wasting features (*table 2*) as determined from a GIS database.

Considering uncertainty of methods and assumptions, presenting erosion rate estimates as a range of values maintains an additional measure of caution with respect to interpretation and use of the estimates. In this study, we present sediment yield estimates as a range of values where possible. Low- and high-range rates were calculated for landslide soil creep per *table 2*.

Table 2—*Rock slide and earthflow soil creep rates (after Fuller and Custis 2002, adapted to include mass wasting classification of Kleinfelder 2007).*

Mass Wasting Process	Low-range creep rate (ft/yr)	High-range creep rate (ft/yr)
Historically active earthflow	0.427	0.984
Dormant earthflow	0.033	0.066
Historically active rock slides, slumps, flows and translational slides	0.082	0.164
Dormant rock slides, slumps and slump flows	0.016	0.033
Background creep (ordinary slope conditions)	0.005	--

Methods for estimating road erosion rates used by Pacific Watershed Associates (PWA) for the project area and for TSD field sites were similar. However, project area data are based on direct field observations of 37 miles of project area roads (PWA 2005a, 2005b), whereas the TSD estimates are based on a smaller sample of field plots supplemented by aerial photo interpretation. The mean historic road erosion rate estimated for the project area over the past 30 years was 85 t/mi/yr. The estimated sediment yield from existing roads in Soda Springs Creek is the product of historic erosion rate and road length (*fig. 2*).The TSD estimate is comprised of mass wasting (landslides), stream crossing failures, gullying and surface erosion, and is estimated to be 920 t/mi^2/yr. TSD-estimated sediment delivery from roads for Soda Springs Creek is a function of watershed area. Low-range rates of road-related erosion were those estimated from field surveys in the study area; high-range rates of road erosion were those estimated in the TSD.

Sediment yield measurement by TTS

Hydrologic data used to estimate sediment loads were collected through use of the TTS system developed at the Redwood Sciences Laboratory (RSL), a field office of the USDA Forest Service, Pacific Southwest Research Station (Eads and Lewis 2002). “TTS was designed to permit accurate determination of suspended sediment loads by establishing a relation between SSC and turbidity for each sampling period with significant sediment transport. It does so by collecting pumped suspended sediment samples when pre-selected turbidity conditions, or thresholds, are satisfied. During analysis the relations are applied to the nearly continuous turbidity data for the respective sampling periods to produce a continuous record of estimated SSC. The product of discharge and estimated SSC is then integrated to obtain accurate suspended sediment yields.”

Data collection began in mid-December 2005 at Soda Springs Creek. Full implementation of TTS with SSC sampling continued through May of 2007.The number of samples processed was 285 in HY 2006 and 161 in HY 2007.Continuous stage and turbidity are still being measured using the TTS program as of June 2011, however SSC sampling was discontinued after HY 2007.

Data processing and sediment load estimation procedures

Sediment loads for Soda Springs Creek have been estimated for HY 2006 to HY 2011 using TTS analysis procedures (Lewis and Eads 2009).Raw stage and turbidity data were compiled and adjusted using the TTS Adjuster software developed by the USDA Forest Service Redwood Sciences Laboratory[4]. Adjusted stage values were used to estimate instantaneous discharge, and adjusted turbidity was used as a surrogate for SSC. Discharge is multiplied by SSC to compute sediment yield for each data interval (10 minutes) and summed to estimate the sediment yield for any given period. For HY 2006 to HY 2010 turbidity was the only SSC surrogate used. During HY 2011 there were some periods with questionable turbidity data related to fouling of the turbidity sensor. For these periods, flow was used as a surrogate for SSC. Stage-discharge relationships (rating curves) for HY 2006 and 2007 are the same equations used by Lewis (2009).The rating curve for HY 2008 to HY 2011 represents the most up to date stage discharge relationship as of June 2011.

Annual sediment loads were calculated with R software according to TTS procedures (Lewis and Eads 2009). Sediment loads were estimated using four different regression relationships of SSC on turbidity: 1) simple linear regression: y = ax + b, 2) simple linear regression on the natural logarithms of both variables[5]: log (y) = a + b log (x), 3) power function: y = axb, 4) loess (non-parametric locally-weighted) regression (Cleveland and Devlin. 1988). Each of these models was fitted to three different data sets: HY 2006 only, HY 2007 only, and HY 2006-2007 pooled. The resulting 12 sediment load estimates for each year provide an indication of the uncertainty associated with model selection and interannual sampling variability.

Results

Sediment source assessment

Sediment yield estimates for Soda Springs Creek (sub-watershed of Buckeye Creek) from this analysis (Project) are compared with sediment yield estimates from CGS and the TSD (*table 3*). CGS estimates only the natural background rate associated with soil creep. Nevertheless, the CGS upper range estimate (3,019 t/mi^2/yr) is about twice the TSD and Project (upper range) estimates of 1,400 t/mi^2/yr and 1,688 t/mi^2/yr, respectively. The CGS low-range estimate (994 t/mi^2/yr) is 71 percent of the TSD estimate and 59 percent of the low range estimate for the Project (1,168 t/mi^2/yr).

Comparison between estimates from the TSD and the Project are more relevant given the comprehensive scope and spatial scale of erosion processes considered. The Project estimate is based on some of the same rate estimates used in the TSD. The TSD natural mass wasting rate (170 t/mi^2/yr) is added to rock slide/earthflow creep calculated for the Project. The high range road estimate and the skid trail/timber rate for the Project are from the TSD estimates for Buckeye Creek.

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[4] http://www.fs.fed.us/psw/topics/water/tts/adjuster/AdjusterManual.html.

[5] In this analysis, all logarithms are natural logarithms (base e = 2.718282).The bias of retransformation, i.e., that introduced when transforming log(*y*) to *y*, was corrected using the minimum variance unbiased estimator (Cohn et al 1989).

The Project low-range and high-range estimates are 83 and 121 percent of the Buckeye Creek estimates, respectively. The range of natural background sediment yield estimated for the Project is greater than that estimated by the TSD, 454 to 648 t/mi^2/yr versus 360 t/mi^2/yr. The range of natural background sediment yield for the Project (Soda Springs Creek) is 126 to 180 percent of that estimated by the TSD for Buckeye Creek.

Table 3—*Summary of sediment yield estimates (t/mi^2/yr) using sediment source assessment methods.*

Sediment Source	CGS (Gualala River, 298 mi^2)	TSD (Buckeye Creek, 40.3 mi^2)	Soda Springs Creek (1.53 mi^2)
Natural Mass Wasting	975 – 2,998	170	343 - 537
Stream Bank Erosion	19 - 21	190	111
Road-Related Erosion	--	920	594 - 920
Skid-Trail/Timber Harvest	--	120	120
Total	994 – 3,019	1,400	1,168 – 1,688

Measured sediment yield

Turbidity and SSC data were closely correlated and least-squares fits to the data (e.g., *fig. 3*) were strikingly similar in HY 2006 and 2007.The coefficient of determination (r2) for all models and all data sets is relatively high, ranging from 0.88 to 0.96.For models fitted to the pooled data, r2 ranged from 0.90 to 0.92.

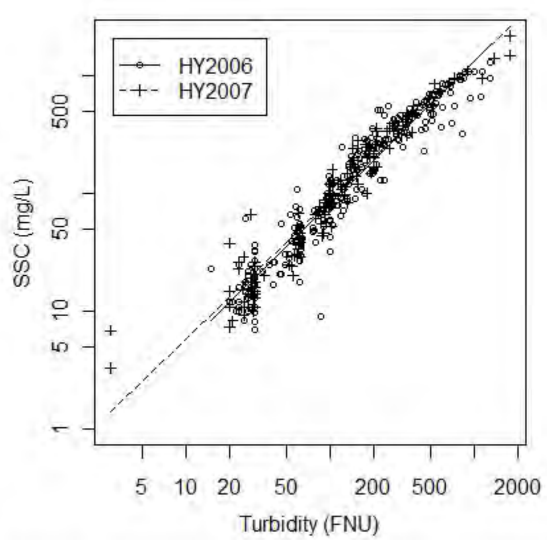

Figure 3—Regression on logs of turbidity and SSC

While an unknown error is introduced in using HY 2006-2007 data to estimate sediment yields for HY 2008 to 2011, the likely magnitude of that error appears to be small. The HY 2006 relationship produced mean annual sediment yield ranging from 186 to 218 t/mi^2/yr, while the HY 2007 relationship produced only slightly higher mean annual sediment yields ranging from 192 to 219 t/mi^2/yr. The loess model (r^2 = 0.92) produced the lowest estimates and the power model (r^2 = 0.91) the highest. The model based on the relationship for pooled data gave mean annual sediment yield ranging from 194 to 216 t/mi^2/yr.

Sediment yield (fig. 4) measured in HY 2006 is by far the greatest during the period of record, ranging from about 620 to 710 t/mi^2/yr. The lowest sediment yield, about 30 t/mi^2/yr, was measured in HY 2009.

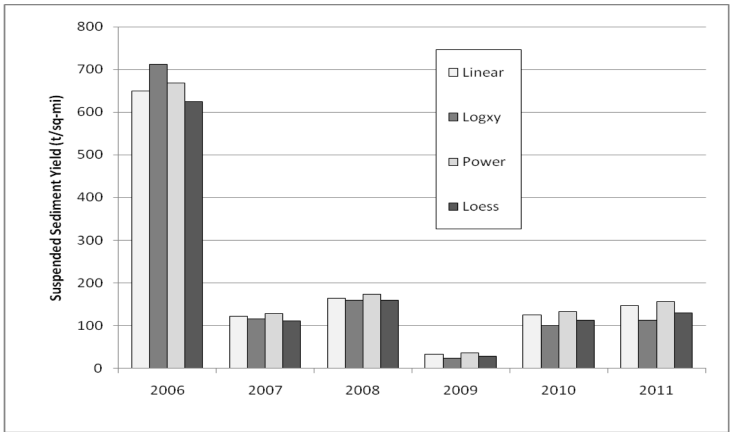

Figure 4—Sediment yield estimates based on HY 2006-07 pooled data regressions.

Discussion

SSA yield estimates substantially exceed measured SSY for Soda Springs Creek. In HY 2006, measured sediment yield (about 620 to 710 t/mi^2/yr) represents about one-half that estimated by the TSD[6]. The HY 2006 measured sediment yield is 37 to 61 percent of estimated yield from the Project SSA (*table 3*). In subsequent years, measured yield is about ten percent of yields predicted by SSA. The difference between SSA-predicted yield and measured SSY is considerably less than the magnitude of uncertainty in measured SSY attributable to sampling and modeling error hypothesized above.

Part of the discrepancy between predicted and observed sediment yield is attributable to unmeasured bedload sediment transport. Bedload (typically sand and gravel) is transported infrequently and over short distances relative to suspended sediment load. The TTS method does not account for bedload sediment. The magnitude of bedload sediment yield can be estimated using regional ratios of bedload to suspended load. Bedload and SSY have been estimated for North Fork Caspar Creek (1.8 mi^2), a watershed with geologic and hydrologic characteristics similar to Soda Springs Creek. Estimates of the ratio of bedload to suspended load range from 15:85 (Napolitano 1996) to 30:70 (Cafferata and Spittler 1998). Based on these ratios, total sediment yield would be about 18 to 43 percent greater than SSY. Total sediment yield for HY 2006 would range from about 730 to 1,020 t/mi^2/yr, a rate that approaches the low-range prediction for Soda Springs Creek (*table 3*). Adjusted for unmeasured bedload transport, mean annual sediment yield for HY 2006 to 2011 ranges from 229 to 308 t/mi^2/yr. Unmeasured bedload does not appear

[6]A portion of HY 2006 (October, November and early December 2005) was unmeasured as monitoring stations were being installed, hence the sediment yield measured for HY 2006 is an underestimate. However, the unmeasured flows represent less than ten percent of flow recorded in that year based on comparison with other regional data.

to be great enough to account for the discrepancy between measured yields and those estimated by sediment source methods.

SSA methods rely on numerous assumptions to develop quantitative estimates of erosion and sediment delivery rates. Field measurements of landslide scarps and gullies can be made with reasonable accuracy, however, the timing of erosion and the proportion of soil delivered to a stream from a source is difficult to determine. Soil creep rates estimated from literature values may represent average conditions, but variability related to geologic conditions and climate is complex. Eroded sediment may be temporarily deposited on hillslopes, in stream channels, and on alluvial fans, awaiting geomorphic events capable of remobilizing it. Erosion processes are episodic, with periods of quiescence punctuated by extreme events associated with large storms and floods in the California Coast Range. Hence it is not surprising that estimates of sediment yield from source assessment methods could diverge from measured sediment yields.

Conclusion

Sediment source assessment methods that estimate mean annual sediment yield do not always provide accurate estimates of actual sediment yields, nor do they account for inter-annual variability related to climate conditions. Measurement of sediment yield using TTS methods allows for validation of sediment source assessments and provides an alternative means of evaluating watershed sediment yield. Field studies involving more direct measurement of erosion and sediment transport processes appear to be required to validate sediment source assessments.

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Fluorometry as a Bacterial Source Tracking Tool in Coastal Watersheds, Trinidad, CA[1]

Trever Parker[2] and Andrew Stubblefield[3]

Abstract

Bacterial counts have long been used as indicators of water pollution that may affect public health. By themselves, bacteria are indicators only and can not be used to identify the source of the pollutant for remediation efforts. Methods of microbial source tracking are generally time consuming, labor intensive and expensive. As an alternative, a fluorometer can be used to measure fluorescence in natural creeks as an indicator of concentrations of optical brighteners found in laundry detergent. In this way, fluorescence can be used as a tool for identifying failing or malfunctioning onsite wastewater treatment systems. Fluorometry was used in watersheds draining into northern California's Trinidad Bay in Humboldt County, in conjunction with bacterial sampling and measurement of rainfall and turbidity for correlation and comparison. Results showed that optical brighteners, when coupled with rainfall and turbidity data, can accurately predict whether bacterial standards will be exceeded within individual watersheds. Fluorescence of optical brighteners was also shown to be a useful tool for discerning the most impacted watersheds and tributaries for more detailed investigation of individual bacterial pollution sources.

Key words: bacteria, fluorometer, optical brighteners, septic, source tracking, wastewater

Introduction

Four types of bacteria are routinely used to assess and regulate water quality in terms of human health: total coliforms, fecal coliforms (a subgroup of coliforms), *Escherichia coli* (a specific fecal coliform) and *Enterococcus*. These bacteria types are found in fecal matter, in the presence of enteric pathogens, they have good survivability in the environment, and are relatively easy to isolate and identify (McCorquodale et al. 1996). Current methods of bacterial testing take 24 to 48 hours to resolve, which is problematic for posting and removing public health warnings in a timely manner (Ely 2006). It is also impossible to determine the source of pollution using just the Most Probable Number (MPN) of the indicator bacteria that most tests provide. Human fecal contamination poses the greatest threat to public health because the associated pathogens are more transmittable. Because the indicators are found in the environment and in the feces of other warm blooded animals, the human health threat is not known even when their numbers are high.

In order to implement actions to improve water quality, the specific source (e.g. pets, livestock, septic systems) and its location, need to be identified. Source tracking is the term for a process used to identify the origin of fecal contamination in water. Unfortunately, most methods currently in use tend to be time consuming, expensive

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[1] This article has been adapted from the thesis of the same title by T.A. Parker, presented May 2011, Humboldt State University, Arcata, CA 95521.

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and / or unreliable and are not practical for small government jurisdictions or volunteer groups to use (Stoeckle 2005). Optical brighteners (OBs) are an emerging chemical method of source tracking human bacterial pollution from septic systems. These organic compounds are added to laundry detergents and other products where they absorb UV light and re-emit that energy in the visible blue spectrum. When added to white textiles, or paper, it makes the objects appear less yellow and more blue, giving them a ‘whiter’ appearance (Hagedorn et al. 2005). OBs are added to 97 percent of detergents sold in the U.S. Because OBs are readily adsorbed to particles, including soil, a properly functioning onsite wastewater treatment system (OWTS), or septic system, should remove nearly 100 percent of any OBs discharged to the system (Dates 1999). In addition, OBs do not readily biodegrade in the environment, unless they are exposed to sunlight (Hagedorn et al. 2005). Recent studies (e.g. Hagedorn et al. 2003, Hartel et al. 2007) show that monitoring OBs with a fluorometer holds promise as a relatively inexpensive source tracking method for suspected bacterial pollution of septic origin (Hartel et al. 2008).

The first goal of this study was to test the viability of using a fluorometer to detect the presence of OBs to confirm and track OWTS pollution in the greater Trinidad area; consideration was given to both spatial and temporal patterns. The fluorometer was used as a source tracking tool both to confirm anthropogenic sources of bacterial pollution and to identify and prioritize areas contributing the greatest amount of bacteria. The second goal of this study was to evaluate the effectiveness of this relatively new, cheap and quick method of identifying and locating human sources of bacterial pollution that can be used by even small municipalities and volunteer monitoring groups and to provide recommendations for its use.

Study area

This study took place in and around the City of Trinidad in Humboldt County, California (*fig. 1*). The study area consists of a pre-defined project area related to water quality and watershed management / planning grants received by the City of Trinidad. The entire area consists of seven watersheds, four of which were utilized in this study – Mill, Parker, Luffenholtz and Joland Creeks – as well as beach and seep samples. The entire planning area contains a total of 6,498 acres and 32.7 miles of stream. This area is characterized by heavily wooded, steep slopes, particularly near stream channels and ocean bluffs, interspersed with flatter plateaus. Baseline water quality monitoring efforts have shown periodic, widespread bacterial contamination as high as 60 times, or 6000 percent of the State’s contact recreational standards[4].

The major land uses in the study area are timber production and rural residential. Much of the lower watersheds consist of residential development with small pockets of commercial and public land, and the upper watersheds are all commercial timberland. The residential development is concentrated around the communities of Trinidad and Westhaven. All residences in the study area utilize onsite wastewater treatment systems (OWTS), including septic systems, to dispose of wastewater (*fig.*

[4] Parker, T. 2008. **Trinidad-Westhaven coastal watershed program: preliminary watershed assessment**. Unpublished report, prepared as part of the requirements for a Proposition 50 Watershed Planning Grant from the State Water Resources Control Board to the city of Trinidad. City of Trinidad, Trinidad, CA.

Figure 1—Project location and vicinity.

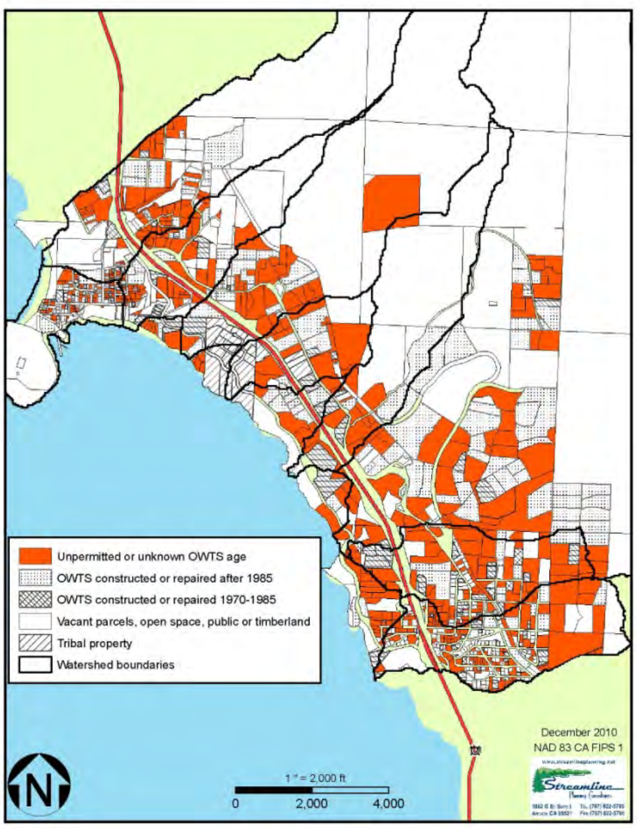

Figure 2—Permit data gathered from the Humboldt County Division of Environmental Health showing age of OWTS or lack of permit data, Trinidad, CA.

2). There are approximately 1,000 developed parcels in the study area. Many are less than ¼ acre in size, much smaller than current regulations require for use of an OWTS. Most of these parcels were created and developed prior to current regulatory standards (*fig. 3*), and therefore the OWTS are often unpermitted (built prior to permit requirements) or otherwise do not meet current requirements (City of Trinidad 2010).

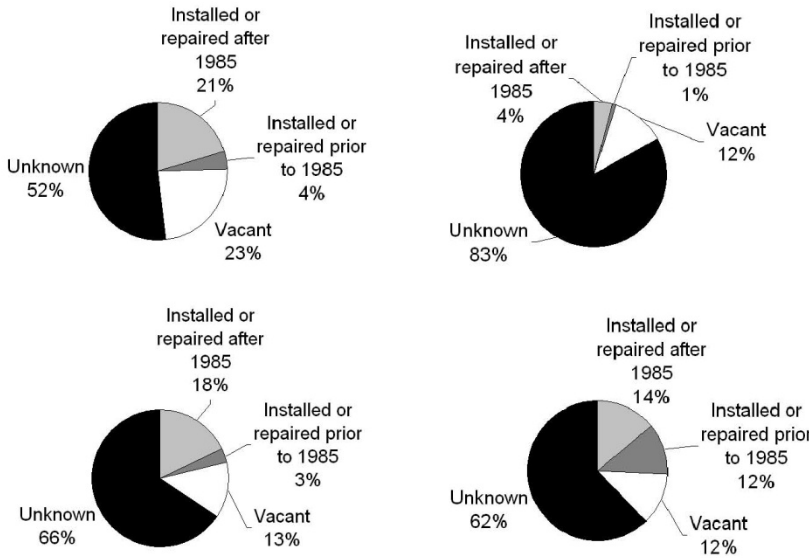

Figure 3—Percentages of OWTS by age in (from left to right, top to bottom) Mill, Parker, Luffenholtz and Joland Creeks watersheds. 'Unknown' indicates there is no file information for those parcels, which means they were installed prior to 1970 or without permits.

Materials and methods

For this study we used an AquafluorTM handheld fluorometer and turbidimeter (Turner Designs, Sunnyvale, CA). The UV-445 wavelength filter was chosen based on the recommendation that it is the most ideal setting (available) for detecting OBs while eliminating other sources of fluorescence such as hydrocarbons and organic matter (Hartel et al. 2007, Turner Designs 2007).

Samples were taken from five different watersheds: beach/seeps, Mill Creek, Parker Creek, Luffenholtz Creek and Joland Creek, on 18 different dates starting on September 17, 2008 and ending April 15, 2010 (*fig. 4*). Samples were taken at various times and locations with varying weather, seasons and conditions in an attempt to get a broad view of data patterns. Field crews started at the mouth of the creeks, or another easily identifiable starting point such as a culvert. Samples were taken approximately every 100 feet upstream, and also at tributaries and other inlets. Bacterial tests included total coliform, *E. coli* and *Enterococcus*. Daily rainfall was collected from records at the city's water plant.

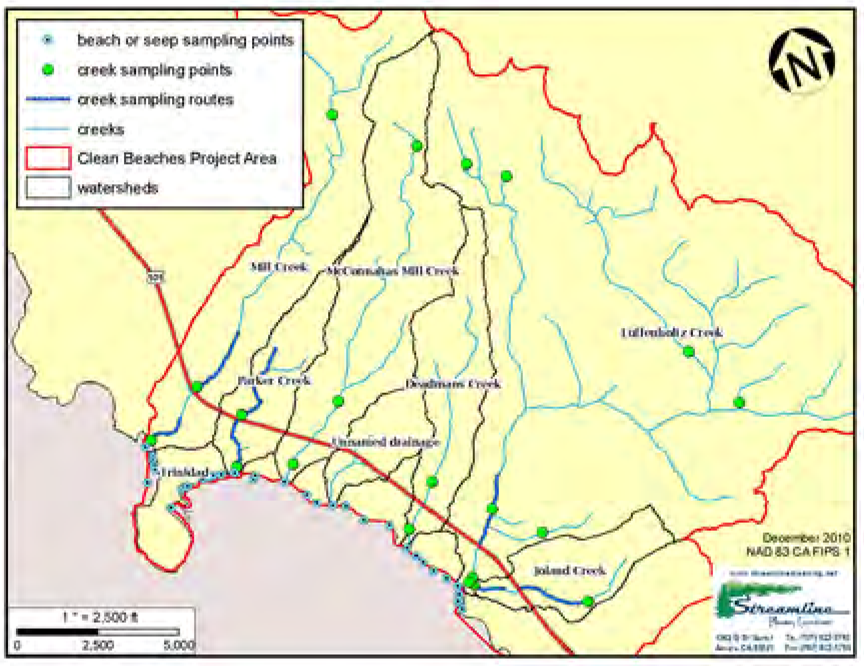

Figure 4—Sampling locations and transects in this study, Trinidad, CA.

All calibrations, sampling, testing and data management occurred according to a quality assurance project plan prepared in accordance with State Water Quality Control Board guidelines. The fluorometer was calibrated with Tide liquid detergent and distilled water. Samples were taken in accordance with Standard Operating Procedures. Measurements for OBs were taken in the field according to the manufacturer's instructions. Turbidity readings were taken immediately following the fluorescence reading for most samples. Bacteria samples were also taken at the same time and delivered to the Humboldt County Public Health Lab.

Data recorded were as follows: sample point identification, watershed, date, OB first reading, OB second reading, total coliform, *E. coli*, *Enterococcus*, turbidity, rainfall in the past 24 hours, rainfall in the past 3 days (72 hours), rainfall in the past 2 to 3 days (24 to 72 hours) and rainfall in the past week. In addition to the directly sampled data, several derived data categories were utilized. This included the difference between the first and a second OB reading (called 'difference') and whether the bacterial results were in exceedance of the state contact recreational standards.

Results

The results showed consistent patterns between sampling events and watersheds with each watershed having a consistent raking for each of the water quality parameters tested (*table 1*).

Table 1—*Median values of variables and their ranks of all data sampled between September 17, 2008 and April 15, 2010 from five watersheds, Trinidad, CA.*

Water-shed	Optical Brighteners		Total Coliform		*E. coli*		*Entero-coccus*		Turbidity	
	Med.	Rank	Med.	Rank	Med.	Rank	Med.	Rank	Med.	Rank
Beach	1.1	5	31	5	1	5	1	5		
Mill	8.0	4	424	4	10	3	6	4		
Parker	17.0	1	1935	1	31	1	121	1	50.8	1
Luffen-holtz	12.6	2	504	3	20	2	41	2	23.6	2
Joland	8.6	3	697	2	10	3	20	3	20.5	3

The most promising statistical results were obtained using logistic regressions to predict if bacterial counts would exceed state standards for contact recreation using OBs, rainfall and turbidity as the independent variables (*table 2*). The bacterial standards are as follows: total coliform 10,000 MPN/100 ml; E. coli 400 MPN/100 ml; and *Enterococcus* 104 MPN/100 ml. These regressions were run with all the data together, starting with the following independent variables – OBs, difference, turbidity, rain 1-day and rain 2-3 days. Total coliform and *E. coli* could not be analyzed, because there were too few bacterial exceedances. For the same reason, logistic regression could not be used on the data for each individual watershed.

Table 2—*Result of logistic regression analysis for the aggregated data from five watersheds in Trinidad, CA sampled between September 17, 2008 and April 15, 2010.*

Independent Variables	Exceedances Correctly Classified	Non-exceedances Correctly Classified	Model
OB 1st[a], Difference, Turbidity, Rain 1-day, Rain 2-3 days	82.4%	96.1%	0.860 - 0.237*Diff + 0.219*OB_1st - 2.14*Rain_1_day + 1.22*Rain_2_3_days - 0.0536*Turbidity
OB 1st, Turbidity, Rain 1-day, Rain 2-3 days	82.4%	96.1%	0.917 + 0.215*OB_1st - 2.13*Rain_1_day + 1.23*Rain_2_3_days - 0.0536*Turbidity
OB 1st, Turbidity, Rain 1-day	76.5%	96.1%	1.73 + 0.194*OB_1st - 2.03*Rain_1_day - 0.0508*Turbidity
Turbidity, Rain 1-day	61.8%	98.0%	3.33 - 0.831*Rain_1_day - 0.0486*Turbidity
Turbidity	58.8%	98.0%	2.92 - 0.0494*Turbidity
OB 1st	38.6%	83.6%	2.80 - 0.170*OB_1st

[a] OB 1st refers to the first optical brightener or fluorescence reading.

Linear regressions between OBs and the three different bacteria measurements essentially showed no direct relationship. Multiple regression was more successful. Based on the results of variable selection routines performed, three of the variables were consistently the best predictors of bacteria, are easy to collect, and provide good power for the multiple regression models: OBs, turbidity and rain 1-day (24 hours). We used these three variables as independent variables in multiple regressions against each of the three bacterial variables for each watershed with good predictive results (*table 3*). We did not run this analysis for the beach / seep dataset due to the lack of both turbidity readings and rainfall. For Mill Creek we used only OBs and rain 1-day since there were no turbidity samples taken.

Table 3—*Results of multiple regression analysis from four watersheds in Trinidad, CA sampled between September 17, 2008 and April 15, 2010.*

Mill Creek	**Model**	**r^2**
Total coliform	104.5+ 40.5*OB_1st[a]+ 28,681.6*Rain_1_day	0.83
E. coli	0.33+ 0.67*OB_1st+ 345.6*Rain_1_day	0.77
Enterococcus	-53.5+ 6.83*OB_1st+ 2,493.2*Rain_1_day	0.80
Parker Creek		
Total coliform	1,393.9-56.6*OB_1st+ 527.1*Rain_1_day+ 9.87*Turbidity	0.48
E. coli	-39.4+ 1.28*OB_1st+ 41.8*Rain_1_day+ 0.78*Turbidity	0.78
Enterococcus	1.41-0.30*OB_1st+ 116.5*Rain_1_day+ 0.82*Turbidity	0.71
Luffenholtz Creek		
Total coliform	NA	
E. coli	12.9-6.74*OB_1st+ 66.5*Rain_1_day+ 2.41*Turbidity	0.59
Enterococcus	46.2-25.2*OB_1st+ 220.6*Rain_1_day+ 7.62*Turbidity	0.68
Joland Creek		
Total coliform	4,687.9-773.1*OB_1st+ 5,109.6*Rain_1_day+ 106.9*Turbidity	0.84
E. coli	-184.6+ 27.1*OB_1st+ 648.3*Rain_1_day- 2.89*Turbidity	0.91
Enterococcus	-92.2+ 7.57*OB_1st+ 836.1*Rain_1_day+ 0.27*Turbidity	0.88

[a] Refers to the first reading of optical brighteners / fluorescence

We graphed the sample results from the longitudinal stream profiles to provide visual clues as to pollution sources within the watersheds. The following chart (*fig. 5*) is one selected example (out of a total of 10 transects). Regardless of the statistical relationships between the variables, these transects may provide a more reliable analog for what is actually happening in the environment. They remove the influence of rainfall, because each transect was taken on a single day under the same conditions. Anomalous points can be looked up in the field notebook for clues as to what is causing the differences.

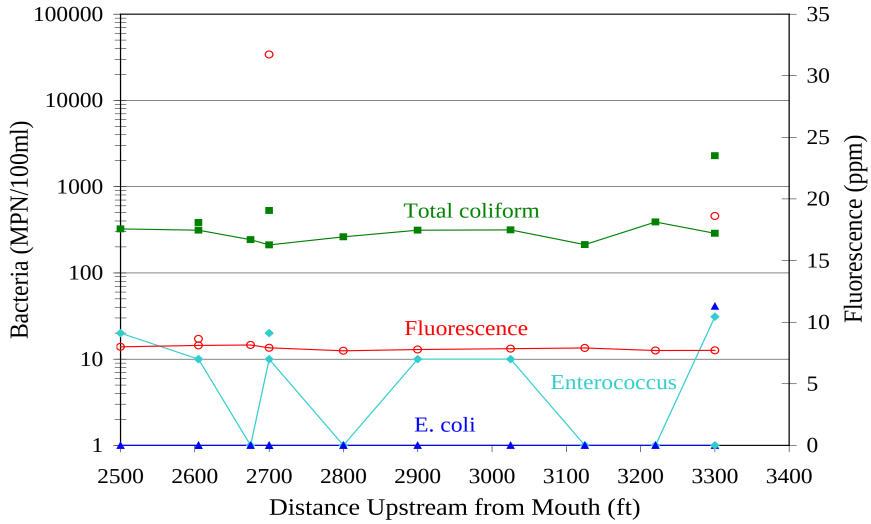

Figure 5—Upstream transect on Mill Creek showing longitudinal trends. These samples were taken on 11/17/08 with 0.0 in of rainfall in the previous three days. Note that points without lines indicate seeps or tributaries draining into the main channel.

Discussion

When OBs were included with rainfall and turbidity as independent variables in multiple or logistic regressions with the bacteria indicators the results showed strong predictive value. The logistic regression analysis showed that these variables can be used to predict exceedances of bacteriological standards (*table 2*). Multiple regression was improved when the data was analyzed separately for each watershed (*table 3*). These results could be used to protect public health by allowing warnings on beaches to be posted immediately after rainfall.

Patterns of OBs and bacteria were consistent with expected priority areas based on land use and watershed characteristics. The results of this study showed that each creek had consistent rankings when considering the median values of each variable (*table 1*). These rankings can be explained by the land use characteristics of each watershed. For example, of the four creeks, Mill had the lowest medians. The upper and lower portions of the Mill Creek watershed are mostly undeveloped timber and park land. It also has a high percentage of undeveloped and open space land, large residential lot sizes and, maybe most importantly, the lowest percentage of old and unpermitted onsite OWTS (*fig. 3*). Similarly, Parker Creek had by far the highest median values and also has the greatest percentage of old and unpermitted OWTS (*fig. 3*). Parker Creek is also densely developed with a high percentage of impervious surfaces and the greatest diversity of land uses. These correlations are strong indicators that the pollution is in fact coming from failing or poorly functioning OWTS. Similar to this study, Hartel et al. (2007) consistently showed high fluorescence and high bacteria when the source was human/residential.

Based on past water sampling results, it was known that higher bacteria counts occur after rain, particularly if there was some prior dry weather. This study confirmed that both bacteria and OBs were influenced by rainfall. Rainfall in the previous day (24 hours) was the best predictor of bacteria counts. Rainfall in the previous 2 to 3 days (24 to 72 hours) was also a useful predictor as was turbidity. However, rainfall also increases the amount of turbidity and organic matter in the water. Since organic matter fluoresces and can interfere with OB readings (Hartel et al. 2007) it may be beneficial to conduct fluorometry based source tracking during summer low-flow periods. Unfortunately, our study found that there was a lack of variability in bacteria counts during low-flow periods, possibly because the samples were diluted at a ratio of one to ten in order to pick up the higher bacteria counts, which are of greatest concern and interest.

The longitudinal creek profiles may provide the best information for source tracking individual failing OWTS or at least the worst contributing neighborhoods or subwatershed. *Figures 5* and *6* show points that form peaks or valleys for one or more constituents. The points can be identified in the graphing program, and then looked up in the field notebook for further information and / or investigated in more detail in the field. Some generalizations can be made when looking at the longitudinal profile data. Points with high levels of bacteria and OBs should be considered priority areas for further investigation of OWTS failures. High bacterial readings with low OBs could either indicate animal sources of bacteria, or an OWTS source that lacks laundry capabilities. High OBs with low bacteria likely indicate a greywater source, or a high concentration of organic matter.

The Mill Creek transect (*fig. 5*) shows sharp spikes of both bacteria and fluorescence in tributaries at 2700 and 3300 ft. The high point at 2700 ft was associated with a seep draining into the creek from an area occupied by an illegal settlement of trailers. Since the time these samples were taken, the County Division of Environmental Health investigated and condemned the site partially based on inadequate sewage disposal. The high point at 3300 ft was taken from a small tributary draining into the main channel at that location. This tributary drains a small residential neighborhood that would be a good candidate for further investigation.

This study helped confirm that OWTS are the cause of bacterial contamination in the study area creeks. This study also showed that a fluorometer was a viable tool in an easy to use form over a wide variety of conditions that is applicable to other government jurisdictions and non-profit organizations that are involved in water quality monitoring, but that do not have the funds for traditional bacterial source tracking. One of the important things this study showed was that the results are much more accurate when considered within a single watershed rather than the entire study area. OBs can provide a quick, cheap and easy method to help characterize different watersheds and to prioritize areas for further investigation and remediation that are contributing the most pollution. Other variables do have to be considered, including the natural variability of the system, seasonality and rainfall.

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Use of BasinTemp to Model Summer Stream Temperatures in the South Fork of Ten Mile River, CA

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Abstract
We used *BasinTemp* to predict summer stream temperatures in South Fork Ten Mile River (SFTMR), Mendocino County. *BasinTemp* is a temperature model that attempts to quantify the basin-wide effects of high summer stream temperatures in basins where the data inputs are scarce. It assumes that direct solar radiation is the chief mechanism behind stream summer heating in mid latitudes. We applied it in the SFTMR basin to understand the dynamics of summer stream temperatures, the influence of timber management and the effects of climate change on water temperatures, and its implications for coho salmon habitat. Three vegetation scenarios (current vegetation conditions, reference vegetation conditions and topography only conditions) and their effects on the Maximum Weekly Average Temperatures (MWATs) throughout the basin were analyzed for 3 different years. We predicted a significant increase in water temperature related to the decrease in shade as vegetation conditions shift from reference to current and to topographic conditions under timber management. We also found that current vegetation conditions account for significant amounts of shade that reduce solar radiation and stream temperatures and that discharge has a growing effect in water temperatures as the basin approaches topographic conditions.

Key words: BasinTemp, riparian shade, stream temperature, model, watershed management

Introduction

The effect of high water temperatures on fish populations is well documented and substantial (Berman 1998). Potential effects of water temperatures may become more pronounced if summer temperatures continue rising as climate change models predict (Kiparsky and Gleick 2003). There is considerable interest in understanding and quantifying the interactions between summer stream heating dynamics, aquatic habitat quality and quantity, and the influence of land management activities on water temperature and its effects on fish populations. Theory supported by considerable empirical research indicates that in mid-latitude regions, direct solar radiation is the most important mechanism driving summer stream heating (Bartholow 2000, Beschta et al. 1987, Brown 1969, Ice 2001), comprising more than 80 percent of the heat budget (Monteith and Unsworth 1990). In these locations riparian shade and topography are the most important controls on the amount of direct solar radiation reaching the water surface (Ice 2001).

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Shading supplied by topography and riparian vegetation during the summer months is considered to be essential for moderating high stream temperatures (Bartholow 2000, Beschta et al. 1987, Brown 1969, Ice 2001, Johnson and Jones 2000, Rutherford et al. 1997); particularly in small, low-order streams that are highly sensitive to changes in the thermal environment due to the relatively low heat carrying capacity of these streams.

BasinTemp, developed by Allen and Baker in 1999, addressed the need to find a simple way to predict high summer stream temperatures at a watershed scale. Since BasinTemp model was presented at the 2004 Redwood Region forest science symposium and its methods were discussed in a paper by Allen et al. 2007 in the symposium's proceedings, we will only refer to them briefly here. BasinTemp is a mechanistically based model which assumes that direct shortwave radiation drives stream summertime temperatures, emphasizing the role of the riparian vegetation in the control of insolation and water temperatures. It is composed of several modules including (1) a GIS Pre-processor that assembles a channel network, elevations and vegetation heights; (2) a radiation model that generates daily-averaged, spatially explicit shortwave radiation predictions ($W/m^2/day$) from the above data; (3) a simple hydrology model that assumes that groundwater accretion is a linear function of drainage area and estimates low wetted widths from empirical values; (4) a simple 1D heat balance model which uses solar radiation, discharge, groundwater accretion and hydraulic geometry values to predict MWAT values for the whole basin; (5) an optimization routine which improves model predictions using field measured temperature data derived from a number of thermistors deployed in the watershed.

BasinTemp was applied to the South Fork Ten Mile River (SFTMR) Basin to confirm the effectiveness of BasinTemp in a current vegetation conditions scenario, to assess and quantify spatially explicit summer stream temperatures in the basin and more explicitly to study the influence of topography, vegetation patterns, climate change and the effect of land management on these and on stream temperatures. The study attempts to answer the following questions: (1) What is the role of shading by riparian vegetation in controlling summer stream temperatures? (2) How does the importance of riparian shade vary across the basin, and if so? (3) How do different riparian vegetation conditions affect stream temperatures locally and downstream? (4) How might changes in regional climate patterns affect stream temperatures in the South Fork Ten Mile River Basin?

Study Area

The SFTMR is a small sub-watershed (100 km^2) in the Ten Mile River Basin, 13 km north of Fort Bragg, CA (*fig. 1*), with elevations that range between 6 m and 950 m. It has four main tributaries, three of them coming in from the right (Smith, Campbell and Redwood creeks) and one from the left (Churchman Creek).The climate in the SFTMR basin is characterized by two distinct seasons: a hot, dry summer and a cool, wet winter. This is modified by the effect of the coastal fog which towards the ocean, moderates summer temperatures (Albin and Law 2006). Temperatures range between 12°C at the coast to 30°C inland.

We subdivided the watershed into three main zones depending on the reach and influence of the fog layer: (1) a coastal or fog zone, extending approximately 4 km inland, where August average maximum daily air temperatures are below 20°C; (2) a transition zone, extending from 4 km to 9 to 10 km from the coast, where temperatures increase from 20 to 26°C; (3) an inland zone, extending farther than 9 to 10 km from the coast, where temperatures are predominantly over 26°C. Results from the fog zone were discarded in the optimization and in the validation of this study because the model is inappropriate for conditions where incoming shortwave radiation is no longer the dominant heat energy flux (Allen 2007).

The geology is comprised of homogeneous Franciscan Coastal Belt composed of greywacke sandstone, shales, siltstones and conglomerates and has relatively small influence in the model output due to its homogeneity. Redwood (*Sequoia sempervirens*) in the coast and Douglas-fir (*Pseudotsuga menziesii*) inland are the dominant tree species in the basin.

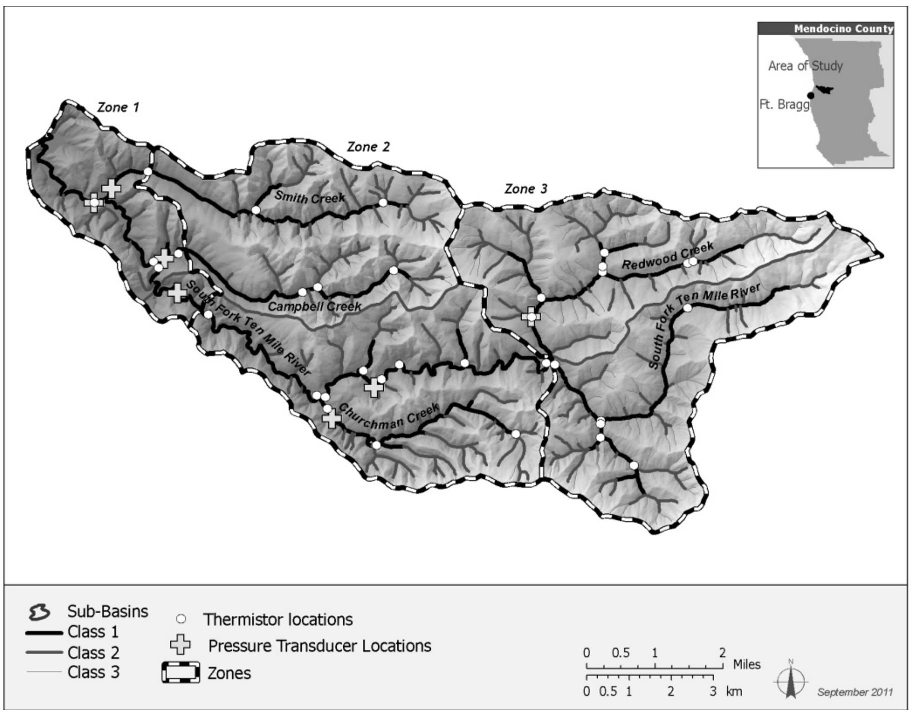

Figure 1—Study area.

Currently all of the SFTMR basin is privately owned. Over 90 percent is owned by Campbell Timber Management. The watershed was heavily logged between 1910 and 1940 and has been recovering by natural means since 1930, although in 1945 a wildfire burnt most of the inland part of the watershed to the south of Smith Ridge to the north and to the west of Sherwood Ridge delaying its recovery substantially (Ambrose et al. 1995). This can be seen today as late seral stages dominate the western half of the basin while early seral stage stages dominate the eastern half. Native fish species include Chinook salmon (*Oncorhynchus tshawytscha*), steelhead (*O. mykiss*), coho salmon (*O. kisutch*), and sculpin species (*Cottus spp*.). This study focuses on the effects on the temperature sensitive juvenile life-stage of coho salmon, which are listed as endangered by both the federal government and the State of California. The input data used in the study are shown in *table 1*. The basic stream channel network was provided by Campbell Timberland Management (CTM). It was expanded to include headwater swales and split into 30 m segments which constitute the minimum unit for which MWATs, solar radiation, discharge and wetted widths are modeled.

Three scenarios were tested: (1) topographic conditions, representing the worst case scenario assuming no vegetation anywhere and topography derived shade only; (2) current vegetation conditions representing shade derived from topography plus vegetation heights as of 2006; and (3) reference vegetation conditions representing shade from undisturbed riparian conditions and assigning late seral tree heights to the riparian vegetation.

Table 1—*Data types used for BasinTemp in the SFTMR basin.*

Data type	Data used for SFTMR
Stream network	Modified USGS 1:24,000 DLG hydrography
Topography	USGS 10m DEM
Tree heights	Landsat TM imagery classified according to the California Wildlife Habitat Relations (CWHR) system (Fox et al. 1997)
Channel geometry	Power-law relationship between drainage area and field measured low-flow widths throughout the SFTMR basin
Low flow discharge	7-day mean daily discharge data from pressure transducers set throughout the basin by CTM and from the Noyo River USGS gage near Fort Bragg
Observed stream temperature data	2006, 2007 and 2010 thermograph data compiled by CTM

Absolute tree heights are obtained from combined topographic data and estimated tree heights derived from vegetation information (Allen 2007). Topographic elevations were extracted from a USGS 10 m DEM and were used with no modification in the topographic conditions scenario. Estimated vegetation tree heights were added to the 10 m DEM elevations for the current and reference vegetation condition scenarios.

For current vegetation conditions tree heights were generated using vegetation data from 1994 Landsat Thematic Mapper (TM) imagery classified according to a modified California Wildlife Habitat Relationships (CWHR) classification scheme (Fox et al. 1997) where DBH-height relationships were established for current and reference condition vegetation as shown in *table 2*. These height were validated by 2005 CTM stand data summarized into average tree heights and if within 90 m of any streams, updated with 2005 1 m National Agricultural Imagery Program (NAIP) color orthophotography resampled in several steps to 30 m.

Water temperature data was provided by CTM for 2006, 2007, and 2010. Thermostats were evenly distributed throughout the basin and located upstream and downstream of major confluences. Of these 3 years, 2007 was the most typical of hot and dry conditions while 2006 was an abnormally hot, useful for estimating temperature increases that might result from climate change. The year 2010 was the most representative of average temperature and discharge conditions (*fig. 2*). As mentioned above, to calibrate the model, sites in the fog zone (Zone 1, *fig. 1*) were removed from the analysis. Hobo™ data loggers were used to measure actual stream temperatures throughout the basin for 3 years. Thirty two sites were studied in 2010, twenty two in 2007 and twenty in 2006. They recorded temperatures at 15 minute intervals which were combined into daily averages. These were calculated into 7 day moving averages or average weekly average temperatures (AWATs). The maximum AWAT in the year (MWAT) was extracted for each station and the most common MWAT value for each station was defined for every year: July 27th for 2006 and 2007 and July 21st for 2010. Wetted widths were measured in August of 2006 and 2007 and extrapolated to 2010 due to the lack of measurement. July monthly average maximum air temperatures, 21°C for 2006 and 2007 and 15.8°C for 2010, were derived from 4 km Oregon State University PRISM grids. A single basin-wide average was used as an input parameter in the heat transfer model.

Table 2—*Diameter at breast height (DBH) to tree height relationships for CWHR vegetation assemblages.*

DBH range (inches)	**Tree height (m)** Mixed Hardwood	Mixed Pine	Mixed Fir	Mixed Conifer and Hardwood	Mixed Oak Woodland	Mixed Hardwood and Conifer	Mixed Conifer
6-Jan	10	7	7.5	7.5	5	10	10
11-Jun	15	10	15	15	10	15	15
24-Nov	20	17.5	20	22.5	15	20	25
> 24	25	25	30	30	20	27.5	35
> 36[a]	-	-	-	-	-	-	45

[a] Only applies to the Mixed Conifer class.

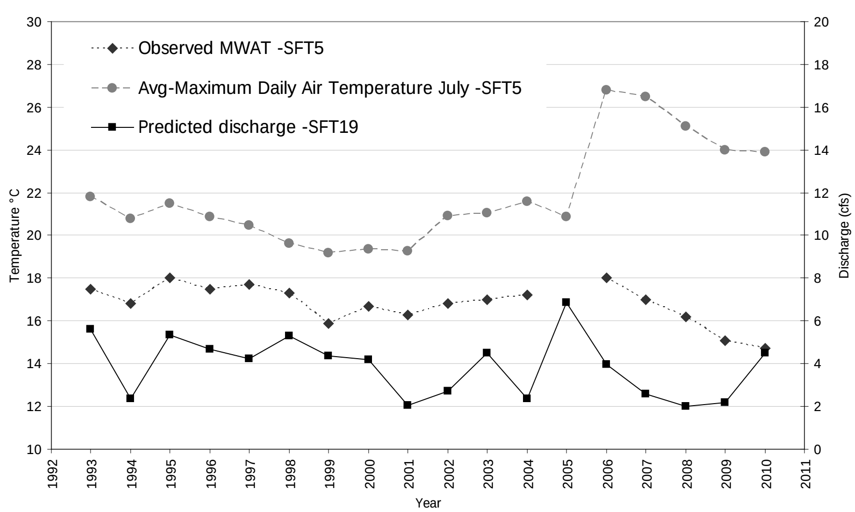

Figure 2—Long-term water temperature (MWAT), air temperature (average maximum daily air temperature from PRISM) at SFT5, and estimated discharge for site SFT19 in the South Fork Ten Mile basin, 1993 to 2010.

In 2006 and 2007 discharge values at SFT19, the lowest station in the watershed, were 3.7 m^3/s and 2.7 m^3/s respectively, some of the lowest values since 1993 (*fig. 2*). The calculated 2006 and 2007 accretion rates were 0.000414 and 0.00030571 m^3/km respectively. With no 2010 discharge data in the SFTMR basin, we estimated the discharge at SFT19 based on values from the USGS gage in the Noyo River near Forth Bragg and in their high correlation with SFT19 discharge in other years. Discharge for 2010 at SFT19 was estimated to be 4.5 m^3/s and the accretion rate 0.000504 m^3/km.

Results

A high correlation between observed and modeled results was observed in 2010 (average water year) with a root mean square error (RMSE) = 0.3°C and a R2 = 0.89). 2007 (dry water year) had a RMSE of 0.59°C and an R2 of 0.67. Model parameters derived from these best-fit relationships were later applied for subsequent temperature prediction scenarios (reference vegetation conditions, and topographic shade-only). We assume the higher correlation and lower RMSE in the 2010 data to be a result of a lower temperature range in the observed temperatures in 2010 (2.9°C) than in 2007 (3.1°C). The better placement of the thermistors after several years of field surveys is possibly another reason for the better fit.

The spatial distribution of predicted MWATs closely tracks the observed MWAT temperatures. Warmest predicted MWATs relative to drainage area are generally observed in the eastern portion of the basin in upper Redwood Creek and the Upper South Fork Ten Mile River near the confluence with North Fork Redwood Creek (*fig. 3*). Downstream of that confluence, temperatures warm up to 17°C, over the preferred range for coho salmon, during the dry water year and remain at 17°C until the river reaches the fog zone. Temperatures for the average water year (2010) warm up in the mainstems of Redwood Creek and South Fork Ten Mile River in the inland zone but at 15°C stay within the preferred range for coho and remain at close to 15°C until the waters reach the fog zone where the water cools down. This warming is apparently due to the combined influence of warm July air temperatures and reduced shading provided by early seral-stage vegetation. A net water loss of the reach (personal observation) may be another cause of the warmer temperatures in the reach. Overall, less than 1 km of stream length has temperatures higher than the preferred range for coho salmon in the dry water year, while all predicted temperatures are below 15.5°C in the average water year.

Upstream of Churchman Creek (drainage area 10.3 km^2 (*fig. 3*), tributaries draining into the mainstem South Fork Ten Mile below its confluence with Redwood Creek are all small and hence have relatively small thermal dilution effects. Water temperatures cool downstream of the Churchman Creek confluence, where temperatures less than 13 or 15°C (for average or dry water year types, respectively) empty into the mainstem South Fork Ten Mile River.

MWAT temperature predictions for shade provided by topography only were generated for average and dry water year data. Under current vegetation conditions, the average water temperatures are 0.9°C and 1.3°C (average and dry water years, respectively) cooler than under topographic-shade-only conditions. Water temperatures in the headwaters start out cool at less than 13°C due to cold groundwater temperatures, but due to the lack of vegetative shading heat loading is high, resulting in temperatures predicted to exceed 18°C in the mainstem South Fork Ten Mile River for all of Zone 2 and much of Zone 3 in 2007. In 2010 there is less heating in the mainstem. Temperatures reach 17°C below the confluence of South

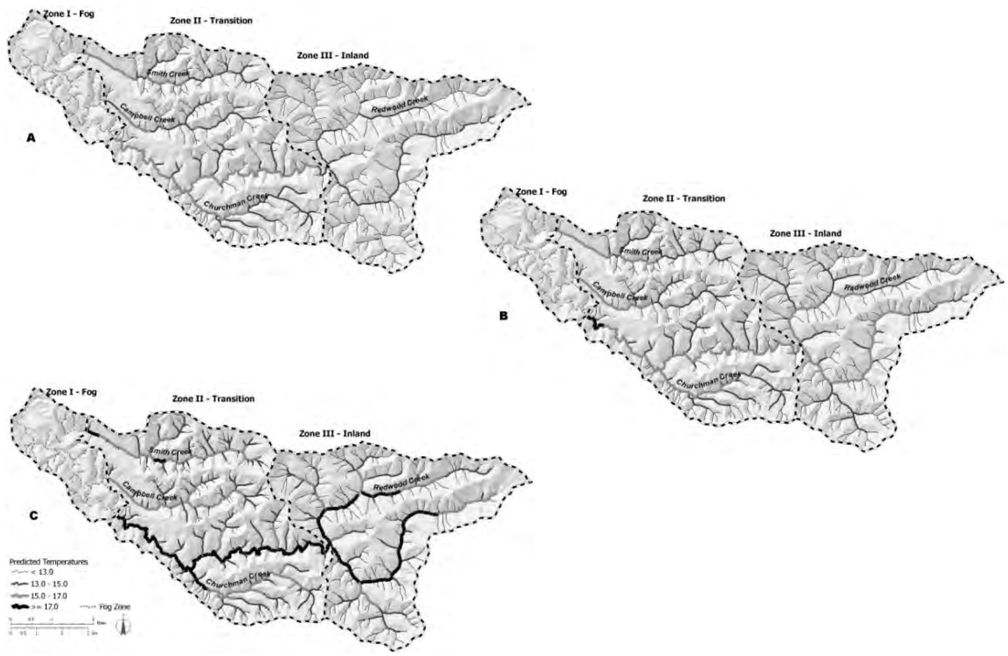

Figure 3—2010 MWAT predictions for SFTMR for (A) reference vegetation, (B) current vegetation and (C) topographic conditions.

Fork Ten Mile and Redwood Creeks, increasing until 18°C below the confluence with Churchman Creek to cool down when we enter the fog zone. Although the difference with temperatures observed under current conditions is smaller that in the dry water year, observed temperatures are still higher.

The year 2006 was considered to be a climate change scenario. Current climate change models for northern California (Cayan et al. 2006) predict a trend of increasing summer air temperatures. Total annual precipitation is predicted to remain similar to current conditions, with most rainfall continuing to occur during a relatively short winter season. Summer air temperature predictions range from increases of about 2 to 5°C over the next 50 to 100 years depending on the scenario. To predict water temperatures based on climate change models, we modeled shade under current and reference conditions based on data collected during a record-breaking heat wave event that was experienced between July 16 – 26, 2006. The water temperature dynamics during summer 2006 were very similar to what is predicted for northern California climate change. This scenario predicted that over 27 km of stream channel would have temperatures greater than 17°C under current vegetation conditions, and over 7 km under reference vegetation conditions. Based on our results, if these climate models are accurate, we predict a trend of increasing MWATs in the South Fork Ten Mile River, regardless of land management actions. Under these conditions, more of the mainstem South Fork Ten Mile River will become too warm for coho salmon, and cooler tributaries, such as Churchman and Smith creeks, and areas located downstream within the cooler coastal fog zone, will likely play a heightened role in overall salmon productivity. Precisely how exceptional this period was and how it affected water temperatures in the South Fork Ten Mile Basin can be seen by comparing MWATs recorded in 2006 compared with average MWATs recorded at the same gages between 1993 to 2004 (*fig. 2*). Observed MWATs for 2006 gages were all warmer than MWATs in previous years, some significantly so.

Discussion

The study shows an expected rise in average basin wide MWATs with an increase in air temperatures (See *table 3* and *fig. 2* to compare the 3 year's average air temperatures). Predicted MWATs also increases regardless of the scenario, 2.3 °C under current and reference vegetation conditions and 2.0 °C under topographic conditions. We found that MWATs increased 0.4°C from reference to current conditions in 2006 and 2010 and 0.8°C in 2007 and 0.6°C, 0.9°C and 1.3°C between current vegetation conditions and topographic conditions in 2006, 2010 and 2007 respectively. Even though 2006 was an extremely hot year (*fig. 2*), the difference in predicted MWAT values is smaller than in 2007, a cooler year but with higher discharge.

Table 3 shows the percentage of the stream network outside the fog zone with MWATs under 15°C. Over 90 percent of the stream channel network is within the preferred range of coho salmon (15°C MWAT) for reference vegetation conditions in 2007 and 2010, and 74 percent in 2006. The decrease in the percentage of channel network under 15°C as we depart from the reference conditions is 16 percent in 2006 (the climate change scenario) and 8 percent in 2010 (the average water year). MWATs under current vegetation conditions are closer to MWATs under reference vegetation conditions (best case scenario) than to topographic conditions (worst case scenario), indicating the positive effects of current land management conditions in maintaining low stream temperatures.

Table 3—*Average predicted MWATs (°C), percentage of total stream length and percentage of Class I stream length with MWATs under 15°C in zones 2 and 3 for topographic, current and reference vegetation conditions in years 2010, 2007 and 2006.*

Year	Topographic Conditions	Current Vegetation Conditions	Reference Vegetation Conditions
2010	14.2°C / 92% / 68%	13.3°C / 100% / 99%	12.9°C / 100% / 100%
2007	15.7°C / 73% / 5%	14.4°C / 83% / 35%	13.6°C / 90% / 62%
2006	16.2°C / 58% / 1%	15.6°C / 68% / 20%	15.2°C / 74% / 23%

Table 3 also shows the percentage of the stream network outside the fog zone with MWATs under 15°C, but this time constrained to the mainstem SFTMR and main tributaries as defined by STREAM CLASS = 1 (*fig. 1*). This query extracts the main channels from the boundary of the fog zone to the upper extent of the fish bearing streams allowing us to see the impact of tree shade on the channel network's aquatic reaches. The reduction of channel length under 15°C is concentrated in the main channels rather than in the headwater channels and it is more pronounced as the air temperature increases. We find that 68 percent of the main channels outside of the fog zone have MWATs less than 15°C in an average water year (2010) even in topographic conditions while only 1 percent have in the climate change scenario reinforcing the importance of vegetation shade in the climate change scenario.

The effects of vegetation shading on the main channels are reduced substantially with higher air temperatures. In the climate change scenario (2006) only 1 percent of the channels are under the 15°C threshold in the climate change scenario under topographic conditions and 23 percent under reference conditions. But a small decrease in air temperature or increase in discharge (2007) results in a significant increase in the length of main channel under the 15°C MWAT to 35 percent in current conditions and 62 percent in reference conditions. As the air temperature increases the importance of the role of the vegetation in the reduction of the temperatures also increases. It is interesting to observe how current land management practices maintain main channel MWATs closer to reference conditions than topography conditions.

BasinTemp has been used as a tool in this study to predict basin-wide temperatures in the SFTMR with a few, simple parameters and with a limited field effort. We consider the model to be a valid tool to get a good first estimate of a basin's temperature and complementary to reach-based temperature models. RMSE's for the 3 years studied ranged between 0.3 and 0.6°C.

We found land management in the SFTMR basin to keep the riparian vegetation in conditions that are closer to reference heights than bare ground conditions thus helping current MWATs to stay below 15°C in an average water year (2010) and in most reaches in a dry water year (2006). In the 3 years shown here, current temperatures are closer to reference than to topographic conditions.

With regard to the main four questions stated in the introduction we found that: (1) Riparian shading in the SFTMR reduced summer stream temperatures in all 3 years studied. This reduction of predicted MWATs in the whole basin was greatest in 2007 and 2006 and we found that riparian shading effects were reduced in 2010 due to higher discharge and lower air temperatures (*table 3* and *fig. 2*); (2) Riparian shade is important throughout the basin. Shaded lower order streams feed colder water to higher order tributaries reducing water temperatures in these reaches. Riparian shade has greater effect in reducing MWATs in Class I streams as it lowers stream temperatures and their propagation downstream (*table 3, fig. 3*); (3) Reference vegetation conditions help keep stream temperatures low in upper stream reaches feeding colder into larger tributaries and reducing the propagation of high temperatures downstream in Class I streams reducing temperature impact on salmonids. Current land management practices help maintain riparian shade effects closer to reference vegetation conditions than topographic conditions keeping most of the channels under 17°C; and (4) Predicted climate change scenarios leading to higher air temperatures and reduced flows are expected to reduce the influence of vegetation shade in the control of stream temperatures increasing stream temperatures.

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Comparing Hydrologic Responses to Tractor-Yarded Selection and Cable-Yarded Clearcut Logging in a Coast Redwood Forest[1]

Leslie M. Reid[2]

Abstract

Initial increases in dry-season flow after selective logging of second-growth coast redwoods (*Sequoia sempervirens*) in the 424 ha South Fork Caspar Creek watershed disappeared by 7 years after logging ended, and low flows then dropped to below expected values for the next 20 years. During the 16 years after clearcut logging in the 473 ha North Fork watershed, late-summer flows increased to nearly twice those expected and then declined to pre-treatment levels on a trajectory that suggests further decline is likely. This contrast in dry-season flow responses is consistent with expected differences in post-logging recovery rates for transpiration after selective and clearcut logging. The South Fork showed a delayed peakflow response relative to that in the North Fork, and a maximum 3 year increase (per unit area of clearcut equivalent) about 40 percent lower. South Fork peaks remained slightly elevated for more than 20 years after logging ended, and North Fork peaks remained elevated for at least 12 years after logging.

Keywords: cumulative watershed effects, logging, low flow, peakflow, recovery

Introduction and study site

Studies at different sites often provide conflicting information about the hydrologic effects of logging (e.g., Moore and Wondzell 2005), indicating that different settings, forest types, or silvicultural strategies may produce different outcomes. Silvicultural preferences shift through time, and the extent to which past experimental results apply to new management strategies is often unknown. Uneven-age management is again being used in some coast redwood (*Sequoia sempervirens*) forests instead of clearcutting, so it would be useful to know how hydrologic responses to selective logging and clearcutting compare. This study evaluates low flow and peakflow responses to these silvicultural strategies during two watershed experiments.

The Caspar Creek Experimental Watersheds (N39°21´ W123°44´; *fig. 1*) are underlain by sandstone and shale of the Coastal Belt of the Franciscan Complex. Rainfall averages 1170 mm/yr and about half reappears as streamflow. Snow is uncommon. October through May account for 95 percent of the rainfall, and annual minimum flows generally occur in early October. By 1962, the old-growth forests logged in 1865 to 1905 had regrown into mature stands dominated by coast redwood

[1] This paper was presented at the redwood science symposium: coast redwood forests in a changing California. 21-23 June 2011, Santa Cruz, California.

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and Douglas-fir (*Pseudotsuga menziesii*). The 424 ha South Fork Caspar Creek watershed was selectively logged during the first of the two experiments, and the 473 ha North Fork watershed was partially clearcut during the second. Similar forest types at similar sites were thus subject to different silvicultural strategies, allowing comparison of their hydrologic responses.

However, the two studies used different experimental designs, produced different kinds of data, and weathered different storms. Because the control watershed for the first experiment became the treatment watershed for the next, post-logging results for the first study spanned only 12 years. Previous reports (described by Ziemer 1998) presented short-term North and South Fork results. To compare responses over a longer period, the current analysis required development of methods to permit tracking of South Fork responses after logging began in the North Fork.

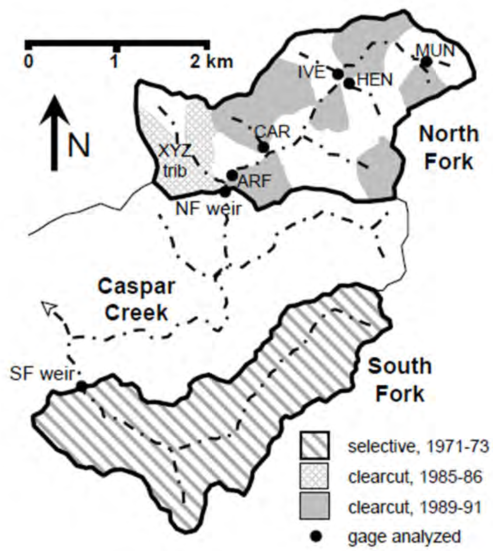

Figure 1—North and South Fork Caspar Creek watersheds.

Methods

The South Fork experiment began in 1962 with 4 years of monitoring to calibrate relations between measurements at the North and South Fork gaging weirs. More than 4 km of mainline road were constructed along the South Fork riparian zone in 1967, and the watershed was then selectively logged from 1971 to 1973 using tractors, with about 2/3 of the timber volume removed. Treatment effects were evaluated until 1985, when logging began in the control watershed.

Thirteen new stream gages were installed in the North Fork watershed in 1984 and 1985, with three gaged subwatersheds selected to be left unlogged as controls. XYZ tributary enters the head of the North Fork weir pond and was not a part of the second study; 60 ha of XYZ were clearcut in 1985 and 1986. Hydrologic recovery could not be assured by then at the South Fork, so neither weir record could be compared to that from a calibrated control after 1985. The second experiment instead compared records from 10 of the new stations to those of the control tributaries (HEN, IVE, and MUN). Calibrations were established from 1985 to 1989, then 37 percent of the watershed was clearcut in 1989 to 1992, with selectively logged buffer strips left along channels draining more than about 10 ha. Cut logs generally were cable-yarded to ridge-top roads. About 20 percent of the watershed was burned after logging, and most units were pre-commercially thinned a decade later.

Rainfall has been monitored near the South Fork weir since 1963 using a series of weighing and tipping-bucket gages, but dry-season data were not collected consistently until 1972. The National Weather Service recorded rainfall for most of this period 12 km north in Fort Bragg. Summer storm totals of >0.6 mm at the Fort Bragg and South Fork (S620) gages are strongly correlated for 1989 to 1999 ($Rain_{S620} = 1.03\ Rain_{FtBragg} + 0.73$ mm, $r^2 = 0.89$), but smaller events often reflect coastal fog and are less common at S620. This study combines the daily rainfall measured at S620 for October through May (accounting for 98 percent of the annual rainfall) with Fort Bragg data for daily rainfalls >0.6 mm in June through September to construct a continuous rainfall record from September 1962 through August 2008.

The North and South Fork gaging structures are concrete sharp-crested compound weirs. Weir pond stages have been measured since 1962 in stilling wells, first using strip-chart recorders and then data-logged pressure transducers, and flow is calculated from pond stage using standard equations. Flows are monitored in Parshall flumes or rated sections at each upstream or tributary station by using data loggers to record stage at 10-minute intervals. Most tributaries run dry by late June, so dry-season flow analyses are based on data from the North and South Fork weirs.

Keppeler (1998) evaluated changes in low flow after North and South Fork logging, and her data suggest that South Fork flows may have dropped below pre-treatment levels by the late 1980s. However, results were complicated by the lack of a control record after the onset of North Fork logging. Because summer rainfall is minimal, late-summer flows are sustained primarily by moisture stored from winter storms. The current study thus uses relations between runoff and antecedent precipitation indices (APIs) to calculate expected flows directly from rainfall, allowing flow deviations to be estimated after the control record ends.

Periods in August and September were identified that had no rain in the preceding 3 days and <9 mm in the preceding 30 days. Mean daily flow was tabulated for 3 to 5 such days each year, with dates selected to be at least 6 days apart and to have records for both the North and South Forks, if possible. Records were not used during periods when weir ponds were drained to remove sediment. APIs were calculated for each selected date using a range of coefficients (0.99 to 0.60). The selected North Fork flows between 1963 and 1984 were regressed against each of the API sets, and a recession coefficient of 0.977 was found to best predict flows:

$$L_N = 0.0272\ API_{977} + 0.0366 \qquad r^2 = 0.87 \qquad n = 46 \qquad (1)$$

where API_{977} (mm) is calculated as 0.977 times its value on the previous day plus the current day’s rainfall, and L_N (L km^{-2}s^{-1}) is the mean discharge for the day at the North Fork weir (*fig. 2A*). The same approach was used to develop a predictor for expected South Fork flows (L_{S1}, L km^{-2}s^{-1}) based on data from 1963 to 1970 (*fig. 2B*), with API_{985} calculated using a coefficient of 0.985:

$$L_{S1} = 0.0143\ API_{985} - 0.0320 \qquad r^2 = 0.94 \qquad n = 22 \qquad (2)$$

An equation was also developed to predict South Fork flows from those observed at the North Fork (L_{NO}, *fig. 2C*):

$$L_{S2} = 1.52\ L_{NO} + 0.0706 \qquad r^2 = 0.91 \qquad n = 22 \qquad (3)$$

An analogous regression using expected North Fork flows produced a relation not significantly different from equation 3, allowing expected South Fork flows to be calculated from either observed or expected North Fork flows.

August and September South Fork flows expected in the absence of logging can now be estimated using either equation 2 or equation 3, and results can be compared to observed values to evaluate long-term patterns of deviation after selective logging. Similarly, North Fork deviations can be calculated by comparing measured values with those predicted from rainfall records using equation 1.

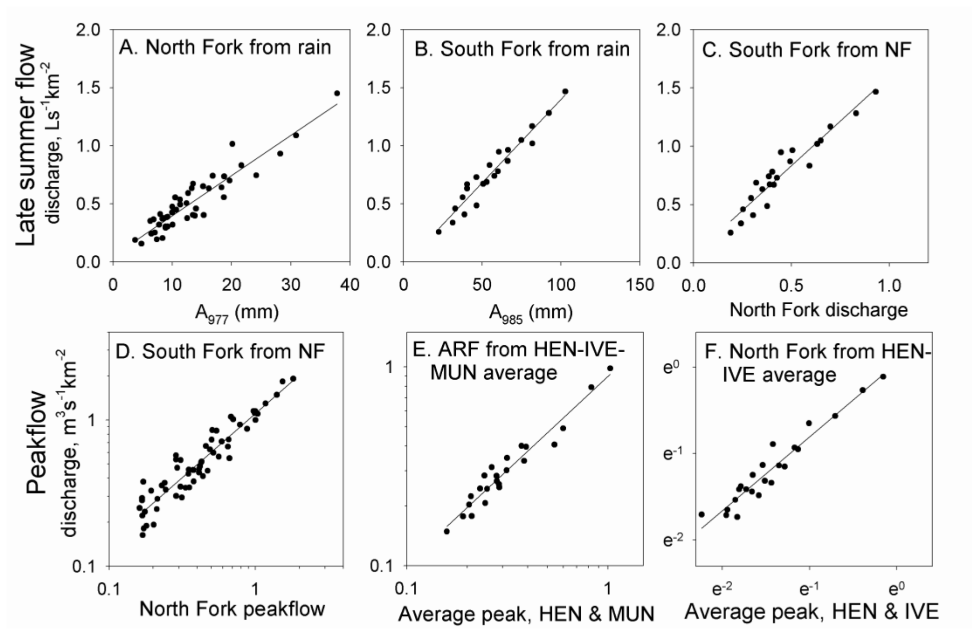

Figure 2—Calibrations between A. North Fork late-summer flows and API_{977}; B. South Fork late-summer flows and API_{985}; C. South Fork late-summer flows and those observed at the North Fork; D. South Fork and North Fork peakflows; E. ARF peakflows and the average of those at HEN, IVE, and MUN; and F. North Fork peakflows and the average of those at HEN and IVE.

Past analyses of South Fork peakflows compared pre-logging correlations to North Fork peaks with those from between the onset of logging and either 1 (Ziemer 1981) or 2 years (Wright et al. 1990) after logging ended, thus including 3 years of logging. No significant increase was found for moderate to large peaks. However, results of the North Fork study later showed that effects of logging may not appear until the following wet season (Lewis et al. 2001), so inclusion of data collected during logging may increase variance enough to hinder detection of responses.

The present peakflow analysis incorporates all peaks higher than 0.16 $m^3s^{-1}km^{-2}$ in the North or South Fork. Pre-logging South Fork peaks were first regressed against observed North Fork peaks (P_{NO}, $m^3s^{-1}km^{-2}$) from 1963 through 1971 (*fig. 2D*):

$$P_S = 1.130\ P_{NO}^{0.876} \qquad r^2 = 0.88 \qquad n = 61 \qquad (4)$$

allowing expected peakflows in the South Fork (P_S) to be estimated from observed or expected North Fork peakflows.

Between 1985 and 1989, only the new North Fork control tributaries are assured to provide stable records. Two methods provide estimates of expected North Fork peaks. First, the ARF gage (384 ha) is upstream of the XYZ confluence, so ARF peaks were unaffected by XYZ logging and can be calibrated against the average of peaks at control catchments HEN, IVE, and MUN (P_{HIM}, $m^3 km^{-2} s^{-1}$) for 1986 to 1989 (*fig. 2E*), providing a predictor for expected ARF flows (P_A) from 1985 to 1995:

$$P_A = 0.775\, P_{HIM}^{\ 0.943} \qquad r^2 = 0.91 \qquad n = 21 \qquad (5)$$

Second, because only 13 percent of the North Fork watershed was logged in 1985 and 1986, North Fork peakflows are likely to have changed little over this period. If the North Fork record is assumed to be stable for the period, calibration of weir peaks to the mean of peakflows at controls HEN and IVE (P_{HI}, $m^3 km^{-2} s^{-1}$; *fig. 2F*) provides a predictor for expected North Fork weir peakflows (P_N) for 1985 to 2004:

$$P_N = 1.06\, P_{HI}^{\ 0.854} \qquad r^2 = 0.93 \qquad n = 23 \qquad (6)$$

If North Fork peakflows had increased during the calibration period due to XYZ logging, use of equation 6 would provide lower estimates of change per percent forest removed than would an analysis based on equation 5. The validity of the assumption of stability can thus be tested.

Results and discussion

Application of equation 2 to the rainfall record provides estimates of expected late-summer North Fork flows for the selected dates in August and September from 1963 to 2007. Observed and expected flows agree relatively closely until 1985 (*fig. 3A*), when variance increases and several flows exceed expected values by >50 percent. Flows again increased after completion of most logging in 1991, and between 1996 and 2000 the mean of sampled flows is 1.7 times that expected. An unusual 50 mm/day storm in June 2001 produced a higher summer API than baseflows reflect, so expected flows are anomalously high in 2001. Flows begin to decrease after 2003, and by 2007 the mean again equals the expected value, but the slope of the recovery curve suggests that flows will continue to decrease. Values since 2001 reflect effects of both clearcutting and pre-commercial thinning. Overall, dry-season flow remained higher than expected for at least 15 years after North Fork clearcutting ended.

Low-flow deviations show a different pattern in the South Fork. Analyses based on equations 2 and 3 show similar results (*fig. 3B, C*): dry-season flow was enhanced for at least 4 years after South Fork logging began and reattained pre-logging values by 10 years. By 15 years after the onset of logging, flow had consistently dropped below expected levels, reaching a minimum at 21 years. Flows neared pre-treatment levels once again about 30 years after selective logging began. More data are needed to determine whether the apparent renewed downturn at 36 years represents the start of a new trend or simply reflects a short-term aberration.

Late summer flows began to show an effect during the first year of logging in the South Fork (*fig. 3D*), while North Fork responses to the major period of logging were not evident until the third year. The downstream portion of the South Fork was logged the first year, so hydrologic changes would have occurred first near the gage. North Fork logging progressed in the opposite direction, so riparian vegetation along the entire channel length could utilize increased flow before it reached the gage. At each site, the response became evident soon after areas near the gage were logged.

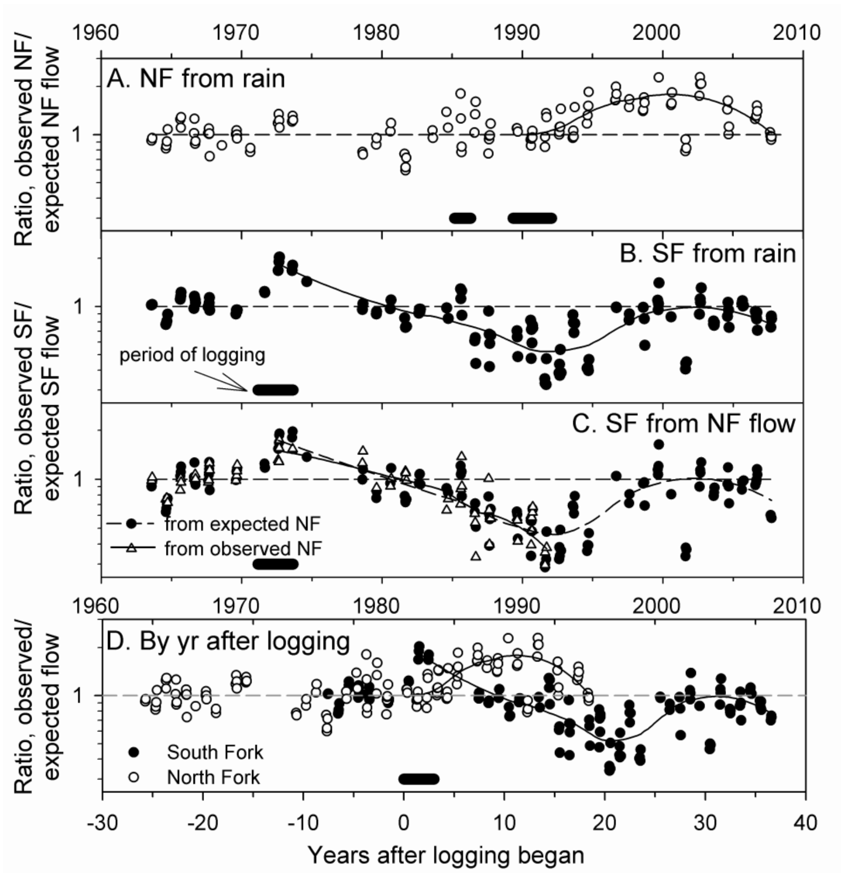

Figure 3—Ratio of observed to expected late summer flows at A. the North Fork weir; B. at the South Fork weir calculated from rainfall; C. at the South Fork weir calculated from North Fork flows; and D. at both weirs (calculated from rainfall) as a function of years after logging began. Post-logging curves are fitted using loess regression and exclude data from 2001.

Once the response is under way, the contrast in response trajectories is consistent with the differing silvicultural strategies used. South Fork selective logging left a third of the stand well distributed across the watershed. Because summers are dry, potential transpiration exceeds actual transpiration after May, and trees must compete for a limited water supply. When competition is reduced by selective logging, the remaining trees can quickly begin to use moisture that in the past would have sustained neighboring trees, and soil moisture reserves are again depleted by the end of the dry season. Such a response would be particularly rapid in stands containing coast redwoods because most second-growth redwoods originate as stump-sprouts, so neighboring trees often share root networks. In contrast, trees left uncut after clearcutting are not within reach of most the surplus moisture, and hydrologic recovery must rely on establishment of new vegetation on the clearcut surface. In addition, North Fork burned units were treated with herbicides in the mid-1990s, and most units were pre-commercially thinned in 2001, further prolonging the effects.

Reduced interception and transpiration appear to largely explain hydrologic changes after North Fork logging (Reid and Lewis 2007), with altered interception dominating wet-season changes. If summer rain is minimal, dry-season transpiration and residual moisture storage from the wet season are major influences on late-summer flow, so both transpiration and interception may be influential. Comparison of sampled late summer flows with those from 60 days earlier (with <20 mm of rain between the dates) shows that a given early summer flow leads to higher than expected late summer flow in the immediate post-logging period at both sites (*fig. 4*) and to lower than expected flow during the later period of flow depression after South Fork logging. A major causal mechanism thus is active during the summer, suggesting that altered summer transpiration is an important influence. The North Fork record is not yet long enough to test for depressed flows.

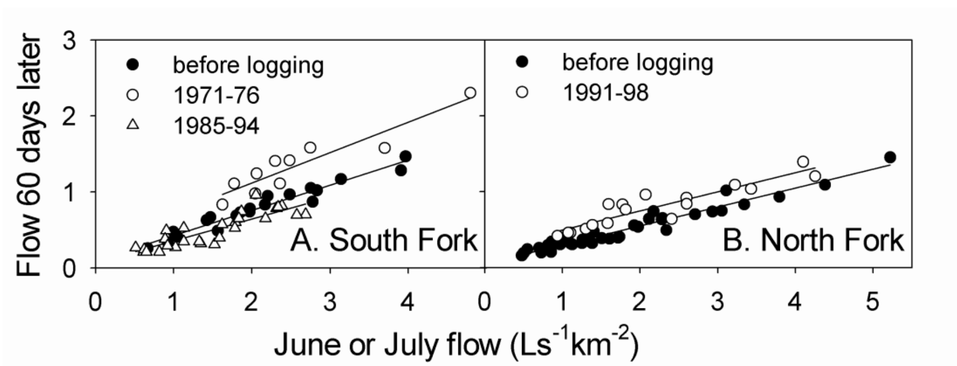

Figure 4—Comparison of early and late dry-season flows before and after logging at the A. South Fork and B. North Fork Caspar Creek weirs.

South Fork logging may have contributed to low-flow depression in several ways. First, selective logging promotes growth of young trees among the residuals. Water competition may thus be more intense than in the original stand, in which mortality during drought years likely achieved a long-term balance between stand density and water availability. Second, the lack of buffer strips increased light in riparian zones, increasing growth at sites where plants may disproportionately affect summer flow by removing water directly from the hyporheic zone. Third, regrowth along the South Fork included many red alders (*Alnus rubra*), which tend to use more water than conifers (Hicks et al. 1991). Fourth, a young, dense, rapidly growing age cohort may simply use more water than a mature stand (Moore et al. 2004). Low-flow depression has also been observed in other young stands in the Pacific Northwest (Perry 2007).

Peakflow deviations at the ARF gage after North Fork logging show increased peaks after 1990 (*fig. 5A*), a pattern also shown by deviations calculated using the assumption that weir peakflows did not change after XYZ logging (*fig. 5b*). For 1992 to 1994, the North Fork at ARF and at the weir both show a mean discharge-weighted peakflow increase of 0.62 percent per percent of forest logged in the catchment upstream, where percent logged at the weir includes only the portion logged between 1989 and 1992. This agreement suggests that North Fork peakflows were not strongly affected by XYZ logging. The weir record indicates that precommercial thinning in 2001 may have renewed the effect, and pre-treatment levels had not been reattained by 12 years after logging ended. The temporal pattern shown by *figure 5B* is similar to that identified for the 27-ha CAR sub-watershed (*fig. 5C*), which was 96 percent clearcut in 1991 (*fig. 1*).

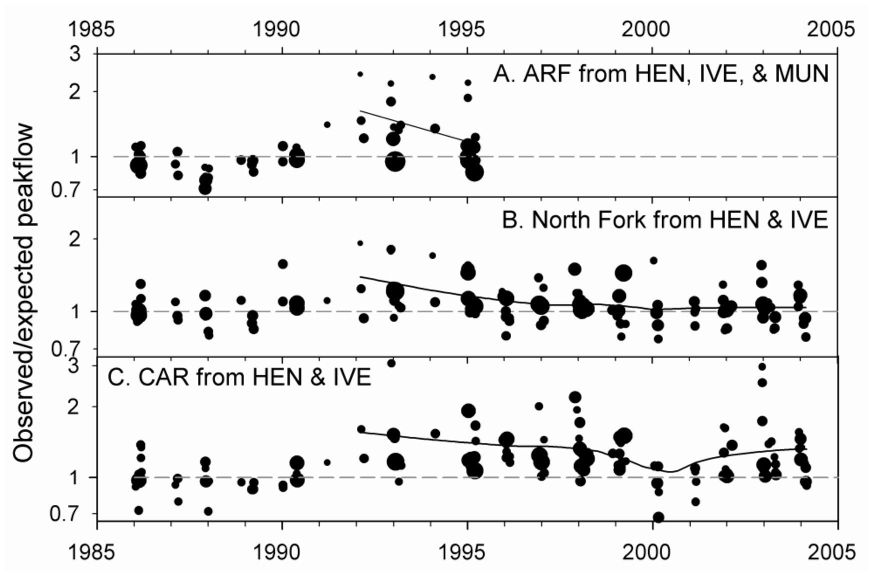

Figure 5—Ratios between observed and expected peakflows at A. ARF from average of peaks at HEN, IVE, and MUN; B. the North Fork weir from average at HEN and IVE; and C. clearcut sub-watershed CAR from average at HEN and IVE. Symbol areas are proportional to unit area discharges, and post-logging curves are fitted using loess regression for B and C, and linear regression for A.

Comparison of observed South Fork peakflows with those expected on the basis of correlations to observed or expected North Fork peaks shows that most South Fork peakflows were higher than expected 3 to 11 years after logging began (*fig. 6A*), and that they appear to remain slightly elevated until about 1995. If expected North Fork peaks are overestimated due to increased flow from XYZ logging during the calibration period, South Fork deviations after 1986 would be larger than shown. Overall, South Fork peaks showed a maximum 3-yr discharge-weighted mean increase of 0.36 percent per percent of forest logged in the catchment, about 60% of that seen in the North Fork. The South Fork response also appears to be delayed relative to that in the North Fork (*fig. 6B, C*).

Most peakflows at Caspar Creek are more strongly influenced by interception than transpiration (Reid and Lewis 2007), and recovery trajectories for interception and transpiration differ. Interception may be influenced both by a stand's canopy and by the volume of absorbent bark present. Leaf area can recover relatively quickly while bark storage cannot, so interception-controlled peaks may remain slightly elevated long after either clearcutting or selective logging ends, as appears to be the case at Caspar Creek.

Wet- and dry-season hydrologic responses appear to be decoupled in the South Fork, with peakflows remaining slightly elevated during the period of summer flow depression. Such decoupling is consistent with the presence of multiple mechanisms of hydrologic change, each with its own recovery trajectory and distinctive effects.

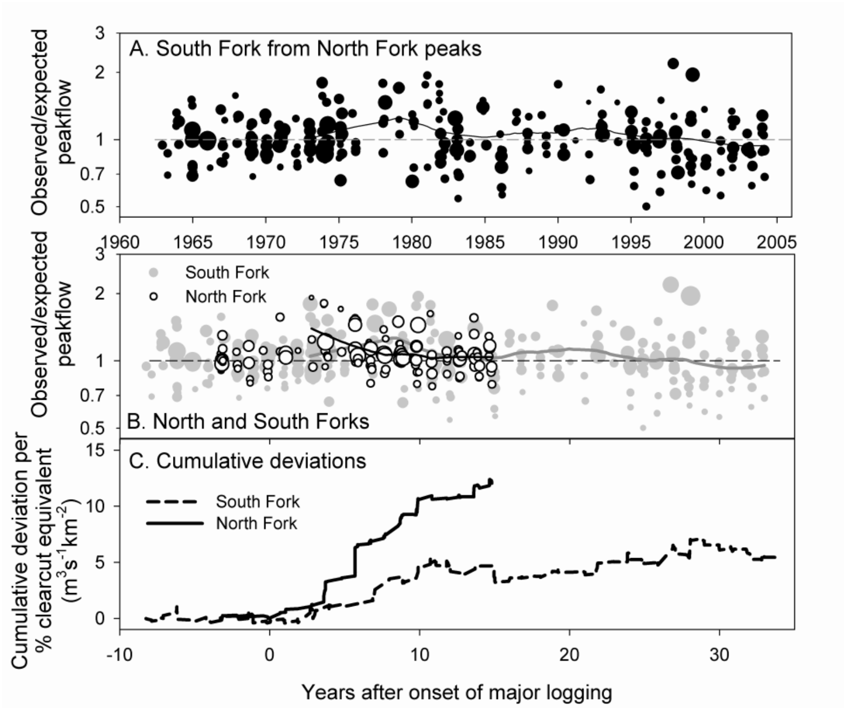

Figure 6—A. Deviations from expected peakflows at South Fork calculated from North Fork peakflows, B. comparison of recovery trajectories for North and South Fork peakflows by time after logging, and C. cumulative deviations. Symbol areas are proportional to discharges, and post-logging curves are fitted using loess regression.

Conclusions

Long-term flow records are rare for watersheds the size of the North and South Forks of Caspar Creek. For those that do exist, the questions the original monitoring studies were designed to address have often been superseded by new issues that had not yet been recognized when monitoring began. The Caspar Creek data stream, initiated a half-century ago, has provided the information needed to address two generations' controversies and concerns. Many of the watershed-related issues that have arisen since Caspar Creek monitoring began could not be resolved without long-term hydrologic data, and among these is the current need to understand mechanisms for long-term cumulative watershed effects associated with forest management.

Cumulative effects can accrue if changes resulting from a management activity are superimposed on changes induced by other contemporary or past activities. First steps in understanding the potential for cumulative hydrologic effects are to determine the length of time required for hydrologic recovery from a particular activity and to understand the factors that may affect the recovery rate. After selective logging of second-growth redwoods at Caspar Creek, it has taken at least 30 years for dry-season flows to reattain pre-treatment levels, and additional data are required before recovery can be assured to be complete. Long-term monitoring results also demonstrate that clearcut logging produced a very different recovery trajectory than selective logging, with the post-logging period of enhanced dry-season flow lasting about twice as long. Peakflow changes also show evidence of long-term persistence and differences between silvicultural treatments. Where hydrologic change is an issue of concern in a watershed, it may be useful to plan management to ensure maintenance of a diversity of stand ages and silvicultural strategies in order to desynchronize post-logging hydrologic changes.

Acknowledgments

This study is part of the cooperative Caspar Creek research program conducted since 1960 by the U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station and the California Department of Forestry and Fire Protection. John Griffen, Peter Cafferata, and an anonymous reviewer provided very helpful reviews.

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Landslides After Clearcut Logging in a Coast Redwood Forest[1]

Leslie M. Reid[2] and Elizabeth T. Keppeler[2]

Abstract

Landslides have been mapped at least annually in the 473 ha North Fork Caspar Creek watershed since 1985, allowing evaluation of landslide distribution, characteristics, and rates associated with second-entry partial clearcut logging of 1989 to 1992. Comparison of sliding rates in logged and forested areas shows no appreciable difference for streamside slides (size range: 7.6 to 380 m^3). However, the incidence of large landslides, including both streamside and upslope slides of 98 to 4900 m^3, varied by treatment. Such slides displaced 12 m^3/yr per km^2 of unlogged forest but showed rates one and two orders of magnitude higher in logged areas and along roads, respectively. Moreover, the volume rate of sliding from roads in logged areas was more than three times that from forested roads. The largest slides occurred 9 to 14 years after logging, when root cohesion is expected to be near its minimum value; and within a few years of pre-commercial thinning, when hydrologic changes are again evident. Large slides may strongly influence suspended sediment yields both by increasing yields for several years after the slide and by emplacing temporarily stable channel and floodplain deposits, which then provide a sediment source for future gully and bank erosion.

Key words: cumulative effects, erosion, landslides, logging, roads, sediment budget

Introduction and site description

Associations between logging and increased landslide incidence have long been recognized, and many studies attribute the effect largely to reduced root cohesion (for example, Wu et al. 1979). However, because second-growth coast redwoods (*Sequoia sempervirens*) usually survive logging, altered root cohesion is rarely considered important after redwood logging. Recent studies have identified another influence that may be more relevant in redwood forests: reduced rainfall interception during major storms after logging could increase pore pressures on marginally stable slopes, contributing to their destabilization (Keim and Skaugset 2003, Reid and Lewis 2009). We here evaluate landslide distribution in the 473 ha North Fork Caspar Creek Experimental Watershed (N39°21´ W123°44´), near Fort Bragg, California, to 1) quantify the influence of roads and logging on landsliding in the area; 2) assess interactions between the effects of roads and logging on landslide generation; 3) evaluate storm characteristics associated with landsliding; and 4) assess the contribution of landsliding to the watershed's suspended sediment yield. This study uses a longer record to build on analyses by Cafferata and Spittler (1998).

The North Fork Caspar Creek watershed is underlain by marine sandstones and siltstones of the Coastal Belt of the Franciscan Complex. Soils on ridge-top marine

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[1] This paper was presented at the redwood science symposium: coast redwood forests in a changing California. 21-23 June 2011, Santa Cruz, California.

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terrace deposits are deep, sandy loams, while those lower on the slopes are shallower with higher clay and gravel contents. Many 3rd and 4th order channel segments flow through inner gorges, and drainage density is 4 to 5 km per km^2. Annual precipitation averages 1170 mm and falls mostly as rain; about half runs off as streamflow. The watershed's old-growth redwood forest was logged between 1860 and 1904. Second-growth stands had matured by 1960, when the Caspar Creek watersheds were selected as a location for watershed-scale experiments.

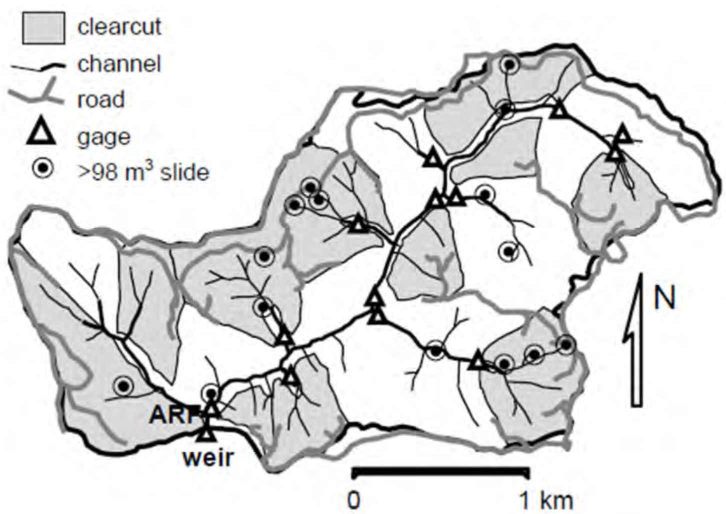

Figure 1—North Fork Caspar Creek watershed.

Stream gaging began in 1962 at the newly constructed North Fork weir (*fig. 1*), and the North Fork watershed remained forested as a control for the next 23 years. Thirteen new gaging stations were installed in 1984 for a study of scale-related hydrologic effects of clearcutting in a second-growth redwood forest. The ungaged XYZ tributary, which enters the head of the weir pond at the basin mouth, was not included in the experiment and was about 64 percent clearcut in 1985 and 1986. Catchments above five gaging stations were then 90 to 100 percent clearcut in 1989 to 1992, while those above three others were left forested; five downstream gages provided information for partially clearcut watersheds of increasing size. Overall, 13 percent of the watershed was logged in 1985 and 1986 and another 37 percent between 1989 and 1992. Logged units were mostly cable-yarded from ridge-top landings. About 18 km of road are present and are located primarily on or near ridge-tops, with 45 percent of the road length constructed immediately prior to logging at each unit. Selectively logged buffer strips were left within 40 to 60 m of channels draining more than about 10 ha.

Methods

Landslide data available for the watershed were grouped into two partially overlapping data sets. First, "streamside slides" consist of landslide events larger than 7.6 m^3 that were observed within 15 m of a gaged tributary channel. These features have been mapped annually since 1985 along 72 percent of the length of 2nd- and higher-order channels in the watershed; observations are also made after major storms. Slide dimensions, associations with treethrow and roads, and the volume of debris still present are noted for each feature. Mapped events in some cases represent progressive enlargements of previously mapped landslides, so the number of landsliding “events” is larger than the number of independent landslides mapped. The second data set consists of “large slides”: those observed anywhere in the North Fork watershed that are 98 m^3 or larger. Many of these were initially observed during the channel-based surveys, but most are located >15 m from a mapped channel. For both data sets, we estimate that volumes are accurate to better than ±25 percent.

Management effects were evaluated separately for each data set using post-1991 data; all land-use categories thus had experienced the same storms. Because buffer zones were selectively logged, they are either evaluated separately or grouped with logged slopes, depending on the analysis. For “streamside slide” data, the inventoried area includes the 15 m strip on either side of each gaged channel, while the entire watershed area above the weir is represented by the “large slide” data set.

A slide is assumed to be associated with a road if it is adjacent to the road or is immediately downslope of a drainage outlet. Areal sliding rates from roads are calculated per unit area of road surface, where road areas are estimated by applying an average width for each road type (mainline or spur road) to the length of that road type present. Only half the road width along North Fork boundary ridges is included because half the drainage from such roads is expected to enter other watersheds. To compare sliding rates on roads in different vegetation types, we calculated rates of volume displacement per unit length of road, and the total “road-km-yr” represented by a road segment in a particular land class is calculated by multiplying the segment’s length by the time it spent in that class. Roads traversing level terrain (gradient <10 percent) are disregarded for this portion of the analysis. These low-gradient surfaces were delineated using aerial photographs and generally correspond to the ridge-top marine terraces. Spur roads on steep ridges often abut a logged slope on one side and a forested slope on the other; in such cases, half the segment length is assigned to each class.

Cafferata and Spittler (1998) identified threshold rainfalls associated with large North Fork slides between 1962 and 1998, and we reevaluate thresholds to account for rainfall interception and transpiration. Rainfall has been monitored since 1962 at the SFC620 gage, located about 3 km downstream of the North Fork weir. Storms are identified as consecutive rain days bounded by days with <2 mm of rain and having a total rainfall >25 mm. The maximum 24-hr rainfall is identified for each storm using consecutive 1 hr rainfalls. The 2nd-growth forest at Caspar Creek intercepts and reevaporates 21 percent of the above-canopy rainfall (Reid and Lewis 2009), so subcanopy rainfall is calculated as 79 percent of the gage rainfall, while that in clearcuts is represented by the gage rainfall. A long-term antecedent precipitation index (API) is assessed for the day preceding each storm by applying a recession coefficient of 0.97 to calendar-day rainfalls. APIs for forested conditions are calculated using subcanopy rainfall, and estimated daily transpiration (Reid and Lewis 2009) is also subtracted. APIs of less than 0 are assigned a value of 0.

Because gaging stations are visited at least daily during storms, failure timing is often known to within days. Most other slides can be attributed to particular sequences of storms occurring over a 2- to 3-week period, and we assume that such landslides occurred during the most severe storm of the sequence.

Suspended sediment loads have been monitored at the North Fork weir since 1962 and at the sub-watershed gages during water years 1986 through 1995;

measurements continue at six of these gages. Loads were originally estimated using sediment rating curves constructed from gravimetrically determined sediment concentrations. Since 1995, load estimates are refined using turbidity readings logged at 10 minute intervals. The potential relative importance of landsliding as a sediment source is evaluated by comparing rates of sediment displacement by landslides with annual suspended sediment loads. An estimated 50 to 75 percent of the sediment displaced by landslides is expected to be of small enough grain size to be carried in suspension, and an additional component would break down to suspendible sizes during transport.

Results and discussion

Between 1985 and 2006, 91 events larger than 7.6 m^3 were observed in the North Fork watershed, and 83 of these occurred after 1991. Of the 65 post-1991 events that were mapped within 15 m of gaged channels (*table 1*), 14 were reactivations of previously mapped slides, so 51 independent slides of 7.6 to 380 m^3 are represented in the streamside slide data set, displacing 2280 m^3 of sediment. "Legacy" slides that originated at an old splash dam or along skid roads built during old-growth logging of the 1800s account for 29 percent of the streamside slide volume.

Table 1—*Landslides in the North Fork Caspar Creek watershed, 1992-2006.*

	Legacy	Forest	Buffer	Logged	Road	Total
Streamside slides > 7.6 m^3						
bank km-yr		140	70.2	34.6		244.8
number of events	7	30	19	9	0	65
volume m^3	670	880	520	210	0	2280
m^3/km-yr		6.3	7.4	6.1	0	9.3
All slides ≥ 98 m^3						
km^2-yr present		34.1	2.9	32.3	1.6	70.9
number of slides	2	3	1	3	4	13
volume m^3	600	420	140	3620	7170	11950
m^3/km^2-yr		12	48	112	4480	169

Between 1992 and 2006, differences in streamside landsliding rates were inconsequential among treatments, with forested, buffered, and clearcut reaches producing averages of 6.3, 7.4 and 6.1 m^3/yr per km of channel bank, respectively (*table 1*). These results may be complicated by influences of logging on adjacent and downstream sites. Two forested slides were associated with blowdown along clearcut margins, for example, and failures along one forested reach began when increased runoff from upstream logging appears to have accelerated headcut retreat at the site, undermining adjacent banks. Events that may have been influenced by nearby logging account for 21 percent of the post-1991 sediment displacement from forested streamside slides. On average, 55 percent of the displaced sediment remained within the landslide scars.

Although rates of sediment displacement from streamside slides are similar among treatment types, associations between streamside events and potential destabilizing features show several differences. In forested areas, 86 percent of the streamside slides are associated with treefalls or undercut banks, while only 70 percent of buffer strip slides and 33 percent of those in clearcuts show such associations. All streamside events at logged sites show evidence of hydrologic influence, either through their association with tunnel erosion or bank erosion, or, in one case, by mobilization of landslide debris into a short debris flow.

Fifteen slides larger than 98 m^3 (including seven also tabulated as streamside slides and three progressive failures) displaced about 13,500 m^3 of sediment in the watershed between 1985 and 2006 (*fig. 1*); 13 of these occurred after 1991 (*table 1*). Debris flows triggered during two events accounted for 2500 m^3 through channel scour and undercutting of banks. In contrast to the smaller streamside slides, the post-1991 rate of sediment displacement from large slides in forested areas (12 m^3/yr per km^2 of unlogged forest) was appreciably less than in treated areas (48 and 112 $m^3km^{-2}yr^{-1}$ in buffer strips and on clearcut slopes, respectively). The four slides associated with roads after 1991 produced about 4500 $m^3yr^{-1}km^{-2}$ of road surface. For large slides, an average of 21 percent of the displaced volume remained on the scars.

Two large slides occurred along forested roads, contributing 16 $m^3km^{-1}yr^{-1}$ for the 97 road-km-yr of forest-lined road present between 1985 and 2006. Both slides occurred before 1992. The four large slides along road segments within logged areas all occurred after 1994 and together displaced an average of 53 $m^3km^{-1}yr^{-1}$ for the 135 road-km-yr of logged road present between 1985 and 2006. The period during which all road types were exposed to the same storms, 1992 through 2006, showed no displacement for the 60 road-km-yr of forested roads present and 61 $m^3km^{-1}yr^{-1}$ for the 118 road-km-yr of logged roads.

Logging roads in steep terrain have long been recognized to be relatively unstable features (for example, Swanson and Dyrness 1975), and Bawcom (2007) notes the prevalence of road-related slides in and near the Caspar Creek watershed. The distribution of slides found by the current study to be associated with North Fork Caspar Creek roads suggests that influences of roading and logging may interact. Piezometric data from North Fork hillslopes (Keppeler and Brown 1998) indicate that such interactions could result from hydrologic change. Measurements in a swale upslope and downslope of a road corridor showed little change during the winter after road construction, though downslope readings showed brief pore pressure spikes that had not been observed previously during similar storms. Piezometers above the road remained dry both before and after roading. However, after the swale was clearcut the following year, positive pressure heads developed upslope of the road and increased as the wet season progressed. The road prism may thus have been retarding subsurface drainage, but changes became apparent only after reduced transpiration and interception increased water inputs to subsurface storage.

The temporal distribution of landslides in the North Fork watershed reflects a strong hydrologic influence. Each of the four storms that generated a total slide volume greater than 100 m^3 in forested areas had a sub-canopy API of >180 mm, a sub-canopy storm rainfall >115 mm, and a maximum 24 hr sub-canopy rainfall >75 mm (*table 2*). These were four of only five storms between 1985 and 2006 that shared these characteristics, and the fifth (December 2005) generated the largest landslide observed in the North Fork watershed between 1985 and 2010.

If these subcanopy rainfall thresholds are applied to the gage record, thus removing the influence of rainfall interception, five additional storms surpass the thresholds, and three of these generated significant landsliding in logged, roaded, or buffered areas. A fourth predated most logging, and the fifth closely followed another slide-generating storm. Three other notable slide-generating storms occurred between

1985 and 2006 (*table 2*), including a major windstorm (December 1995), in which all slides were associated with treefalls, and two storms that strongly exceeded two of the three thresholds. On average, a storm surpassed all thresholds for notable sliding in forests once in 4 years and twice as often for notable sliding on treated slopes.

Table 2—*Storms generating numerous or particularly large landslides, 1985 to 2006.*

Storm	Number of slides	Slide volume (m^3) forest	Slide volume (m^3) treated	Rainfall (mm) storm	Rainfall (mm) 24-hr	Effective API (mm) forest	Effective API (mm) treated
Feb 1986	2	8	1262	310	64	149	212
Jan 1993	8	112	79	146	127	255	345
Jan 1995	9	62	1932	387	107	88	130
Mar 1995	8	67	667	267	97	120	180
Dec 1995	4	55	55	148	69	46	70
Dec 1996	7	154	28	302	104	181	249
Jan 1998	6	37	191	338	85	157	221
Mar 1998	1	103	0	189	124	242	338
Mar 1999	3	0	119	122	95	194	277
Dec 2002	5	102	2035	155	100	230	307
Dec 2005	2	0	4009	266	122	218	291

Landslide risk is likely to be affected by three prominent trends after logging. First, transpiration and interception rates increase once again as regrowth progresses, reducing the likelihood of attaining excessive pore pressures. Second, even though most redwood stumps quickly resprout, a proportion of the original roots die and begin to decay, eventually to be replaced by new roots sustained by the recovering foliage. The minimum root cohesion is expected about 10 to 15 years after logging (J. Lewis, personal communication, March 2011). Third, exposure of trees along clearcut margins initially increases their risk of blowdown, but early post-logging wind storms cull the least stable trees, and regrowth eventually reduces the margins' exposure. Slides associated with blowdown along clearcut margins might thus show reduced frequency with time since logging.

The observed timing of failure events after logging may reflect such trends. All non-road-related slides at either forested or buffer strip sites after 1985 were smaller than 200 m^3. Plots of cumulative landslide volume against years since logging for slides >200 m^3 show slightly different patterns for clearcuts and buffer strips: both show high rates until about 8 years after logging, but slide activity resumes on clearcut slopes at about 13 years, while it does not do so in buffer strips (*fig. 2*). In contrast, all slides larger than 200 m^3 at logged sites occurred 9 to 14 years after logging, when root cohesion on the logged slopes is near its minimum value but regrowing vegetation would ordinarily have ameliorated much of the initial hydrologic change. However, about 70 to 90 percent of the crown area was removed at these sites by pre-commercial thinning 9 to 11 years after logging, and peakflows had again increased to well above those expected for unlogged conditions (Lewis and Keppeler 2007). The largest slides thus occurred at a time when influences of both altered root cohesion and hydrologic change were strong.

The association of landsliding with storms having high 24 hr and storm rainfalls and high APIs demonstrates the dependence of sliding on hydrologic triggers, as does the increased slide frequency for smaller storms after logging. However, the delay in occurrence of the largest slides suggests that while hydrologic influences affect the frequency of sliding, reduced root cohesion may provide an important influence on the size of the slides triggered.

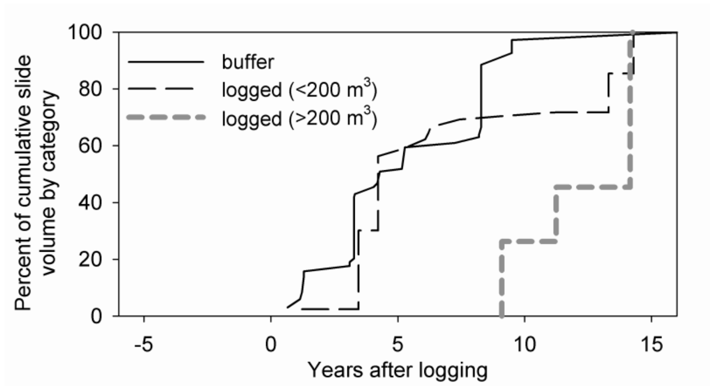

Figure 2—Cumulative landslide volume on treated sites as a function of time after logging.

Annual suspended sediment yield is not significantly correlated with the volume of landslide sediment displaced along gaged channels between 1986 and 1995, the period during which all gages were operating (Reid et al. 2010). All slides that occurred in these sub-watersheds during the period analyzed were < 280 m^3. Although the average sediment displacement rate for streamside slides of 6 to 7 m^3/km of channel bank for 2nd-order and larger channels (*table 1*) would amount to about 20 $m^3km^2yr^{-1}$, more than half of the sediment remains in storage on or near the slide scars and does not consistently contribute to the suspended sediment load.

The largest slides, in contrast, can strongly influence short-term suspended sediment yields. The "Z" slide occurred in 1995 in the ungaged XYZ tributary on a slope logged in 1986 and thinned in 1994. Debris from the 1750 m^3 failure flowed 200 m downstream, scouring sediment from channel banks and triggering streamside failures to mobilize an additional 1000 m^3. Because only XYZ tributary enters the North Fork between the ARF gage and the North Fork weir, excess XYZ inputs can be estimated by comparing measured suspended sediment loads at the weir with those predicted using pre-slide correlations between loads at ARF and at the weir. Results show that the weir station recorded 1300 kg/ha (610 t) more than expected during 1995 to 2000 (*fig. 3*), and that loads had once again stabilized by 2000. The excess suspended sediment accounted for about 14 percent of the sediment displaced by the slide and debris flow, and 55 percent of the slide debris was redeposited near the slide scar, so about 30 percent of the displaced sediment remained within the channel and weir pond. The annual pre-treatment suspended sediment load averaged 68 ± 39 t $km^{-2}yr^{-1}$ at the weir between 1963 and 1985 (incorporating a bias correction suggested by J. Lewis, personal communication, 2007, to adjust for the pre-1976 sediment sampling protocol), so the slide produced the equivalent of about 2 years of suspended sediment yield for the 4.7 km^2 watershed above the weir.

During the 23 years that North Fork sediment loads had been monitored before logging began, only a 3300 m^3 landslide was large enough to increase sediment markedly at the North Fork gage. The 1974 failure occurred on a forested inner-gorge slope about 2 km upstream of the weir (Rice and others 1979). Considering now only the extremely large slides (>2500 m^3) expected to be recognizable from suspended sediment records at the weir, those in the 40 km^2yr of logged area present between

1963 and 2006 displaced an average of 190 $m^3km^{-2}yr^{-1}$, while that in the 173 km^2yr of forest displaced 19 $m^3km^{-2}yr^{-1}$. About 25 to 50 percent of the displaced material is expected to be too coarse to be carried in suspension.

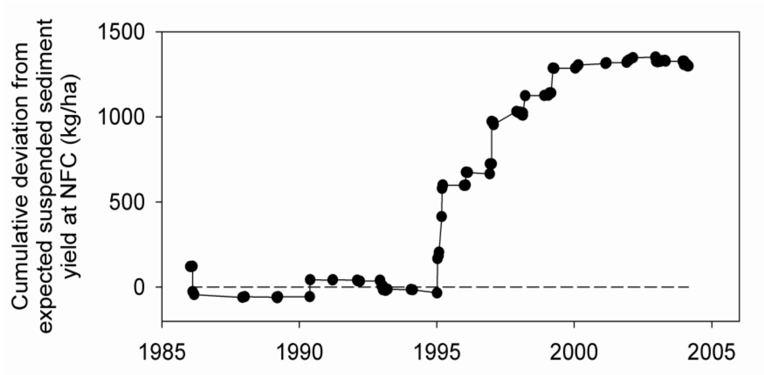

Figure 3—Cumulative deviation in storm suspended sediment yields from expected values at the North Fork weir. The Z slide occurred in January 1995.

The mean pre-logging suspended sediment load of 68 t $km^{-2}yr^{-1}$ at the North Fork weir is equivalent to a volume rate of 52 $m^3km^{-2}yr^{-1}$ if a bulk density of 1.3 g cm^{-3} is assumed. The single extreme slide on undisturbed land thus could have increased the long-term mean suspended yield from forest lands by as much as 20 percent if 75 percent of the displaced sediment eventually becomes suspended load. If the same assumptions are made for extreme landslides associated with disturbed sites, these slides could increase the long-term suspended sediment yield for logged areas by a factor of about 3.1 relative to that expected for pre-treatment conditions. In either case, much of the displaced sediment would be stored along downstream channels and become susceptible to later remobilization by bank erosion and gullying.

Conclusions

Clearcut logging appears to have influenced landsliding in the North Fork watershed primarily through the increased incidence of large landslides and by destabilization of logged slopes adjacent to roads. Clearcutting also appears to have reduced the storm magnitudes required to generate landsliding.

Logging can affect landslide incidence by decreasing transpiration and rainfall interception and by reducing root strength. Hydrologic conditions appear to be the primary influence on the generation of small to medium landslides at Caspar Creek, while root strength may be an important influence on landslide size. Notably, all slides >200 m^3 that occurred between 1986 and 2006 were associated either with roads, buffer strips, or clearcut slopes, and all slides >200 m^3 on logged lands occurred 9 to 14 years after clearcutting and within 5 years of pre-commercial thinning. Three destabilizing conditions were present at that time: 1) root cohesion would have been near its minimum value after logging as roots killed by clearcutting had decayed while regrowing roots were not fully developed, 2) new roots killed by thinning would have begun to decay, further reducing root cohesion, and 3) thinning would have again reduced interception and transpiration, again increasing the hydrologic response to storms. At sites where slope stability is of particular concern, it may be useful to delay thinning to a time when root cohesion has recovered more fully from logging so that the ensuing increase in hydrologic response does not coincide with the period of minimum root cohesion.

An interaction between the topographically destabilizing presence of roads and hydrologic changes induced by clearcutting may be responsible for the higher frequency of road-related landsliding in clearcuts than in forest. In many areas, rates of landsliding along roads have been contrasted to rates on logged slopes, producing the conclusion that logging has a relatively insignificant influence on sliding. However, in areas where road-related instability is expressed primarily after adjacent slopes are logged, the distinction between “logging-related” and “road-related” landslides may not be particularly useful: the cumulative effect of roads and logging may be synergistic rather than additive. The number of slides associated with roads was relatively small in the North Fork Caspar Creek watershed, however, and it would be useful to test for interactions of roads and logging at additional sites.

Comparison of landslide incidence with suspended sediment yields in North Fork Caspar Creek tributaries demonstrates that small to medium landslides have little short-term effect on suspended sediment yields (Reid et al. 2010), while a few extremely large slides can strongly influence the suspended sediment yield for the entire watershed. Much of the sediment generated by landsliding remains stored in and adjacent to channels, making it susceptible to later entrainment through bank erosion and gullying. Sediment inputs during small to moderate storms might thus be influenced by the history of past landsliding.

Acknowledgments

This study is part of the cooperative Caspar Creek research program established in 1960 by the U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station and the California Department of Forestry and Fire Protection. We thank Lynn Webb, Peter Cafferata, and two anonymous reviewers for their comments and suggestions.

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The Impact of Timber Harvest Using an Individual Tree Selection Silvicultural System on the Hydrology and Sediment Yield in a Coastal California Watershed

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Abstract

There is still widespread concern regarding the environmental impact of timber harvest. This is certainly true for timber harvest activities that occur on the Swanton Pacific Ranch, the school forest for the California Polytechnic State University, located in Santa Cruz County, California. A paired watershed study was carried out to help determine the impact of contemporary forest practices on the hydrology and sediment yield of the Little Creek watershed in the Swanton Ranch. Discharge and suspended sediment were measured at two gauging stations, the North Fork and South Fork of Little Creek, from 2002 to 2009. The calibration period extended from 2002 to 2008. Timber harvest occurred in the summer of 2008 and three storms occurred during the winter after timber harvest. During the summer of 2009, the Lockheed Fire occurred and significant portions of both watersheds were burned, thus the South Fork lost its value as a control watershed. Hourly peak flows, total storm quickflow volume, and total storm sediment yield were analyzed for 35 storms during the calibration period and for the three storms that occurred during the post-harvest winter. There was no evidence that the timber harvest had any impact on the hydrology or sediment yield of the Little Creek watershed.

Key words: hydrology, sediment yield, individual tree selection

Introduction

The impact of timber harvest on the hydrology and sediment yield of forested watersheds is a topic of ongoing concern and study. Research results from seminal paired watershed studies in the Pacific Northwest (PNW) show that timber harvest activities do affect annual water yield (Harris 1973, Rothacher 1970, Wright et al. 1990), peak flows (Harr et al. 1975, Jones and Grant 1996, Rothacher 1973, Ziemer 1981), and sediment yield (Brown and Krygier 1971, Fredriksen 1970, Lewis 1998). These results, for the most part, chronicle the environmental impact associated with a bygone era. In that era the trees to be harvested were very large, roads were constructed commiserate with the harvest, the logging machinery was large and had a significant environmental footprint, and finally, there were no Best Management Practices (BMPs) applied. In the ensuing 3 to 4 decades forest practices have changed drastically. Contemporary managed forests are made up of smaller, more uniform, harvest-regenerated stands, the road systems exist, the logging machinery is smaller and has a smaller environmental footprint, most importantly, timber harvest

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activities are highly regulated and BMPs are used. This is certainly the case for the Swanton Ranch in Santa Cruz County, California (Piirto et al. 1999). Nonetheless, there is still ongoing concern regarding the environmental impact of timber harvest activities. The objective of this project was to investigate the impact of timber harvest activities in harvest-regenerated redwood stands with an individual tree selection silvicultural system on the hydrology and sediment yield of the Little Creek watershed.

Methods

Study area

This paired watershed study was carried out in the Little Creek watershed located on the Swanton Pacific Ranch, which is the school forest for the California Polytechnic State University in San Luis Obispo. Swanton Ranch is located 18 km northeast of Santa Cruz near Davenport, California. The two watersheds that made up this study are the South Fork Little Creek (SF), which has an area of 106 ha and was the control watershed, and the North Fork Little Creek (NF), which has an area of 106 ha, was the control watershed (*fig. 1*).

The elevation of the study watersheds ranges from 100 to 580 m and average annual precipitation ranges from approximately 450 to 1400 mm from the outlet to the higher elevations (Gaedeke 2006). For the period of study, 2002 to 2010, the average annual precipitation was 875 mm at the outlet of the study watersheds to 1060 mm on the ridgelines. The soils in the study watershed are deep to moderately deep, well drained to somewhat excessively drained, and have a surface layer of loam, sandy loam, or stony sandy loam (Bowman and Estrado 1976). The forest on the Swanton Ranch in lower Little Creek is a harvest-regenerated stand of redwood (*Sequoia sempervirens*) and Douglas-fir (*Psuedotsuga menziesii*). The stand is, nominally, 100 years old and before harvest has 280 ft2 of basal area, 45 mbf/ac, and an incremental growth of 500 bf/ac/yr. The stands are comprised of 60 percent redwood, 25 percent Douglas fir, and 15 percent tanoak (*Lithocarpus densiflora*) (Piirto et al. 1999).

Timber harvest treatment

During the summer of 2008, timber harvest took place on 116 ac (47 ha) of the Swanton Ranch property above the gauging station on the NF Little Creek. The silvicultural system used was an individual tree selection system in compliance with the Santa Cruz County Rules of the California Forest Practice Regulations. The timber harvest removed 728 trees that accounted for 3,882 ft2 of basal area and 824 mbf (gross scale) of volume. Timber harvest removed, on average, six trees per ac that accounted for 33 ft2 of basal area and 7,103 bf per ac. Those values represent 23 percent of the basal area and volume in unmanaged redwood stand in lower Little Creek. The harvest was carried out with ground based systems, tractors, and aerial systems, slackline skyline. Approximately 62 percent of the volume was harvested with tractor systems and 38 percent was harvested with cable systems. Harvest operations took place, for the most part, off of existing roads.

Data collection

Precipitation was measured with four tipping bucket rain gauges located in the Little Creek study watersheds (*fig. 1*). Streamflow, turbidity, and suspended sediment concentration (SSC) were measured at two gauging stations located at the outlets of the NF and SF Little Creek for the study period (*fig. 1*). Pressure transducers were used to measure stage that was stored on data loggers and converted to discharge with empirical stage vs discharge relationships. Turbidity was measured with real-time, instream turbidity probes. Automatic pump samplers were used to collect water samples during storms that were later analyzed for suspended sediment concentration. Streamflow was measured year round but values of turbidity and water samples for SSC were taken hourly just during storms.

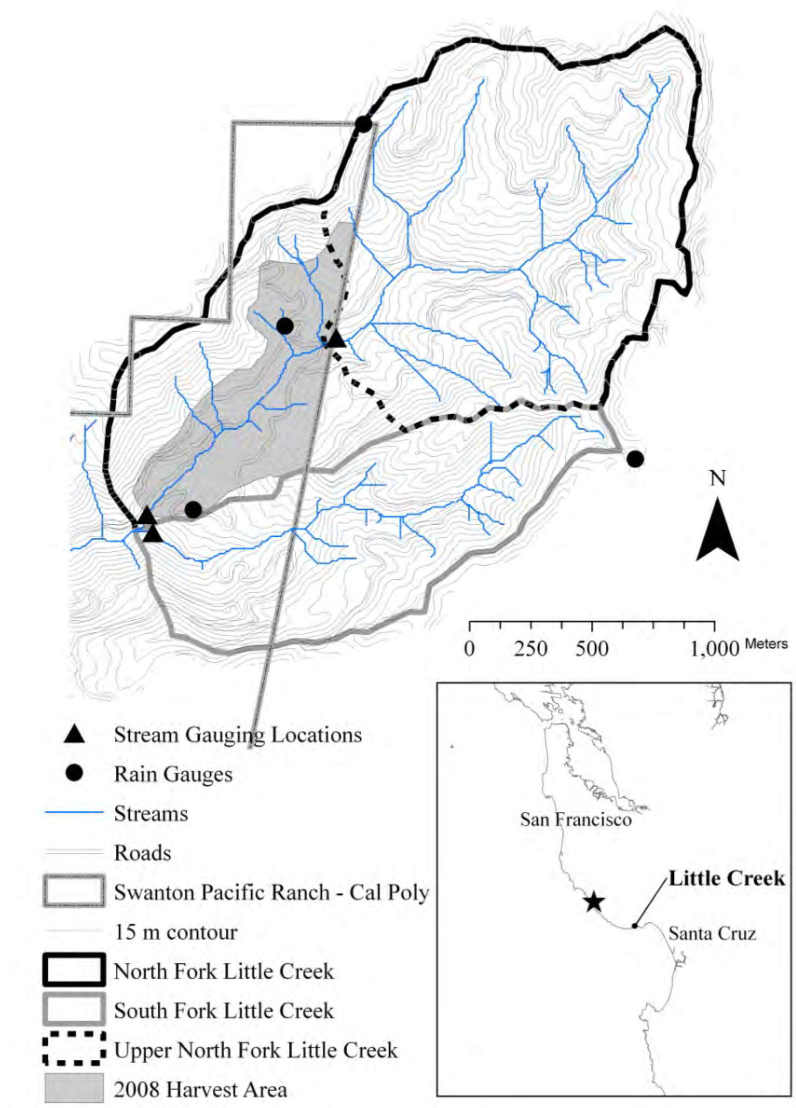


Figure 1—Little Creek study watersheds, California.

Data analysis

Three parameters were selected to use for analysis of the streamflow and sediment yield data; peak flow, total storm quickflow volume, total storm sediment yield. Gaedeke (2006) described how storms were selected from the time series streamflow data. Subsequent analysis of the streamflow data used his approach. Once a storm was defined, the peak flow is the maximum hourly flow in the storm hydrograph and is reported in l/s/ha. A standard hydrograph separation (Hewlett and Hibbert 1967) was performed to determine the total storm quickflow volume and this value is reported in mm. Finally, turbidity vs SSC relationships were developed so SSC values could be estimated. SSC in mg/l were multiplied by hourly discharge in l/s to give an hourly sediment discharge, which were summed to give the mass of sediment for the storm, expressed in kg. Unique relationships between turbidity and SSC were storm and watershed specific. Examples of these relationships and how they were developed is presented in Gaedeke (2006).

For the calibration time period, 2002 to 2008, pairs of values from the control (SF) and the treatment (NF) watershed for peak flow, total storm quickflow volume, and total storm sediment yield were plotted. These data were not homoscedastic, thus a log transform was performed on the variable values to satisfy that assumption. A linear regression was performed on the transformed data and plotted with the data. Then values of the 95 percent prediction limit were calculated and added to the graphs. Log transform values for peak flow, total storm quickflow volume, and total storm sediment yield for the three storms that occurred during the post-harvest water year were added to the plots. The location of the post-treatment data points relative to the 95 percent confidence limits were used to evaluate if a statistically significant, post treatment effect occurred the first winter after timber harvest.

During the summer of 2009 the Lockheed Fire occurred. Significant portions of the NF and SF of Little Creek were involved in that fire. The SF Little Creek no longer had value as a control watershed for the study. Thus, no further post-harvest data are available.

Results

The data set that was analyzed for this study consisted of 38 storms that occurred between December 2002 and March 2009. Of these 38 storms, 35 occurred during the calibration period between December 2002 and February 2008. Three storms occurred during WY 2009 after timber harvest was carried out.

For the calibration period for the SF Little Creek, the control watershed, the average peak flow was 1.08 l/s/ha and the values ranged from 3.46 to 0.13 l/s/ha. The three peak flows for the post logging period were 0.39, 1.34, and 1.02 l/s/ha for an average of 0.92 l/s/ha. For the calibration period for the NF Little Creek, the treatment watershed, the average peak flow was 1.28 l/s/ha and the values ranged from 4.79 to 0.14 l/s/ha. The three peak flows for the post logging period were 0.49, 1.53, and 0.86 l/s/ha for an average of 0.96 l/s/ha. All three of the peak flows that occurred after logging during WY 2009 were within the variability exhibited by the population of calibration peak flows (*fig. 2a*). There was no observed impact of the timber harvest on peak flows.

For stormflow volumes the control watershed, the SF Little Creek, for the calibration period had an average of 16.2 mm and the values ranged from 67.5 to 0.2 mm. The three storms for the post logging period had quickflow volumes of 0.6, 11.8, and 9.9 mm for an average of 7.4 mm. For the treatment watershed, the NF Little Creek, for the calibration period storm quickflow averaged 11.1 mm and the values ranged from 51.4 to 0.4 mm. The three storms in the post logging period had quickflow volumes of 0.2, 14.5, and 10.5 mm for an average of 8.3 mm. Once again, the values of storm quickflow for the three post logging storms fell within the variability exhibited by the calibration data (*fig. 2b*). There was no observed impact of timber harvest on storm quickflow volumes.

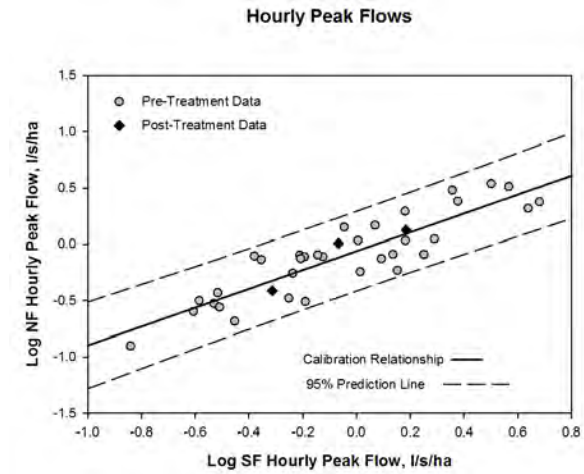
For total storm sediment yield the SF Little Creek, the control watershed, for the calibration period had and average of 31.8 kg/ha and a range from 396 to 0.1 kg/ha. The three storms for the post logging period had total storm sediment yields of 4.9, 64.9 and 9.5 kg/ha for an average of 26.4 kg/ha. For the treatment watershed, the NF Little Creek, for the calibration period the total storm sediment yield averaged 35.9 kg/ha and ranged from 331 to 0.1 kg/ha. The three storms during the post logging period had total storm sediment yields of 0.3, 5.2, and 1.5 kg/ha for an average of 2.3 kg/ha. The total storm sediment yield from the post logging period fits well within the variability exhibited by the calibration data (*fig. 2c*). There was no observed impact of timber harvest on total storm sediment yield.

Discussion

It should not be surprising that there is no detectable impact of the timber harvest that occurred in the NF Little Creek in 2008 on the hydrology and sediment yield of the NF the first winter after logging. The reasons for this are the scale and intensity of the timber harvest operations in the NF Little Creek watershed. In published results from the seminal watershed studies the size of the treatment watersheds were often very small, 70 to 101 ha, compared to the size of the NF Little Creek watershed, 291 ha (Brown and Krygier 1971, Rothacher 1970). The Caspar Creek study watersheds had comparable sizes, 508 and 425 ha (Lewis 1998).

But more important than the scale of the timber harvest activities is the intensity. In all of the seminal paired watershed studies the silvicultural system was clearcut and the percent area affected ranged from 30 to 100 percent (Brown and Krygier 1971, Fredriksen 1971). The exception, again, is the Caspar Creek study where the South Fork was harvested with an individual selection system but 100 percent of the area supported operations and 59 percent of the volume was removed (Lewis 1998). The silvicultural system for the NF Little Creek was individual tree selection, harvest operations were conducted on only 20 percent of the area above the gauging station and only three percent of the volume and basal area of the forest above the gauging station was removed.

Bosch and Hewlett (1982) report that it takes removal of forest canopy on at least 20 percent of the area of a watershed to detect an impact on annual water yield. While this analysis did not investigate annual water yield, the 3 percent volume and basal area removed is well below the published threshold of 20 percent. The scale and intensity of the harvest operations essentially precluded a significant effect on hydrology.



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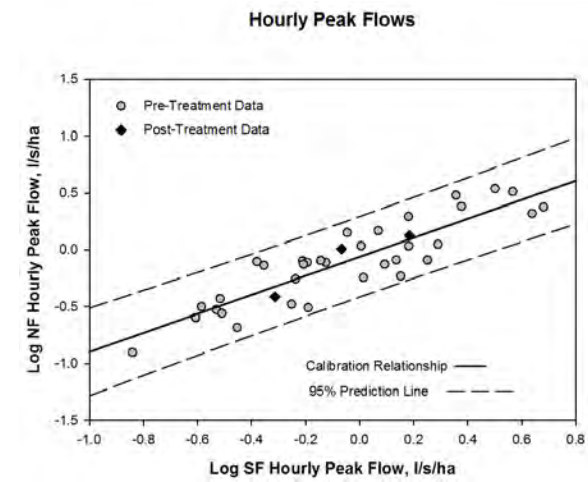

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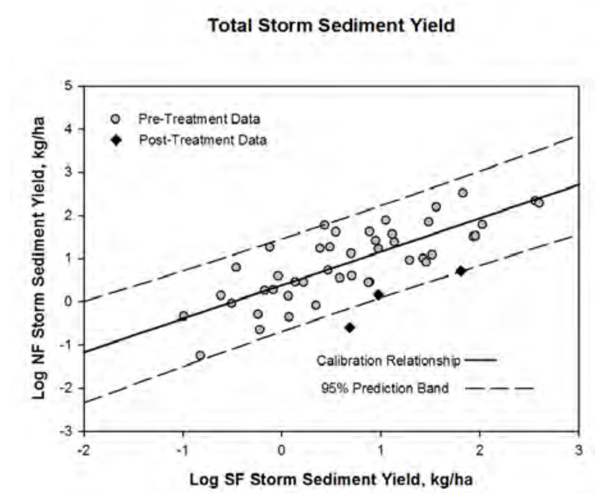

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Figure 2—Three graphs showing data for the control (SF) vs the treatment (NF) watersheds for the pre-treatment and post-treatment data for a) hourly peak flows, b) total storm quickflow, and c) total storm sediment yield.

There is a concern that given the paucity of post-harvest data this study may not give a true test of the impact of the forest practices used in the NF Little Creek on the hydrology and sediment yield of the watershed. This is not the case. While there are differing opinions regarding the impact the timber harvest on peak flows, those differing opinion are usually with regard to large peak flows (Grant et al. 2008). There is widespread agreement that timber harvest can and does affect small peak flows (Ziemer 1981) or the peak flows that may occur many times a year, the first winter after timber harvest occurs. The mean annual flood (MAF) for the NF Little Creek during the period of record, was 1.94 l/s/ha and the three post harvest peak flows, 0.49, 1.53, and 0.86 l/s/ha, were all below the value of the MAF. These are the size flows that could and should be affected by timber harvest the first winter after logging occurs. These flows represent an excellent opportunity to evaluate the impact of timber harvest on the hydrology of a watershed.

While the scale and intensity of the harvest combined with the size of the post-harvest peak flows may help explain the lack of an impact to the hydrology of the NF Little Creek, it doesn’t necessarily explain away the lack of significance in the sediment yield data. It does explain part of the lack of significance in the sediment yield data because sediment yield and streamflow are very tightly correlated. In all of the seminal paired watershed studies an increase in sediment yield was always accompanied by an increase in water yield and storm flow. However, the degree that the increased sediment yields are a function of increased stream power versus an increase in the erosion rate of the watershed has never been fully investigated.

Nonetheless, the two harvest entries for the Caspar Creek Watershed Study did conclusively exhibit the importance of contemporary Best Management Practices in mitigating increases in sediment yield associated with timber harvest. Accelerated erosion associated with timber harvest was reduced by roughly an order of magnitude when contemporary BMPs were used in the 1990s compared with the practices used in the 1960s and 1970s. A compelling hypothesis to put forward is that the combination of the reduced scale and intensity of timber harvest activities in conjunction with the use of contemporary BMP’s resulted in levels of accelerated erosion, which must occur, that remained within background variability.

Conclusion

Hourly peak flows, total storm quickflow volume, and total storm sediment yield were analyzed for 35 storms that occurred prior to timber harvest activities and three storms that occurred during the winter after timber harvest occurred. There is no evidence in any of the results that the timber harvest activities had any impact on the hydrology or sediment yield of the NF Little Creek watershed. This finding should not be surprising given the scale and intensity of the timber harvest activities. Timber harvest removed only about two percent of the basal area in the NF Little Creek watershed above the gauging station effectively negating a discernable impact on the hydrology. It is hypothesized that in the absence of a significant impact on the hydrology, the forest practice regulations and BMP’s were effective in keeping the accelerated erosion, which must occur, within background variability.

Acknowledgments

This research was supported by a grant from the California State University Agricultural Research Institute. Data from Little Creek was provided by the students of California Polytechnic State University San Luis Obispo, Swanton Pacific Ranch; specifically we thank Drew Loganbill, Drew Perkins, and Michael Gaedeke. We would also like to thank Steve Auten for information regarding the 2008 timber harvest in Little Creek.

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Summer Water Use by Mixed-Age and Young Forest Stands, Mattole River, Northern California, U.S.A.

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Abstract
Resource managers have noted a decline in summer flow levels in the last decade in the Mattole River watershed, Humboldt County, California. Reduced river flows pose a threat to endangered coho and chinook salmon in the watershed, as stream heating is inversely proportional to discharge. While the cause of the reduced flow is unclear, several factors have been cited: increased groundwater pumping from residential development in the area, regional climate shifts tied to global warming, and the recovery of forest cover after widespread deforestation in the 1950s and 1960s. The goal of this project was to gain insight into the effect of stand age and composition on forest water consumption. Quantitative information on tree and stand level transpiration was collected in order to inform comprehensive hydrologic budgets being developed for the Mattole River watershed under existing conditions and resulting from prospective forest management activities. Granier thermal dissipation probes were inserted into 18 Douglas-fir (*Pseudotsuga menziesii*) trees in mixed and even-aged stands in order to record water use over the course of the 2008 summer dry season. Trees ranged in size from 10 to 91 cm diameter at breast height (DBH). A tight relationship was found between sapwood area (cm^2) and water use (liters/season, $y = 7.68x – 638.6$, $r^2 = .86$). Strong positive relationships were also found between DBH (cm) and water use ($y = 92.40x – 1068.4$, $r^2 = .90$), and for basal area (cm^2) and water use ($y = 1.261x + 241.57$, $r^2 = .94$). The relationship between basal area and water use was much steeper for the youngest trees ($y = 3.42x – 233.15$. $r^2 = .76$), indicating a steep increase in water use with increasing tree size at the lower end of the size range. This information was used to model stand level water use with the current composition and under future scenarios using Forest Vegetation Simulator. Results indicate that the water use of Mattole River forests will decline in coming decades as the high numbers of young (< 5 cm DBH) trees decline from canopy closure and stem suppression.

Key words: sapflow, transpiration, *Pseudotsuga menziesii*, forest management, water balance, coastal California, Forest Vegetation Simulator.

Introduction

Resource managers have noted a decline in summer flow levels in the last decade in the Mattole River watershed, northern California, U.S.A. (*fig. 1*). While the cause of the reduced flow is unclear, several factors have been suggested: increased groundwater pumping from residential development in the area, regional climate shifts tied to global climatic disruption and the recovery of forest cover after

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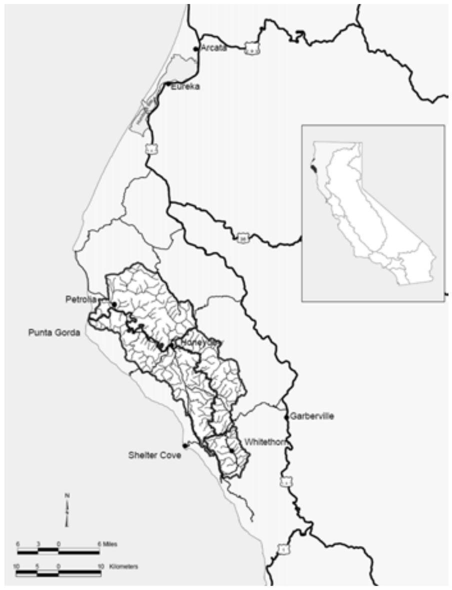

Figure 1—Study site location, Mattole River watershed, California.

widespread deforestation in the 1950s and 1960s. Reduced flows pose threats to aquatic conditions as the amount of stream heating is inversely proportional to discharge. Tributary drainages have gone completely dry in recent summers, or been limited to isolated pools. Mattole River salmonid populations are declining for three threatened species under the U.S. Endangered Species Act: coho salmon (*Oncorhynchus kisutch*), chinook salmon (*O. tshawytscha*), and steelhead trout (*O. mykiss*).

The goal of this research is to gain insight into the effect of stand age and composition on forest water consumption. By collecting quantitative information on tree and stand level transpiration we hope to inform comprehensive hydrologic budgets being developed for the Mattole River watershed under existing conditions and resulting from prospective forest management activities.

To examine water use for trees of different size and stands of different composition we instrumented 11 trees in a young stand and seven trees in an adjacent mature stand with thermal dissipation probes. We then used relationships between tree size and water use in conjunction with outputs from the forest growth model Forest Vegetation Simulator (Dixon 2002) to estimate changes in stand water use as the trees mature, canopy closure and stem exclusion processes take place. Given the tight relationship between leaf area index and sapwood area observed by other researchers (Waring et al. 1982) we hypothesized that larger dominant trees found in the mature stand would show greater maximum transpiration rates, daily and seasonal water consumption levels than younger trees and suppressed, understory trees, but that on a stand basis, water use would be reduced in mature stands because of lower numbers of stems per acre.

Study area

The study was conducted at two adjacent sites near Ettersberg, California in the Mattole Watershed, Humboldt County (40°9′ N, 123°59′ W, elevation ~300 m). Dominant tree species were Douglas-fir (*Pseudotsuga menziesii*), tanoak (*Lithocarpus densiflora*), and pacific madrone (*Arbutus menziesii*). Stands were located midslope, with SE aspect. Southern Humboldt County has a Mediterranean climate with cool rainy winters and hot dry summers. Average annual precipitation is 110 cm.

The first site was a mature stand composed of large Douglas-fir (61 to 91 cm Diameter at Breast Height (DBH)) with a mid story of tanoak in the larger size classes (30 to 46 cm DBH), and an understory of smaller tanoak and suppressed Douglas-fir. The stand was closed except to the east, which was bordered by a dirt road. The second site was a densely stocked with small Douglas-fir (10 to 25 cm DBH) with occasional larger black oak and pacific madrone, and a small component of tanoak. This stand was estimated to be 20 years old. The canopy was closed.

Materials and methods

The technique used for this study was measurement of individual tree water use with thermal dissipation probes (Granier 1985). In this method pairs of probes are inserted in predrilled holes in the xylem of the tree bole. The probes are linked to dataloggers which continually record temperature. The upper probe of each pair is heated. The lower probe serves as a reference. The rate of sap flowing through the hydroactive xylem is determined by the amount of cooling of the heated probe relative to the reference probe.

The method has been validated by comparison with other techniques such as heat pulse methods (Swanson 1994), gravimetric measurements of pot-grown saplings (McCulloh et al. 2007), and micrometeorological flux measurements (Kostner et al. 1998). Thermal dissipation probes have been used in a numerous applications since their development. Examples include evaluating sapflow in Douglas-fir stands (Moore et al. 2004) and scots pine and spruce stands (Granier et al. 1996).

TDP-80 probes (Dynamax, Inc., Houston, Texas) were used at the mature stand. Instrumented trees are described in *table 1*.The probes consist of 80 mm hypodermic needles fitted with copper-constantan thermocouples. They are inserted into predrilled holes in the sapwood in pairs with an upper probe 40 mm above a lower probe. The upper probe has a constantan heating wire in addition to the thermocouple wire. The lower probe acts as a reference probe, giving the temperature of the sapwood without heating. The temperature difference between the needles is at its greatest when sap flow velocity is minimal and as flow increases the upper needle cools and the difference in temperature decreases. This temperature difference is recorded with Campbell Scientific 1000x Dataloggers (Campbell Scientific Inc, Logan, Utah). The TDP-80 probes have two thermocouples mounted in each probe, allowing for measurement of temperature changes at two depths within the sapwood. TDP-30 and TDP-50 probes were used at the young stand. These probes are a more appropriate length for smaller trees with thinner sapwood (30 mm and 50 mm needles respectively) and have only one thermocouple per needle.

Average sap flow velocity V (cm/s) is calculated from the expression:

$$V = 0.0119*K^{1.231}$$

where the dimensionless parameter K is:

$$K = (\Delta Tm - \Delta T)/\Delta T$$

ΔT is the measured difference in temperature between the heated probe and the reference probe. ΔTm is the maximum value of ΔT when there is no sap flow. (Grenier 1987).

Detailed observations have indicated that when a portion of the probe is inserted into inactive xylem, heat is attenuated more slowly, introducing a bias into probe readings (Clearwater et al. 1999). To adjust for this bias, a correction factor was used when the probe lengths were longer than the measured sapwood depth.

Table 1—*Characteristics of trees instrumented with sapflow sensors, Mattole River watershed, California, 2008.*

Species*	Dia. at breast ht (cm)	Sapwood cross section (cm^2)	Basal area (cm^2)	Probe length[#] (cm)
Douglas-fir	9	52	66	5
Douglas-fir	12	67	107	5
Douglas-fir	12	78	117	3
Douglas-fir	14	95	148	8
Douglas-fir	16	139	195	3
Douglas-fir	18	166	241	5
Douglas-fir	19	190	277	5
Douglas-fir	19	227	293	5
Douglas-fir	20	207	324	5
Douglas-fir	27	401	569	5
Douglas-fir	37	270	1051	8
Douglas-fir	42	397	1413	8
Douglas-fir	55	685	2364	8
Douglas-fir	76	771	4591	8
Douglas-fir	90	1023	6422	8
tanoak	38	865	1110	8
tanoak	52	1490	2088	8

*Douglas-fir (*Pseudotsuga menziesii*), tanoak (*Lithocarpus densiflorus*).
#Trees with 3 and 5 cm probes had one sensor per probe and one probe was placed in each tree. Trees with 8 cm probes had two sensors per probe, and two probes per tree.

The recommended correction factor (Clearwater and others 1999) is:

$$\Delta Tsw = (\Delta T - b\Delta Tm) / a$$

where ΔTsw is the change in temperature of the sapwood, ΔT is the recorded change in temperature of the probe relative to the reference probe, ΔTm is as defined above, a is the proportion of the probe in active xylem, and b is the proportion of the probe in inactive xylem.

The correction factor was not applied for the 80 mm probes as they have two sensors within the probe, one at 15 mm and one at 70 mm. Sap velocity (cm/s) is converted to sap flow rate (cm^3/s) by multiplying by the sapwood area of the tree, determined by coring techniques. The flow rate is then converted to volumes by summation of rates over the time period of interest.

Thermal shielding was installed to insulate the probes from direct sunlight. Two 48 hr power-down tests were conducted in which the heating elements were turned off to verify that the probes were properly insulated, with minimal heating from solar radiation. Probe readings were taken every 60s with the mean value recorded every

30m on dataloggers equipped with 32-channel multiplexers (CR1000, Campbell Scientific, Inc., Logan, UT).

Sapwood area was determined for each tree by the following procedure. Tree DBH was measured with a tape. The thickness of the bark was subtracted from the DBH. The thickness of the sapwood was determined by radial increment boring. Water conducting vascular tissue was identified by semi-translucent properties. Once the radius of the tree and the radius of the sapwood were measured, the radius of the heartwood could be estimated and the following equation applied:

$$SA = \pi (R_T^2 - R_H^2)$$

where SA is sapwood area, R_T is radius of tree (without bark) and R_H is radius of heartwood.

Basal area (BA) was calculated for all plots using tree diameter and the plot expansion factor for a ½ ha (1/5th ac) circular plot to get total BA/ac (cm^2). All trees were tallied within the plot. Species, diameter, and crown ratio were collected on every tree.

Continuous data was collected at 30 min intervals from July 6 to October 8, 2008. Data was summed into 24 hr daily totals. Daily totals were summed to create growing season totals for the period. Regression equations were developed between growing season water use and tree diameter, sapwood diameter and basal area. Sap flux density was calculated by converting liters to kg water, then dividing by sapwood basal area (m^2).

The next step was to generate estimates of seasonal water use for representative stands of the Mattole River watershed moving forward through time. A representative stand was created for a year 2005 starting point using an average created from four Forest Inventory Analysis plots (FIA). FIA is the U.S. Department of Agriculture's national forest inventory program (Woudenberg et al, 2010). The stand composition was then modeled through time using the U.S. Department of Agriculture, Forest Service Forest Vegetation Simulator (FVS, Dixon 2010). Output was generated for 20 size classes (0 to 101 cm) of DBH at 5 year increments for the period 2005 to 2055. Output used for this study included average DBH within each size class, trees/ha, and basal area/ha. FVS output was then used to estimate total growing season water use for each age class at each 5 year increment using the regression equation developed for basal area.

Results

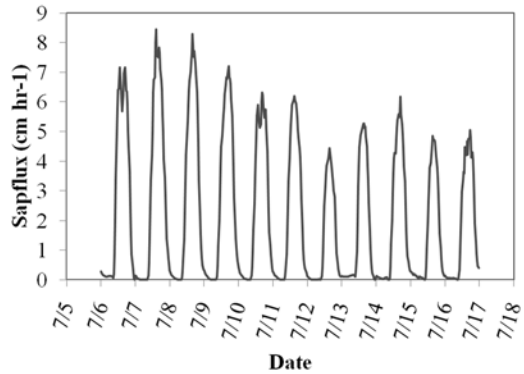

Figure 2—Daily sapflux, June 6 to17, 2008, Mattole River watershed.

Sapflow was observed to rise and fall with solar radiation levels. Representative samples of the daily continuous and weekly data for individual trees are shown in *fig. 2* and *3* respectively. A steady decline in transpiration was observed over the course of the season as soil moisture declined. This matches a drawdown in discharge in the Mattole River (*fig. 3*). The first rainstorm of the fall (Oct 2, 2008) caused a spike in transpiration readings. Total water use for each tree over the growing season is shown in *table 2*. A tight relationship was found between Douglas-fir sapwood area and water use ($y = 7.6861x – 638.59$, $r^2 = .86$). Strong positive relationships were also found between Douglas-fir diameter at breast height (DBH, cm) and water use ($y = 92.397x - 1068.4.4$, $r^2 = .90$), and for basal area (cm^2) and water use ($y = 1.2611x +241.57$, $r^2 = .94$). Regressions were also strong when the smaller trees in the young stand were analyzed separately. The relationship between basal area and water use was steeper for the young stand than for the larger trees ($y = 3.423x – 233.15$, $r^2 = .76$) indicating a steep increase in water use with increasing tree size at the lower end of the size range. To compare water use by tree size, seasonal water use for each tree was divided by the basal area of the tree. A higher range of water use was found for the smaller trees, as compared to the higher end of the size range. Trees below 30 cm DBH had an average water use/ unit basal area of 2.2 (± 1.2) l per cm^2, while trees larger than 30 cm DBH averaged 1.2 (± 0.5) l per cm^2. Mean sap flux density ranged from 123 to 976 kg per m^2 per day. A linear trend was not observed between sapwood area and mean sap flux density.

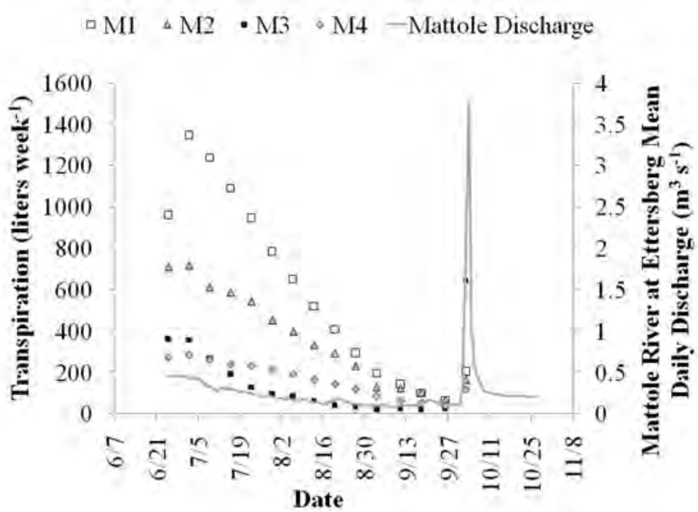

Figure 3—Sapflow in mature Douglas-fir over the 2008 growing season and Mattole River discharge. M1 to M4 refer to individual trees.

Table 2—*Water use in instrumented trees, Mattole River watershed, June 25 to October 8, 2008.*

Basal area (cm^2)	Season water use (liters)	Water use per unit basal area (liter cm^{-2})	Sap flux density (kg m^{-2} d^{-1})
Douglas-fir (*Pseudotsuga menziesii*)			
66	150	2.3	307
107	291	2.7	462
117	122	1.0	166
148	133	0.9	149
195	415	2.1	318
241	458	1.9	294
277	219	0.8	123
293	1415	4.8	663
324	711	2.2	365
569	1786	3.1	474
1051	2476	2.4	976
1413	2343	1.7	628
2364	2038	0.9	317
4591	5383	1.2	743
6422	8948	1.4	931
Tanoak (*Lithocarpus densiflorus*)			
1110	1817	1.6	223
2088	5849	2.8	418

There were not sufficient instrumented tanoak trees to create independent regressions. The data from the two tanoak trees appears to follow the general linear relationship of the Douglas-fir. Adding the two data points to the Douglas-fir data changes the regression equation only minimally ($y = 1.312x + 355.52$, $r^2 = 0.87$).

The initial forest species composition for 2005, determined from FIA plots, indicates a dominance by small Douglas-fir and tanoak trees (10 to 30 cm DBH), with a few larger pacific madrone (20 stems per ha). *Figure 4* shows basal area.

Forest Vegetation Simulator runs indicated a rapid attainment of canopy closure and stem suppression with a steep drop off in individual tree numbers over time. For example in the smallest size class (0 to 5.1 cm DBH), Douglas-fir drops from 525 to 95 trees per ha by 2055, and tanoak drops from 1929 to 887 trees per ha by 2035. As would be expected, a gradual increase in basal area is observed for surviving trees.

These trends are reflected in estimates of stand water use over time generated from applying water use data from this study to the Forest Vegetation Simulator output for future stand conditions. A drop in overall water use was shown for Douglas-fir species (489 to 385 m^3 per ha, *table 3*). This decrease resulted directly from the greatly reduced numbers of young trees. While mid and larger sized trees increased their water use over the time period, they were a much smaller proportion of total water use, so their impact was minimal. For example, summing the water use of the smallest four size classes (0 to 15 cm, DBH) we find a decrease over 50 years from 127 to 23 m^3 per ha, while the largest four size classes (86 to 102 cm, DBH) increased from 65 to 101 m^3 per ha.

Similar results were found for the tanoak and Douglas-fir. While water use of larger trees increases, its effect is eclipsed by the decrease in water use from stem suppression of younger trees. Overall water use drops from 2188 to 1506 m^3 per ha over the season. The largest trees increase water use from 46 to 170 m^3 per ha, while the smallest trees decrease from 813 to 23 m^3 per ha.

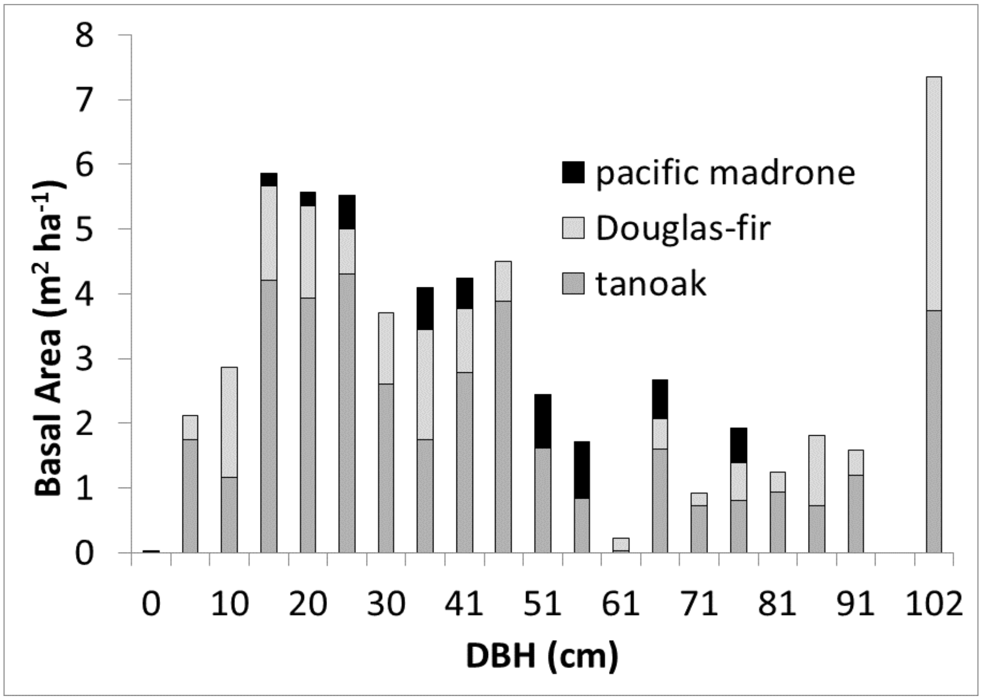

Figure 4—Mattole River watershed forest composition, basal area by DBH size class.

Discussion

Thermal dissipation probes were successfully used to quantify individual tree water use over the growing season, June to October, 2008. The October spike in water use resulting from a rainstorm indicates the sensitivity of tree growth to soil moisture and indicates the sensitivity of the method to fluctuations in tree growth. This was also observed by correlation of solar radiation measurements to measured sapflow: it decreased during cloudy days. Mean sap flux density values closely match values measured by Moore et al. (2004) for similar-aged Douglas-fir in western Oregon.

A tight relationship was found between measures of tree size: basal area, DBH, sapwood area, and tree water use. Previous research (Kostner et al. 1998) indicates that the number of trees measured were sufficient to characterize stand variability for Douglas-fir but not for tanoak. The high r2 values of the regressions are evidence that the variability of individual tree water use was sufficiently captured by the size of the sample.

Table 3—*Growing season water use in Douglas-fir estimated from sapflow measurements and Forest Vegetation Simulator output, Mattole River watershed, 2005 to 2055.*

Size Class (cm)	2005	2015	2025	2035	2045	2055
0.0	127	97	69	34	28	23
5.1	47	29	22	33	26	20
10.2	83	76	54	8	5	1
15.2	37	30	43	76	36	15
20.3	28	34	25	23	53	47
25.4	12	30	27	24	13	31
30.5	17	11	24	22	20	16
35.6	26	16	11	20	27	15
40.6	14	31	27	17	15	31
45.7	9	10	24	25	21	16
50.8	0	16	14	23	25	2
55.9	0	0	11	24	26	2
61.0	3	0	8	4	23	40
66.0	6	3	0	8	5	15
71.1	3	4	4	5	10	3
76.2	8	6	2	2	6	15
81.3	4	10	6	3	1	5
86.4	14	4	10	7	4	2
91.4	5	15	13	12	7	4
96.5	0	4	12	16	15	0
101.6	46	48	51	62	79	95
TOTAL	489	474	456	448	445	395

Individual tree water use was then combined with forest growth model predictions to estimate water use under future conditions. The predicted changes in forest vegetation should be accurate and representative of the Mattole as the input conditions were from four standard Forest Inventory Analysis plots surveyed in the watershed, the FVS model has been widely applied and validated, and was run by experienced practitioners.

The model assumes a stable climate, no thinning or harvest activity, and lack of major disturbance such as fire, windstorm or severe disease outbreak. If we make these assumptions for the purposes of isolating the impact of forest vegetation on water yield, the model output points to two strong and competing trends in forest composition that would stand drive water use. The first is the growth of the stand. We have shown a linear increase in water use with tree size. As the stands age over the next fifty years they will be composed of larger trees and might thus be expected to use more water. The second trend is stem suppression. As the trees grow the canopy closes and the trees begin to compete for available light. Suppressed trees eventually die out. The current composition of high numbers of very small (<5 cm DBH) trees indicates that the total number of trees will drop substantially in coming decades.

Our results strongly support the conclusion that stem suppression will be the dominant trend affecting water use in the Douglas-fir dominated portions of the Mattole river watershed. Water use will be expected to decline in a steady fashion as the number of trees declines. A further implication of this finding is that clearcut harvesting of existing stands would not be beneficial for water yield in the basin beyond the initial regeneration period. It would result in a new crop of dense small trees, and delay the stem exclusion stage that much longer. Selective harvest of small and mid size trees might be expected to increase water yields without producing a thicket of young trees if remnant trees were able to quickly grow into the light gaps.

The results are less conclusive regarding tanoak which is the more dominant species in the Mattole River watershed because only two trees were instrumented. The two trees had similar mean sap flux density values as the Douglas-fir. The water use predictions made using the basal area regression that included tanoak would indicate that stem suppression is also the dominant process affecting water yield for this species over the next fifty years. The numbers of small tanoak in the watershed are quite high.

One way to assess the robustness of this conclusion is to test the sensitivity of the conclusion to input assumptions. To perform this sensitivity test we increased the water yield slope by 50 percent for tanoak (from 1.3123x + 355.52 to 1.968x + 355.52) and then looked at resulting water use predictions for the whole stand over the period 2005 to 2055. The water yield continued to show a decrease over this time period (1927 to 1439 m^3 per ha) as compared to the original (1699 to 1111 m^3 per ha).

Previous research supports our finding of diminished water use with stand age. A sapflow study in western Oregon (Moore et al. 2004) determined that young mature Douglas-fir stands (40 yr) had 3.27 times higher water use than old growth Douglas-fir (450 yr) stands for a similar time period (June to October, 2000). They attributed this difference to greater sapwood area per unit basal area and greater sap flux density in the younger stand, as well as greater numbers of mesic angiosperms. Phillips et al. (2002) found reduced hydraulic conductance in older Douglas-fir stands as compared to young and mature trees.

Conclusions

Sapflow measurements in the Mattole River watershed show strong relationships between total seasonal tree water use and basal area, DBH and sapwood basal area for Douglas-fir. Water use measurements combined with stand growth modeling indicate that the water use of Mattole River forests will decline in coming decades as the high numbers of young (< 5 cm DBH) trees decline from canopy closure and stem suppression. Decreased water use is expected to have beneficial effects on aquatic ecosystems.

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Sediment Yield Response to Sediment Reduction Strategies Implemented for 10 Years in Watersheds Managed for Industrial Forestry in Northern California[1]

Kate Sullivan[2]

Abstract

For the past decade, the productive forestlands now owned and operated by the Humboldt Redwood Company have been managed with low impact practices designed to reduce sediment delivery according to voluntary agreements and regulatory requirements of state and federal agencies. These timberlands located in the erosive sedimentary terrain of the northern coast of California have been extensively roaded and several generations of redwood dominated forests have been clear-cut logged since the 1860s. Intensive watershed and property-wide studies of sediment processes within the past 50 years when information is most reliable have created watershed sediment budgets and documented significant sediment impacts from past management activities. Forest operations now include geologic hazard avoidance and an extensive road upgrading and removal program to minimize the dominant sediment sources. Over the past 10 years, suspended sediment and streamflow have been continuously monitored at a number of locations in the mainstems and major tributaries of Freshwater Creek and Elk River. In this paper we explore the extent of sediment reduction in the watersheds and effects on water quality that may be evident in the relatively short monitoring record.

Key words: sediment yield, suspended sediment, water quality monitoring, logging effects

Introduction

Long-term research on logging effects in coastal California watersheds has found that sediment can be expected to increase concurrent with clearcut timber harvest for some duration, with effects diminishing with time (Lewis 1998, Ziemer 1998). Initial response levels vary with the type of erosion processes that are disturbed and the type of management practices applied (Lewis 1998). However, past effects may be persistent and contribute to sediment yield for decades (Klein et al. 2008, Lewis and Keppler 2007, Ziemer 1998).

The majority of timberlands in Elk River and Freshwater Creek were acquired by the Humboldt Redwood Company (HRC) after financial reorganization of the Pacific Lumber Company in 2008. The watersheds have been actively logged since the 1860s using clear-cut harvest as the primary logging method over the past several decades and has a road system built to various construction standards over the last 50 years. Since 1999, The Pacific Lumber Company and now HRC has been steadily

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[1] This paper was presented at the redwood science symposium: coast redwood forests in a changing California. Jun 21-23, 2011, Santa Cruz, California.

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working to reduce sediment with a combination of state-of-the-art road construction practices, a commitment to reconstruction and removal of older roads, and wet weather use limitations that prevent damage to roads and prevent sediment delivery to streams. Harvest related sediment is controlled through geologic hazard identification and geologist field investigation during timber harvest plan layout, rate of harvest limitations in these two watersheds, and practice of uneven-aged silviculture. Should we realistically expect sediment decline and has any occurred over the past decade?

Study area

Freshwater Creek and Elk River are adjacent watersheds that respectively drain to the north and south ends of Humboldt Bay on the coast of northern California ($40^{o}44'$ N, $124^{o}2'$ W) (*fig.1*). The watershed area of Freshwater Creek is 80 km^2 and Elk River is 111 km^2. The region has warm to hot dry summers and mild wet winters with moderate storm intensities. The watersheds are predominantly forested with second-growth stands of various ages comprised of redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*).

Figure 1—Overview of study watersheds in Humboldt County with the location of hydrologic monitoring stations.

Unlike the controlled experiment at Caspar Creek, this water quality monitoring project can only be described as a messy experiment. There was no pre-measurement period—the experiment was joined in progress along with whatever past impacts and possible recovery may have occurred from high sedimentation documented in the sediment budget from a significant storm that occurred in 1997. Management is ongoing throughout the study period and multiple activities that could either increase or decrease sediment happen simultaneously. Most of the tributary basins have experienced both harvest and road construction or deconstruction during the 8 year monitoring period, often in the same year. As the experiment proceeds, timber management objectives dictate where, when and what activities occur. All of the current management activities are designed to prevent and minimize new sediment input while restoration activities remove existing sources. Is the net effect of the system as a whole leading to a reduction in the sediment yield? An extensive water quality monitoring project was initiated in 2002 to answer these questions.

Watershed sediment budgets

Sediment sources have been extensively studied in Watershed Analysis projects sponsored by the Pacific Lumber Company (PWA 1998, 1999) and in support of TMDL related projects (Manka 2005, NCRWQCB 2011; PWA 2006, 2008). The erosion studies have characterized past and current sediment discharges and compiled sediment budgets (Reid and Dunne 1996) using remote sensing and aerial photography, extensive field surveys, and empirical physical modeling. Although the details within the sediment budgets vary somewhat among the various studies, and remain a topic of debate (NCRWQCB 2011), the general level of sediment as shown in *figure 2* is similar among the studies. HRC updates the budgets based on ongoing landslide inventories and other adjustments with monitoring projects and restoration work. The current estimated budget is also shown in *figure 2* along with the average sediment yield described in the next section. There has been reduction in sediment sources in each watershed over the decade, largely due to significantly reduced landslide rates and road improvements.

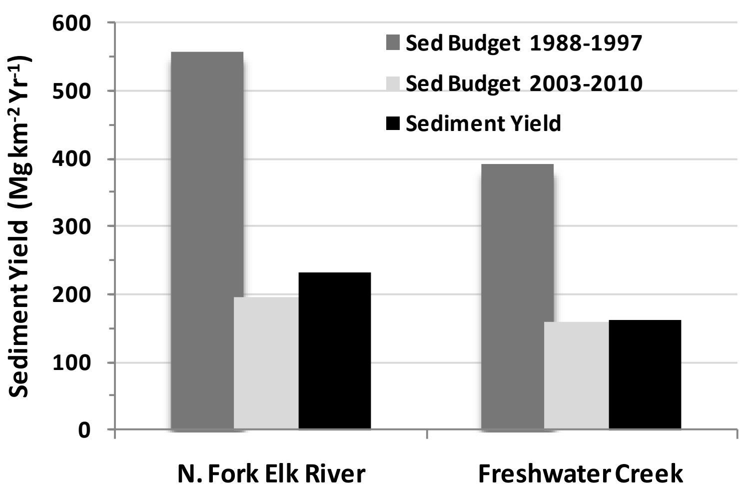


Figure 2—Sediment budget estimates for the period prior to (1988 to 1997) and following adoption of new management practices in the watersheds (2003 to 2010).

Sediment budgets have been a helpful tool that adds insight into expected sediment sediment yield and management effects, especially when available at the level of detailed study performed in these watersheds. Current sediment budgets match observed average sediment yield adjusted for bedload fairly well (*fig. 2*). The study has benefited from a wide range of rainfall during the short 8-year study period that has produced a representative average.

Stream sediment measurement methods

An extensive water quality monitoring program was installed by The Pacific Lumber Company in Elk River and Freshwater Creek in October 2002 and sites are currently operated by HRC. Sites are located in a number of subwatersheds and in the mainstems (*fig. 1*). Sediment and streamflow are measured at nine sites in the Freshwater and 11 sites in the Elk River watersheds. Contributing basin areas range from 0.13 to 112 km^2. The project includes the basin containing the Headwaters Forest Reserve in the Little South Fork Elk River (534). This watershed has an undisturbed old growth forest with only a very minor management impact. This site was initially installed by Humboldt State University (HSU) on Bureau of Land Management Property (Manka 2005) and has been maintained by HRC since 2007.

All sites are run of the river installations. Streams are equipped with continuous measuring turbidimeters and depth recorders that sample every 15 minutes from October to June. Physical sediment samples are collected and streamflow is measured. This combination of measures allows the continuously recorded turbidity and depth to be translated to streamflow (m^3s^{-1}) and sediment load. These are summed to produce annual sediment load, expressed in metric tons (Mg), or sediment yield per unit watershed area expressed as Mg km^{-2}.

Results

A comprehensive presentation of results for all stations and parameters of interest is beyond the scope of this paper. The focus is trends of annual sediment yield within the 8-year period from 2003 to 2010. There is a wide and typical range in sediment yield among sites as illustrated for an average rainfall year (hydrologic year 2010) in *figure 3A*. The old growth watershed in the headwaters of Elk River (534) had sediment yield of 5.9 Mg km^{-2}. Sediment yield is similar among the Freshwater sites, averaging about 50 Mg km^{-2}. This is about 10 times greater than the export from the old growth watershed. Sediment yield is more variable in Elk River reflecting a variety of factors. The old growth site has no management influence, subwatershed 519 has substantial bank erosion and streamside landslides from legacy channel filling (PWA 2008), and stations 510 and 511 are the mouths of the large river basins (drainage areas 50 and 57 km^2, respectively). The relationship between sediment yield and watershed size is shown in *figure 3B*. Site 533 is found in the westernmost portion of the watershed within the Hookton geology, a spatially minor but particularly erosive formation with considerable soil disturbance from past logging.

Data from four sites is used to illustrate the annual pattern of sediment yield for the period from 2003 to 2010 widely observed among most of the sub watersheds in the study (*fig. 4*).

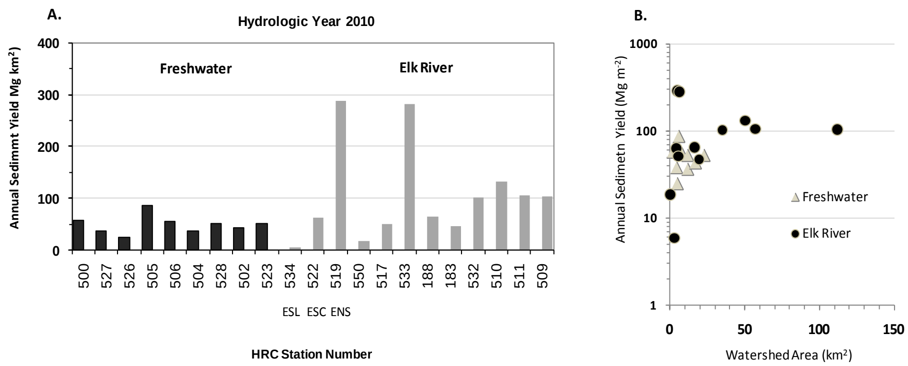


Figure 3—A) Sediment yield in hydrologic year 2010 (Oct 2009-Sept 2010) at each of the hydrologic stations. Original HSU labels for three sites discussed in Manka 2005 are indicated. B) Sediment yield as a function of basin area.

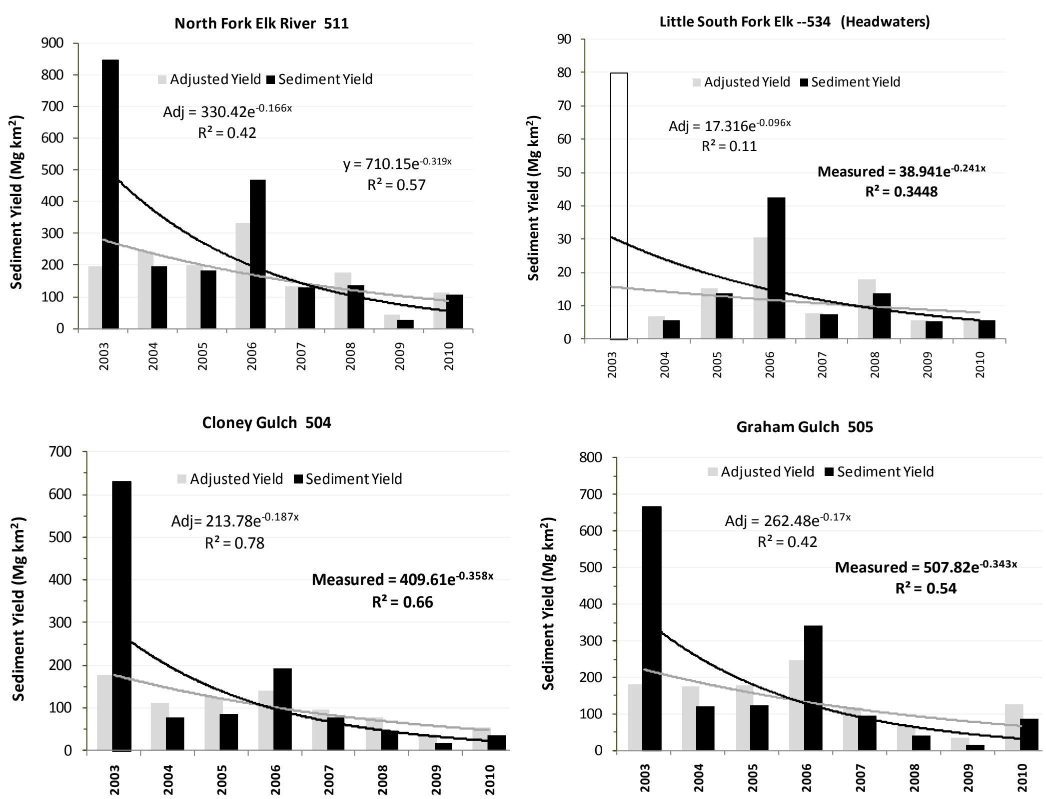


Figure 4—Annual sediment yield at four stations in the study area. A value for site 534 for 2003 has been estimated. The adjusted sediment yield has been normalized for annual climate characteristics using the erosivity index.

The four sites were selected to include a large river (511), the old growth unmanaged site (534), and two moderate sized subwatersheds, each of which had timber harvest and road reconstruction work during the interval (504 and 505). Other stations not shown follow a similar pattern evident at these four.

There is a clear decreasing trend in sediment yield during the measurement period. This largely follows the rainfall and storm pattern experienced during the period. The study period began in hydrologic year 2003. A storm in December 2002 set a number of rainfall records for the 122-yr record at the National Weather Service site in Eureka, including maximum daily rainfall, and caused an estimated 60-yr return interval flood in the study area (Dhakal and Sullivan 2005). Hydrologic year 2009 was a dry year with rainfall at 75 percent of normal and no large storms. Note that site 534 was not yet operative in 2003 and a value for that year has been estimated for illustrative purposes in this figure only.

Beginning the monitoring project coincident with the large storm assures a decreasing trend in sediment yield and related turbidity measures within the record. Hydrologic analysis of water quality trends often requires some method for removing the influence of climate (Helsel and Hirsch 2002). This paper introduces a climate detrending parameter that can be simply calculated with standard reported rainfall data to represent the climate influence, referred to here as the erosivity index. It is calculated as:

Erosivity Index = Annual Rainfall (in.) x Maximum Daily Rainfall (in.)

The index is very efficient at explaining sediment yields in the study area as shown for one representative site in *figure 5*. Sediment yield is highly correlated with the annual maximum peak flow (*fig. 5A*). This storm typically carries between 40 and 80 percent of the annual load each year. Although only rainfall is used in the calculation of this index, it is highly correlated with the annual peak flow (*fig 5B.*) Thus the index shares the same high correlation with sediment yield as the annual peak flow (*fig. 5C*). It is clear from *fig. 5* that one could also use the annual peak flow to index storm effects on sediment yield. One could also fine tune the short-term rainfall metric if the weather records are available. For comparing many stations, the erosion index has the advantage of applying commonly to all sites and it is surprisingly effective at capturing the elements of rainfall that lead to the observed sediment load.

This method based on rainfall was conceived with the goal to meaningfully characterize storm conditions that explain the erosiveness of storms with one simple parameter. The method helps create historical perspective on the relative sediment generating capacity of past storms based on climate records that may exist when long-term flow records may not. This is a question that arises often when constructing sediment budgets backwards in time or even in creating context for modern storms. The historical erosion index reflecting the 125-year rainfall record at the nearby Eureka NWS weather station is shown in *fig. 5D*. The record setting conditions of 2003 were equaled only in 1890. This storm generated at least a 60-year flood event at the hydrology stations in the study area. The period from 1997 to 1999 was a period of significantly higher than average erosivity. With the exception of 2003, the index has been at or below the long-term average during the monitoring period. This index would not effectively represent rain-on-snow events, but these type of events are rare in these watersheds due to low elevation and coastal proximity and have not occurred in the recent monitoring period.

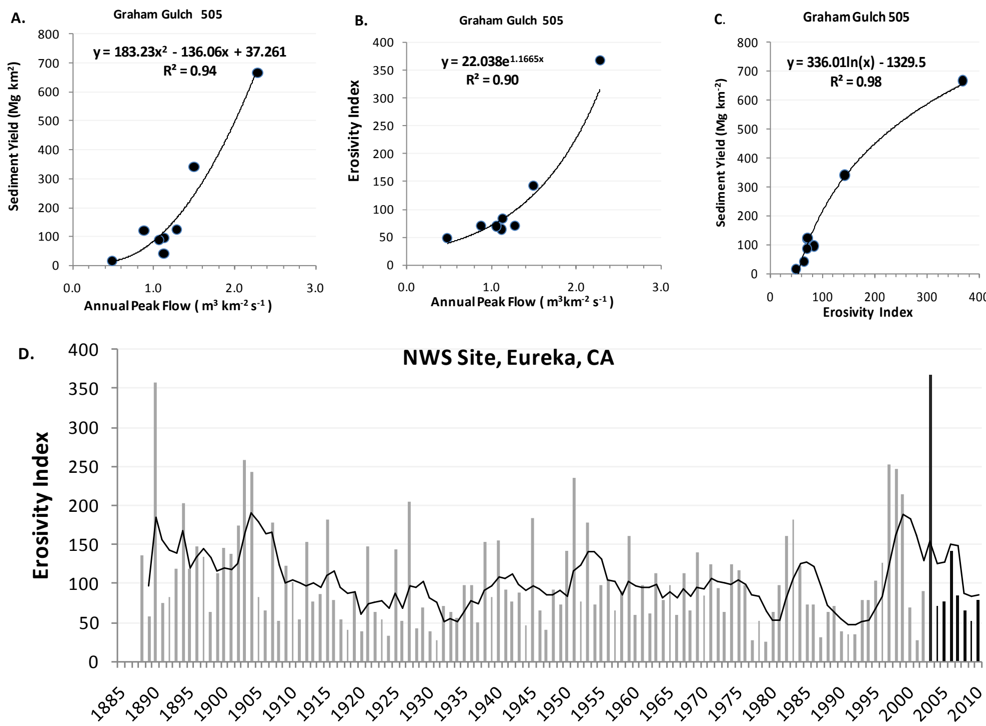


Figure 5—Erosivity index as a detrending variable. Relationships between A). annual peak flow and sediment yield, B) erosivity index and annual peak flow, C) erosivity index and sediment yield. D). Annual erosivity index of Eureka rainfall. Five-year moving average is also traced.

Correlation and multiple regression analysis was applied to the annual sediment yield to determine whether trends exist and whether there are effects of ongoing management activities. *Figure 6* shows the correlation (Pearson's) between various factors and annual sediment yield at individual sites and for all sites combined. The erosivity index as a surrogate for weather factors explains 89 to 99 percent of the variation in sediment yield and turbidity parameters during the study period and is highly significant at each individual site and for all sites combined. The correlation of the other factors was computed controlling for the erosivity index. There is a negative correlation between sediment yield and year at all sites. The correlation is strong but is statistically significant only at site 504 and when all sites are combined ($p < .05$). The trend is weak at site 502. This result suggests a generally decreasing trend in sediment yield with time independent of the climate but varying by watershed.

The correlation of sediment yield with several management related parameters is also shown in *fig. 6* for seven sites. These sites were selected because all data was available for the full monitoring period and to represent the range of watershed size and activity. For these variables, the correlation was controlled for both erosivity index and year. The correlation of annual harvest rate with sediment yield is quite mixed. At several sites the correlation is positive and at others it is negative. Overall, the annual harvest rate is not highly correlated with sediment yield. Klein et al. (2008) used the annual average harvest rate of the previous 10 or 15 years as an index of harvest effects. This parameter tends to have a negative correlation with sediment yield in most watersheds but is positive in several. The correlation is not statistically significant. Finally, the annual sediment removal is the sum of sediment saved at each road sediment source site in each watershed and it is used here as a measure of the amount of disturbance from road construction and deconstruction conducted in the preceding summer. This work potentially generates sediment with post construction site adjustment. This management activity generally has an increasing influence on sediment yield, and was statistically significant for all sites combined.

A multivariate linear regression on the combined data for all the parameters discussed was highly statistically significant and a backward stepping algorithm selected the same 3 significant variables for the combined group as the correlation analysis shown in *fig. 6* ($F = 158.65$, $p < .000$). The variables are erosivity index, year, and sediment removal.

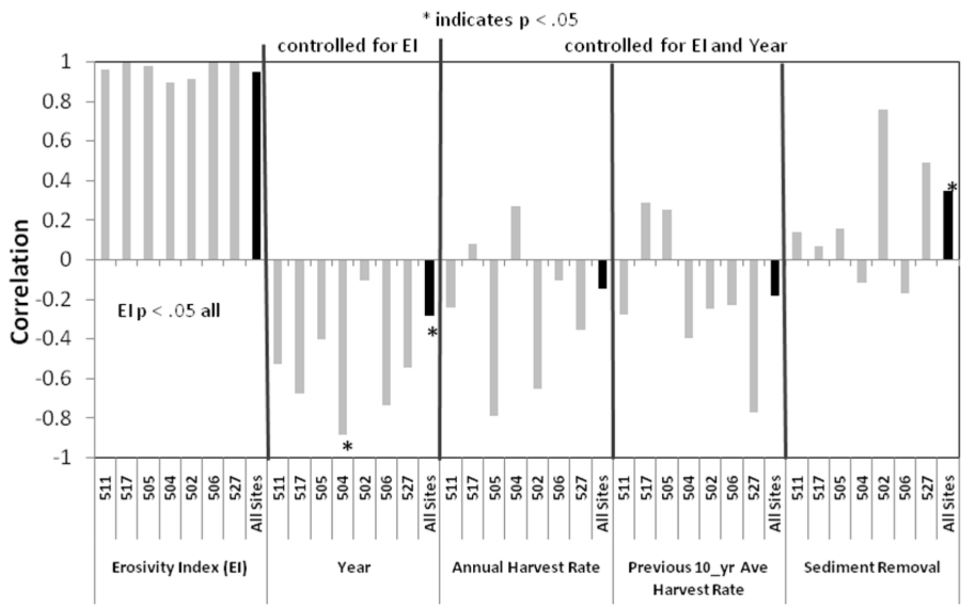
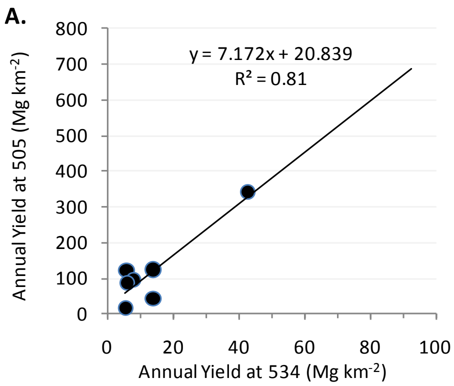


Figure 6—Correlations between annual sediment yield and various weather, time trend and management factors. Management factor correlations were computed by controlling for the erosivity index and year.

Comparison of the sediment yield from the managed sub-basins with the old growth basin (534) confirms that there has been a decreasing trend in sediment yield with time. One of the relationships between a managed basin and 534 is illustrated in *fig. 7A*. This relationship was computed for all sites and the residuals from this regression at each site are plotted in relation to time in *fig. 7B*. This analysis includes 2004 to 2010 when the old growth site (534) was operative. A check of the one high value evident in *figure 7A* did not appear to exert leverage on the results.



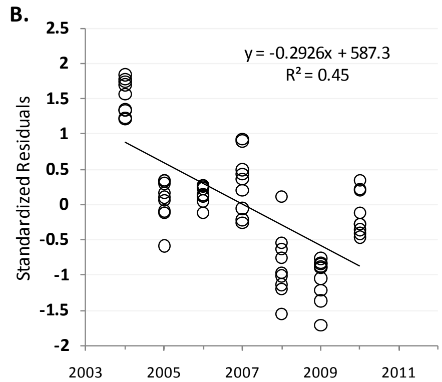


Figure 7—A. Relationship between annual yield in a managed watershed (505) with the old growth watershed (534). B. Residuals of the same comparison with eight managed watersheds as a function of year.

To visually see the effect of climate normalization, climate adjusted sediment yield is shown at the four sites in *figure 4*. This is computed by dividing the observed annual yield by the normalized erosivity index relative to the long-term median value. For example, in 2003 the erosivity index was 4.3 times the long-term median and the sediment yield was adjusted downward accordingly. Years with below median yield are adjusted upward. A negative exponential curve fit to the adjusted yield at the old growth site shows a decline of about 10 percent per year. The decline in slope averages about 18 to 20 percent for many of the managed watersheds but is higher in some. The difference in slope between the control and managed sites suggests that the sediment yields in many of the subwatersheds have declined at least 10 percent and perhaps as high as 18 percent per year during the 8 year study period.

Discussion and conclusions

There has been a small but persistent decline in sediment yield in most of the subwatersheds independent of climate and management factors from 2003 to 2010. The effect of ongoing management activities on annual sediment yield vary with subwatershed. Most have negative correlation suggesting that effects are not large enough to prevent observable recovery. Sediment yield in a few basins is positively correlated with ongoing activities. None are statistically significant.

It is questionable whether this rate of decline can continue. The sediment budgets indicate that roads and landslides were a large part of the sediment sources in the past. A large part of the road system has now been upgraded and landslides have been rare for a number of years. The forest on the landscape continues to mature, especially in steep streamside higher hazard areas that are now off limits to logging. The sediment budgets indicate that legacy sediment sources are now an increasing portion of the annual sediment load. These include such features as old skid trail crossings which are abundant in the two watersheds and eroding banks left from the initial logging (PWA 1999), both of which continue to contribute sediment on an annual basis. Finding solutions for these sources will be challenging.

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An Approach to Study the Effect of Harvest and Wildfire on Watershed Hydrology and Sediment Yield in a Coast Redwood Forest

Christopher G. Surfleet[1], Arne Skaugset[2], and Brian Dietterick[3]

Abstract

The Little Creek watershed, within California State Polytechnic University's Swanton Pacific Ranch, is the location of a paired and nested watershed study to investigate the watershed effects of coast redwood forest management. Streamflow, suspended sediment, and stream turbidity have been collected during storms at two locations on the North Fork Little Creek and at the outlet of South Fork Little Creek from 2002 until present. In 2008, the watershed area between the two monitoring stations on the North Fork Little Creek watershed was harvested with an individual tree selection silvicultural system within the Santa Cruz County Rules of the California Forest Practice Rules. The South Fork Little Creek was left unharvested to serve as a control. In 2009, the Little Creek watershed was burned by a wildfire. The wildfire eliminated our control watersheds for the proposed Before After Control Intervention (BACI) study design. We present an alternative approach at detecting harvest and fire effects that uses rainfall/runoff models, soil erosion models, and sediment runoff relations to simulate runoff and sediment yield from the watersheds. The models and sediment runoff relationships will be developed within the framework of an uncertainty assessment to simulate pre-harvest and pre-fire conditions for the North and South Forks of Little Creek. The modeled results will be used as the control for the study which had been eliminated due to the wildfire in 2009. We use the HBV hydrologic model and sediment runoff relations to demonstrate our approach. An example of post-harvest and post-fire runoff and sediment changes within the uncertainty of the approach are demonstrated.

Key words: streamflow, suspended sediment, stream turbidity, harvesting, modeling

Introduction

The Little Creek watershed, near Santa Cruz, California has been the location of a paired watershed study investigating the watershed effects of coast redwood forest management. The classic method for evaluating forest watershed effects has relied on paired watershed studies (e.g., Bates 1921; Hewlett 1971; Rice et al. 1979) using statistical analysis within a Before After Control Intervention (BACI) study design. In 2009 a fire burned both the treatment and control watersheds of Little Creek. With the loss of a true control watershed following the 2009 Lockheed fire we propose an alternative approach using hydrology and sediment models to discern forest harvest and fire effects on watershed hydrology and sediment yield. A modeling approach to discern treatment effects on watershed hydrology in paired watershed studies has been used in other studies (e.g., Lørup et al. 1998, Seibert and McDonnell 2010, Zegre et al. 2010). The modeling approach offers the benefit of eliminating the need

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for control watersheds, allows testing of alternative treatments, or the ability to evaluate long time durations not easily achieved in field experiments. However, the use of models represents additional uncertainty depending on model capabilities to represent the physical processes or the accuracy or availability of input data to the model. Therefore watershed study using models cannot be used in all situations and results need to be interpreted within the uncertainties associated with model use.

Methods

Study area

The Little Creek study watersheds are located on the Swanton Pacific Ranch, owned and operated by California Polytechnic State University, San Luis Obispo, approximately 18 km northeast of Santa Cruz, California (*fig. 1*). The study portion of the watershed is divided into the 281 ha North Fork Little Creek (NF), the 106 ha South Fork Little Creek (SF), and 191 ha Upper North Fork Little Creek (UNF) sub-basins. Elevations at the study watersheds range from 100 to 580 m. Mean annual precipitation ranged from 875 mm near the outlet of the study watersheds to 1060 mm on the ridgeline during the study period 2002 to 2010. The overstory vegetation is primarily second-growth redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*), with redwood comprising the majority of the vegetation cover. The soils in the study watershed are deep or moderately deep and well drained or somewhat excessively drained. They have a surface layer of loam, sandy loam, or stony sandy loam (Bowman and Estrado 1976).

Data measurement

Three monitoring stations (NF, SF, and UNF) were used to measure streamflow, turbidity, and suspended sediment (*fig. 1*) for the entire study time period. Electronic stage monitoring equipment, instream turbidity probes, and automated pump samplers were deployed at each site. The portion of the North Fork below the Upper North Fork monitoring station was intended as the treatment area, where logging occurred in 2008, while the South Fork and Upper North Fork were intended as the control watersheds.

Streamflow was measured year round; suspended sediment and turbidity were measured during storm events. Relationships between Turbidity and SSC measurements were developed by storm. The total mass or load of sediment for each storm was calculated by multiplying the hourly streamflow volume by the SSC then summing the hourly loads for the storm. Precipitation was measured with tipping bucket gauges at four locations within the Little Creek study watersheds for the study time period (*fig. 1*). Air temperature measurements were from the California Department of Forestry and Fire Protection Ben Lomond climate station (CDEC 2011) approximately 6 km from Little Creek.

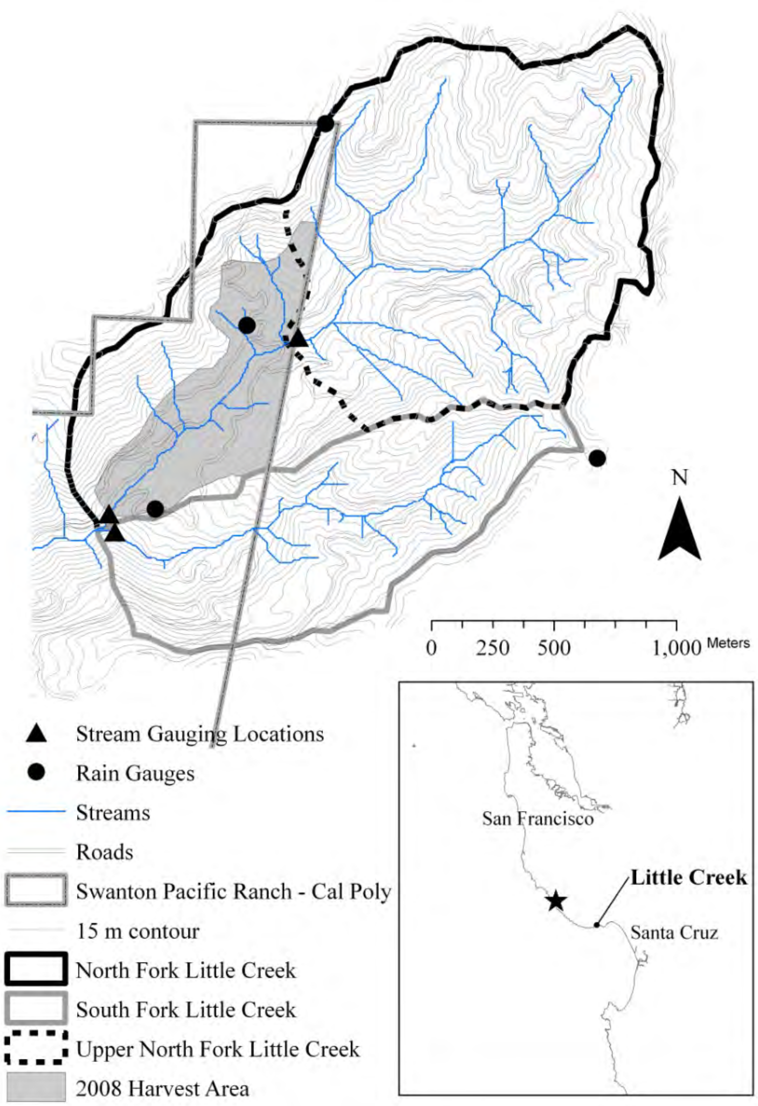


Figure 1—Little Creek study watersheds, California.

Approach to change detection study

A modeling approach will be used to determine if there was change in runoff and storm sediment loads due to disturbances of forest harvest and wildfire. The approach will use the measured runoff and sediment from the pre-treatment time period of 2002 to 2008 water years (WY) for the NF and 2002 to 2009 WY for the SF and UNF to fit hydrologic and erosion models. The differential evolution adaptive metropolis approach (DREAM) (Vrugt et al. 2009) or the Generalized Likelihood Uncertainty Evaluation (GLUE) (Beven and Binley 1992) will be used to address parameter and forcing data (precipitation and meteorological data input to models) uncertainty in the model simulations. The output from each model following the uncertainty analysis will be a range of acceptable answers from the models; this range represents the uncertainty in model use. The model uncertainty in our evaluations will be represented by using the median, 2.5 percentile, and 97.5 percentile model results for each model time step.

Our data analysis approach will be similar to what has been previously used in paired watershed studies. In paired watershed studies statistical models (commonly linear regression) are fit to pre-disturbance measurements for the control and treated watershed. Post-disturbance statistical models between the control and disturbed watershed measurements are compared to pre-disturbance for detection of change. Because our control watershed was disturbed by wildfire we will fit hydrologic and erosion models to the pre-disturbance time period for each station. We will develop regression models between the simulated response (from models) and the measured response. We will use the hydrologic and erosion models fit to the pre-disturbance measurements to model the post-disturbance time period. We then develop post-disturbance regression models between the simulated response, as if harvest or fire had not occurred, with the measured response from harvest and fire disturbance. Prediction intervals will be developed for each of the regression relationships. Change from pre-disturbance will be determined by whether the post- disturbance regression model and model innovations (individual events) are outside the pre-disturbance prediction intervals. This will be replicated for the median, 2.5 percentile, and 97.5 percentile relationships to provide a measure of range of detectability under model uncertainty (Zegre et al. 2010).

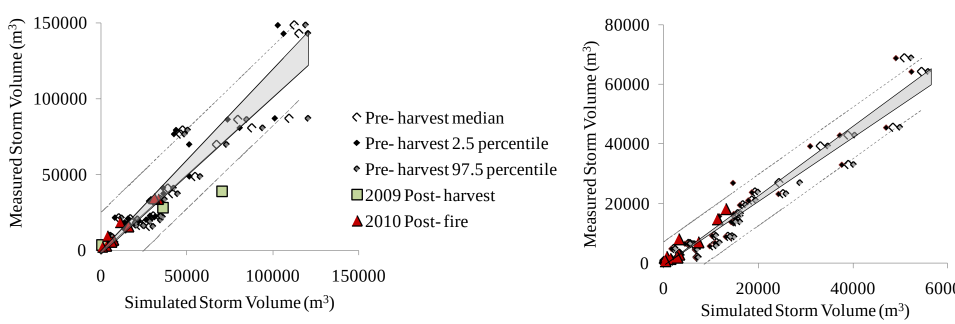
We will use four models for our change detection analysis, two hydrologic models and two watershed erosion models. The hydrologic models to be used are HBV-EC, a conceptual hydrologic model (Canadian Hydraulic Centre 2010), and the Distributed Hydrology Soil Vegetation Model (DHSVM) (Wigmosta et al. 1994) a physically-based distributed model. The watershed erosion models will be the process based model, DHSVM-Sediment (Dooten et al. 2006), and the surface erosion model Water Erosion Prediction Project (WEPP) (Flanagan et al. 1995). Change detection will be analyzed for the storm runoff volume, daily streamflow, monthly streamflow, and storm sediment load. Simple linear regression can be used for evaluating storm runoff volume and sediment load because storm events satisfy the independence requirements of regression analysis. When evaluating daily and monthly streamflow serial auto correlation will be present and will have to be accounted for and used within a generalized linear regression analysis.

Two approaches will be utilized to simulate storm sediment load. The first approach will use the relationship between HBV-EC and DHSVM simulated storm runoff and measured storm sediment load. In the first approach storm sediment load will be calculated by two different methods. The first method simulated storm runoff will be used for linear regression relationships with measured storm sediment load. In the second method simulated storm sediment load calculated from simulated runoff will be used for linear regressions relationships with measured storm sediment load. The second approach to storm sediment load change detection will use two different watershed erosion models to estimate storm sediment load. The second approach to storm sediment load change detection will use two different watershed erosion models to estimate storm sediment load. DHSVM-Sediment and WEPP will be used to estimate storm sediment loads. Simulated storm sediment load calculated from the erosion models will be used for linear regressions relationships with measured storm sediment load. Additionally we will make adjustments to the vegetation, hydrology, and ground cover in DHSVM-Sediment and WEPP to represent both the forest harvest (2009 water year for the NF) and the wildfire (2010). These manipulated models will be run for the post-disturbance time periods for comparison to measured response.

In paired watershed studies the control-treatment watersheds are located as close as possible to eliminate climate variations between the watershed pairs. In our modeling approach we compare pre-and post-disturbance which are independent time frames. The unaccounted variations in climate make possible a rejection or acceptance of the null hypothesis when it should not be, Type I or II errors respectively. To test if climate variation differs in the post-disturbance time period from pre-disturbance total precipitation and mean temperature for storm events for the pre-disturbance (2002 to 2008) and post-disturbance time periods of forest harvest (2008) and wildfire (2009 to 2010) will be tested for difference using a Wilcoxon Rank Sum test.

Example of modeling approach for storm sediment load and runoff using HBV-EC

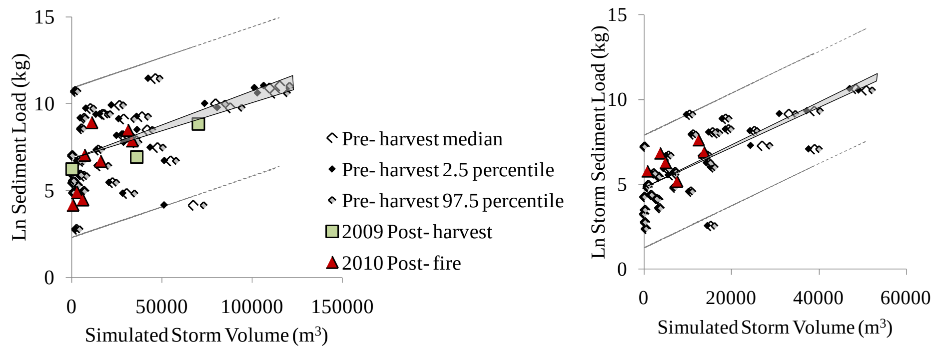
The HBV-EC hydrology model was applied to NF and SF at an hourly time interval to demonstrate change detection of storm runoff and one method of storm sediment load analysis. Comparison of measured and HBV-EC simulated storm runoff volume for the two watersheds in Little Creek, NF and SF indicates that no change is detectable between the pre-harvest relations and the one post-harvest water year for NF (2009) and one post-fire water year for both NF and SF (2010) (*fig. 2*). All innovations of post- disturbance storm runoff volume were found within the 95 percent prediction interval of the pre-harvest regression model, with the exception of one storm from the post- harvest year of the NF, which was lower than pre-harvest values. This was true for the entire range of HBV-EC simulated runoff (median, 2.5 percentile, and 97.5 percentile values).



A) NF B) SF

Figure 2—Storm runoff comparison pre- and post-disturbance for A) North Fork Little Creek, and B) South Fork Little Creek. Grey shaded area represents range of linear regression relationships between simulated 2.5 and 97.5 percentiles and measured storm volume, thick dashed line is regression line post-fire, thin dashed lines are 95 percent prediction intervals for median regression line.

Similarly, comparison of the natural log of measured sediment load and HBV-EC modeled storm runoff volume for the two watersheds in Little Creek, NF and SF, indicates that no change is detectable between the pre-harvest relations and the one post- harvest water year for NF (2009) and one post-fire water year (2010) for both NF and SF (*fig. 3*). All innovations of post-disturbance storm runoff volume were found within the 95 percent prediction interval of the pre-harvest regression model. This was true for the entire range of HBV-EC simulated runoff (median, 2.5 percentile, and 97.5 percentile).



A) NF B) SF

Figure 3—Storm sediment load comparison pre- and post-disturbance for A) North Fork Little Creek, and B) South Fork Little Creek. Grey shaded area represents range of linear regression relationships between simulated 2.5 and 97.5 percentiles and measured storm volume, thick dashed line is regression line post-fire, thin dashed lines are 95 percent prediction intervals for median regression line.

The demonstration of the modeling approach, using the hydrology model HBV-EC, for evaluating harvest and fire hydrologic and sediment effects in Little Creek did not find detectable changes. However, only one year of post-harvest and post-fire events were used. In both these years no large storm events occurred and the number of storm events was low. When you look at the data using only the smaller storm events (< 1 year recurrence), events of similar size for both the pre- and post-disturbance time periods a potential effect from the fire is observed (*fig. 4*). However, we emphasize we do not have evidence this effect occurs for larger events.

Future years of measurement will make the post-disturbance data set more robust and provide more complete results. The analysis does suffer from the results of using one hydrology model. The HBV-EC model is a simple model and did not always quantify the hydrology of Little Creek accurately; see the comparison of pre-harvest measured and modeled storm runoff volumes (*fig. 2*). HBV-EC poorly predicts a few of the storm events creating an artificial variability to the pre-harvest data set. Additional effort at model parameterization would improve the model. Integrating other models into our analysis will allow contrast of the different results and strengths and weaknesses of each model providing improved interpretations of the watershed response.

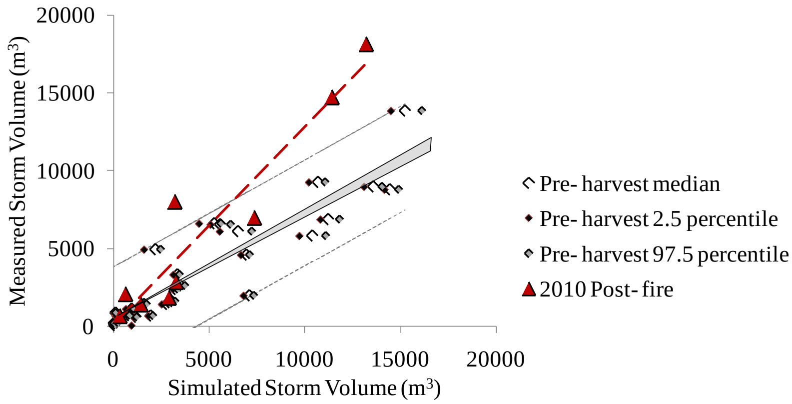


Figure 4—Small storm runoff (< 1 year event) comparison pre- and post-fire for South Fork Little Creek. Grey shaded area represents range of linear regression relationships between simulated 2.5 and 97.5 percentiles and measured storm volume, thick dashed line is regression line post-fire, thin dashed lines are 95 percent prediction intervals for median regression line.

Acknowledgments

This research was supported by a grant from the California State University Agricultural Research Institute. Data from Little Creek was provided by the students of California Polytechnic State University San Luis Obispo, Swanton Pacific Ranch; specifically we thank Drew Loganbill, Drew Perkins, and Michael Gaedeke.

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Delineation of Preventative Landslide Buffers Along Steep Streamside Slopes in Northern California[1]

Jason S. Woodward[2], David W. Lamphear[3], and Matthew R. House[4]

Abstract

Green Diamond Resource Co (GDRCo) applies tree retention buffers to steep slopes along fish bearing (Class I) and non-fish bearing (Class II) streams that are in addition to the standard riparian management zones associated with timber harvest plans. These Steep Streamside Slope (SSS) buffers were designed to reduce the amount of sediment delivering to watercourses as a result of landslides generated by forest management related operations. Initial default buffers were developed through a landslide study during the planning stages of GDRCo's Aquatic Habitat Conservation Plan (AHCP). The continued evaluation of streamside landslides across the property is one of the AHCP's long term research projects, which is aimed at re-defining the SSS "default" prescriptions using a spatially distributed probability based sampling design. Through continued research, our goal is to refine the minimum slope gradients and maximum slope distances associated with the Steep Streamside Slope criteria for each of the 11 Hydrographic Planning Areas (HPA) identified in GDRCo's AHCP.

The HPA to be re-evaluated was the Coastal Klamath HPA, which is a grouping of several watersheds near the mouth of the Klamath River in Humboldt and Del Norte Counties. Sampling consisted of 93 half-mile long stream segments that represent roughly ten percent of the lineal distances of Class I streams and five percent of the lineal distances of Class II streams within this HPA. The sampling method used insured both random selection and spatial distribution of the half mile segments within the study area. The study focused on those landslides that were non road-related, active to historically active, and had observably delivered sediment to a watercourse. The landslide data collected included causal factors, slope characteristics, cross sections, and dimensions of the source and debris, of more than 400 landslides in this region.

Using a topographic ruggedness model on a 50 m LiDAR-based DEM and field-based geologic mapping of the area, we divided the area into distinct morphologic units. The results from the analysis of the landslide data provide new slope gradient and distance thresholds for the SSS default prescriptions. These new SSS prescriptions are unique to each of these morphologic units and provide site specific protections to areas prone to streamside landsliding within the Coastal Klamath HPA. A similar approach will be used for the remaining 10 HPAs.

Key words: shallow landslides, steep streamside slope buffers, LiDAR, topographic ruggedness model, Klamath River

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[1] This paper was presented at the 2011 Coast redwood forests in a changing California: a symposium.

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Introduction

The Steep Streamside Slope Delineation Study is the analysis of streamside landslides across the portions of GDRCo ownership that are bound by the AHCP. The results of this analysis determine the new SSS default protection measures. The primary goal of the initial landslide study, which led to the development of the AHCP SSS "default" prescriptions, was to attempt to reduce the amount of sediment delivering to watercourses as a result of streamside landslides generated by forest management related operations. The AHCP objective of the SSS prescription is to achieve a 70 percent reduction of delivered streamside landslide volumes in comparison to historic management related streamside landslides. The initial study took into account slope distances from the main scarp of observed landslides to the affected watercourse transition line and slope gradients associated with the observed landslides. In addition, the initial study was intentionally biased towards areas that exhibited high concentrations of landslides due to the limited scope and compressed time frame to conduct the study. In order to more accurately define these protection measures and achieve the AHCP objectives directed at streamside landslides, the SSS Delineation Study expands upon the initial landslide evaluation effort. This report presents the findings of the initial phase of the SSS Delineation Study, which was directed at the eighty-thousand acre Coastal Klamath HPA.

Purpose and scope

The purpose of this study is to refine the minimum slope gradients and maximum slope distances associated with the Steep Streamside Slope prescriptions for the Coastal Klamath HPA.

The scope of work for this study is based on the initial "inner-gorge" study that was aimed at delineating an expanded protection zone adjacent to watercourses that would reduce the amount of streamside landslides related to timber harvesting. The study focuses on shallow streamside landslides that were not caused by roads or historic skid trails, were active to historically active, and have observably delivered sediment to a Class 1 or Class 2 watercourse.

Methods

Development of sample reaches

Our sample area involves hillsides adjacent to Class 1, Class 2-2 and Class 2-1 watercourses within the Coastal Klamath HPA. As defined in GDRCo's AHCP Class 1 watercourses are current or historic fish bearing streams and domestic water supplies, Class 2 watercourses contain no fish but support or provide habitat for aquatic vertebrates. Class 2 watercourses are further broken into 1st and 2nd order streams. A Class 2-1 (1st order) watercourse is defined as the first 1,000 ft of a Class 2 stream. Beyond the first 1,000 ft of a Class 2 stream the watercourse becomes a second order Class 2 which we define as a Class 2-2. In addition a Class 2-2 may be formed when two Class 2-1 streams converge. From the point of convergence of the two Class 2-1 watercourses, the watercourse is considered a Class 2-2.

In order to sample random hillside areas within the Coastal Klamath HPA we broke the mapped Class 1 and Class 2 watercourses into half-mile survey reaches throughout the HPA. At the time, the Class 2 watercourses within our hydrography layer had not been broken up into the Class 2-2 and Class 2-1 designations. Therefore we sampled the Class 2 watercourses as a whole and visually checked for a reasonable distribution of Class 2-2 and Class 2-1 watercourses. Once our LiDAR derived hydrography was developed we assessed the actual distribution of the Class 2-2 and Class 2-1 watercourses, which is discussed later under Results.

At the time of development, there were 191 miles of mapped Class 1 and 619 miles of mapped Class 2 watercourses within the Coastal Klamath HPA. Our goal was to survey roughly ten percent of the lineal distance of Class 1 streams (19 miles) and five percent of the lineal distance of Class 2 streams (31 miles) within this HPA. The geographically distributed systematic random sampling method used, insured both random selection and spatial distribution of the half-mile segments within the study area. This method involved delineating whole streams, from the confluence to the upstream end of a Class 2, breaking these streams into approximate half mile sample reaches, sorting them tabularly according to the confluence coordinates and position within the individual stream, and then systematically selecting reaches to gain the appropriate sample percentage. In addition, we stratified the Class 1 watercourses to ensure an even distribution of sample reaches from the lower, middle, and upper portions of these streams. Due to their abundance this was not necessary for the Class 2 watercourses. In all, there were 93 half-mile long stream reaches along Class 1 and Class 2 watercourses. The sample reaches are shown on *fig. 1*.

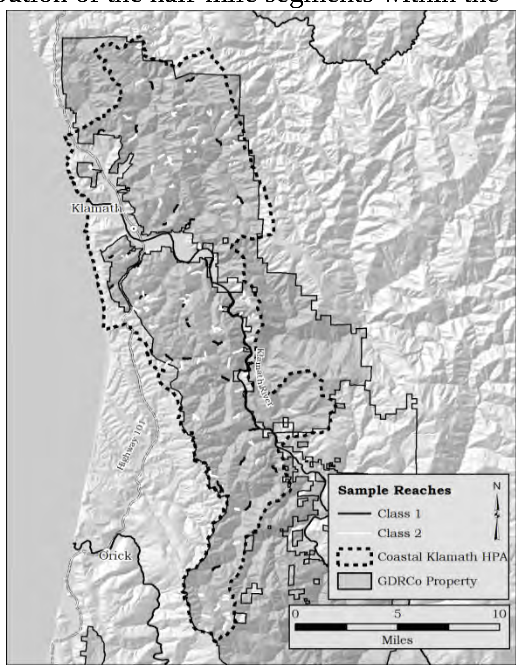

Figure 1—Coastal Klamath HPA sample reaches.

Review of existing documents

Our review of existing documents included a review of published geologic mapping in the area and a comprehensive landslide inventory using aerial photographs from the 1954, 1958, 1969, 1975, 1984, 1997, 2001, and 2004 sets and was focused on the areas within the vicinity of our sample reaches. As is common practice we mapped only landslides that were visible at the scale of the photographs which is 1:12,000 for each of the photo sets. We found that this typically equates to a landslide roughly 50 ft wide by 100 ft long or 5,000 square ft. These landslides were transferred into our GIS landslide layer with associated tabular data that included; photo year and label, land use and 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).

Review of the aerial photographs indicates that the areas west of the Klamath River from Saugep Creek south to Tectah Creek were clearcut beginning in the 1950s and slowed down by the late 1970s. The primary harvest method was ground-based tractor-yarding likely due to the more subdued topography in this region. There are however several areas in this region that are much steeper and were subject to cable yarding in the late 1960s and 1970s such as portions of Ah Pah Creek and the lower portion of Tectah Creek. Along the Klamath River we observed early harvesting in the 1950s and 1960s that consisted of ground-based tractor clear cutting. East of the Klamath River from Mynot Creek south to Bear Creek, timber harvesting began in the mid to late 1960s and continued through the 1980s and slowing down in the 1990s. Mainly clearcut with few selections were observed in these areas with ground-based tractor yarding as the main harvest methods. Cable-yarding however was the primary yarding method being utilized in the steeper terrain within the Hunter and Terwer Creek drainages.

Throughout the Costal Klamath HPA, lay-out construction was also common practice. Lay-outs were mounds of earth constructed in the direction of felling that softened the landing of old-growth timber and helped reduce breakage from the landing. This resulted in a significant amount of ground disturbance throughout the area. Road construction also resulted in significant ground disturbance throughout the area. Until the early 1980s side-cast construction was the common practice. This was a practice where earth materials from road construction were simply pushed off the downhill side of the road rather than being hauled to a stable location. Due to the dense skid road network involved with this initial entry and poor road-building practices, such as side-cast construction, road-related failures were very common throughout the region.

Published mapping in the area includes mapping from Aalto and Harper 1982, Aalto and Kelsey 1981, Dell'Osso et al. 2002, and Ristau 1979a, 1979b. This mapping indicates the majority of the Coastal Klamath is underlain by the broken formation of the Franciscan Complex. Aalto and Harper describe the broken formation as tectonically fragmented interbedded graywacke, shale, and conglomerate. In general the topography in the area becomes more incised and rugged to the east where bedrock is less fragmented. In the western portion of the area topography is more subdued, bedrock is more fragmented, and ridges are frequently capped with alluvial deposits.

Field work, measurements, and calculations

Field work involved surveying the hillsides adjacent to the sample reaches for shallow streamside landslides. Our study focused on those landslides that were non road and non skid-trail related, active to historically active, and had observably delivered sediment to a watercourse. Only landslides greater than 10 ft by 20 ft were included in the survey. The 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 watercourse transition line, and the average slope gradient of the hillside.

Cross sections show the main scarp, projected failure surface, the estimated original surface, extent of slide debris and the associated watercourse. Cross sections were utilized to determine the length of the rupture area, length of debris, estimated thickness of the failure, estimated thickness of the remaining slide debris, and an estimated slope gradient of the hillside.

As mentioned above landslides that were thought to have been caused by roads or skid trails were not included in the analysis. Determining whether or not a slide has been caused by a road is a difficult task; especially if the failure is not a recent one. There are inferences that must be made in attributing a causal mechanism such as roads to the failure of a landslide. As a result we attempted to attribute road caused only to failures that appeared to have a reasonable or obvious negative association with a road or skid trail. Our determination of road caused failures took into account several factors. Fill slope related failures had to truncate a portion of the running surface of the road in order to be considered. In general it is difficult to determine if a failure emanating from the fill slope was caused by the road as most of these failures are older. Therefore we focused on slides that had observably offset at least a portion of the road. Cut bank related failures are typically easier to determine as most slides will take out a cut slope and fail onto or over the road prism or may take out a portion of the road as well. These types of cut bank failures were considered road caused. Skid trails, although no longer being constructed on GDRCo timberlands, were commonly cut across steep terrain adjacent to watercourses and regularly used side-cast construction. In such areas skid-trail related slides were common and fairly easy to distinguish and were considered road-caused. Patterns of failure can typically be captured in aerial photographs. For instance, we observed that road related and skid trail failures were common in the early harvest entries where side-cast construction was the common practice and in areas with dense skid road networks or tall cut banks. Thusly, aerial photographs were also used to assist in the determination of road caused failures.

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 treats the slide rupture area and debris as a half of an ellipse and was obtained from published work by Cruden and Varnes 1996.

Development of morphologic units

The initial default SSS prescriptions were applied uniformly across the Coastal Klamath HPA regardless of morphology or bedrock type, both of which can vary dramatically across the region. Looking at the hillshade model from a 1 meter LiDAR DEM we determined that the morphologic complexity of the region could be separated into at least three discreet units. 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. In order to help define these areas we chose to use a topographic ruggedness model developed by Riley et al. 1999, using a 50 m resampled LiDAR DEM. The topographic ruggedness model computes the Terrain Ruggedness Index (TRI); the average change in elevation (meters) within a three by three kernel (150 m by 150 m area) at each 50 m cell within the region. The resulting TRI's were classified into three groups (0 to 38 m, 38 to 63 m, and 63 to 170 m) using Jenks natural breaks (Jenks 1967). Jenks natural breaks, developed by cartographer George Jenks, are a data classification method that minimizes variance within groups of data and maximizes variance between groups of data. We characterized the resulting three TRI values as subdued, moderately subdued to slightly rugged, and rugged terrain, noting that this characterization is specific only to this HPA and portions of its surrounding areas. Next, we manually divided the HPA into large morphologic blocks based on our observations of the morphology observed in the LiDAR DEM, our observations in the variation of the local bedrock, and the TRI's. These larger blocks have been termed "Steep Streamside Slope Morphologic Units" (SSSMU).

Once the SSSMU's had been delineated we overlaid them with the landslide data. At that time we noticed that we did not have enough landslide data in each of the SSSMU's to define the SSS buffers and slope thresholds. This is not to say that we did not have enough sample area, we simply found very few landslides in certain areas. In one of the SSSMU's there were only 14 landslides to characterize the Class 2-1 streams and only 28 landslides to characterize the Class 2-2 streams. As a result SSSMU 1 and SSSMU 2 were grouped together. For the final grouping SSSMU 1 and SSSMU 2 were combined and renamed SSSMU 1 with TRI values of 0 to 63 and SSSMU 3 was renamed SSSMU 2 with TRI values of 63 to 170. The final SSSMU's are shown on *figure 2*.

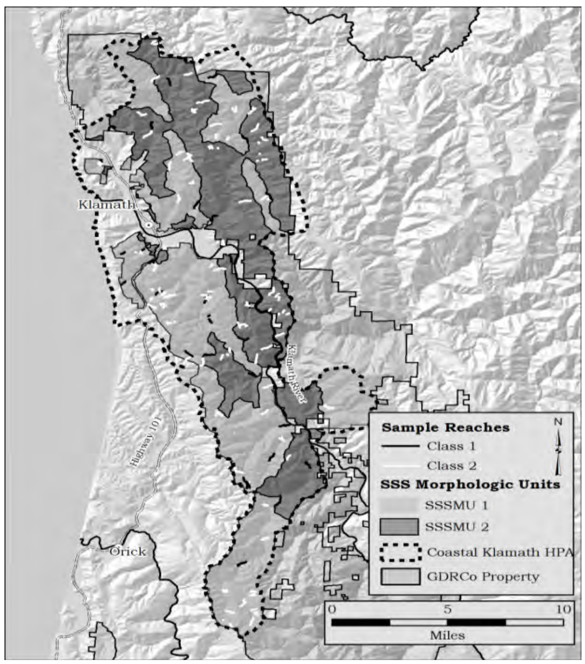

Figure 2— Coastal Klamath HPA.

Cumulative volume

This study included 423 landslides to characterize landslide patterns and develop the SSS buffers for the Coastal Klamath HPA alone. We found that of the 423 landslides, 42 (or 10 percent) of the slides fall in the cumulative volume distribution above the 60 percent threshold. Of those 42 landslides, 33 percent were observed on 1978 photos or earlier, 74 percent of them were observed on the 1997 photos or earlier and 12 percent were not observed in the photo record. Each of the slides observed on the 1997 or earlier photos occurred in areas that were clearcut prior to the retention of riparian buffers of any kind. Although primarily intended for riparian habitat, these buffers also help reduce the potential for streamside landslides. The slides observed on the 1997 and earlier photographs failed under management conditions that have not existed since the mid-1990s and certainly not under current management practices (i.e., those within the last 15 years that have been afforded various protection measures including riparian and geologic). Under current management practices these areas would have been, all or in part, protected by various retention buffers and likely received some level of geologic oversight at the THP level. These data suggest that these larger landslides are rare and may be attributed to historic logging practices. Based on these observations it is our judgment that using the 60 percent cumulative volume threshold will be adequate to derive the SSS distance buffers and be able to achieve the SSS objective of a 70 percent reduction in sediment delivery from management related streamside landslides in the Coast Klamath HPA. It should also be noted that the effectiveness of these new buffers will be evaluated during the AHCP's SSS Assessment project which will take place over the next 11 years.

With the exception of the Coastal Klamath HPA group and Smith River HPA, a cumulative sediment delivery volume of 80 percent was used to determine initial default SSS gradients. In these areas the data from the initial SSS study suggested slope gradient thresholds of 85 percent and 70 percent, respectively. It was assumed that the initial data were biased towards steeper slopes that would not accurately characterize these areas as a whole. The actual slope gradients assigned to the initial SSS default buffers for these areas were reduced to 70 percent and 65 percent respectively. Like what was done for the maximum distance threshold, this was a conservative approach based on limited data to ensure that the SSS buffers would be able to meet the overall sediment objectives from the streamside slopes until a more robust analysis was conducted. Based on the observations described above, it is our judgment that using a cumulative volume threshold of 80 percent will be adequate to derive the SSS minimum gradient thresholds and be able to achieve the SSS effectiveness objectives.

Results

Our sample reaches were mapped in the field and end points marked using a Garmin 60Cx GPS. After transferring these endpoints into the GIS and onto the LiDAR corrected hydrography layer we found that the spatially upgraded sample came very close to our target. Our surveys captured 13 percent of the mapped lineal distance of Class 1 watercourses (target of 10 percent) and 5 percent of the mapped lineal distance of the Class 2 watercourses (target of 5 percent) with a fairly even distribution between the Class 2-1 and Class 2-2 watercourses capturing 4 percent of the Class 2-1 and 6 percent of the Class 2-2 mapped watercourses.

As noted above we used a cumulative volume of 60 percent to determine the new SSS slope distances. There were a total of 423 landslides included in our analysis of the Coastal Klamath HPA. The distribution of landslides by watercourse and SSSMU are shown in *table 1*.

The majority of these landslides occurred along the Class 1 and Class 2-2 watercourses. Few landslides were found along the Class 2-1 watercourses, in fact there were 70 slides observed along the selected sample reaches with a near even split of 34 slides in SSSMU 1 and 36 slides in SSSMU 2. After reviewing the Class 2-1 landslide data within the individual SSSMU's we noticed a high variability within the measured SSS criteria. It became apparent that this would be an insufficient amount of data to delineate buffers for each of the SSSMU's. Therefore

Table 1—*Distribution of landslides by watercourse and SSSMU.*

Watercourse Class	SSSMU	Number of Slides
I	2	67
I	1	68
2-2	2	133
2-2	1	85
2-1	1&2	70

we combined the data for Class 2-1 watercourses and came up with prescriptions for the HPA as a whole for this watercourse class. The remaining landslides provided a sufficient distribution to characterize the SSS buffers in both of the SSSMU's.

Review of the Class 2-2 and 2-1 watercourses and their adjacent slope gradients using the LiDAR indicates that average slopes adjacent to Class 2-1 and Class 2-2 watercourses are essentially the same. However, when we looked at the average maximum slope gradient adjacent to these watercourses we found that Class 2-2 watercourses are located in proximity to much steeper slopes, by up to 22 degrees. This suggests that there may be fewer steeper slopes adjacent to Class 2-1 watercourses and may explain why there were fewer landslides observed in these areas. However at this point this is simply a correlation and requires more investigation which we may be able to address at a later date.

The revised SSS slope distances have been calculated. The distribution of cumulative volume of delivered sediment versus landslide distances from crown to watercourse is shown on *figure 3* and a summary table of these results is shown in *table 2*.

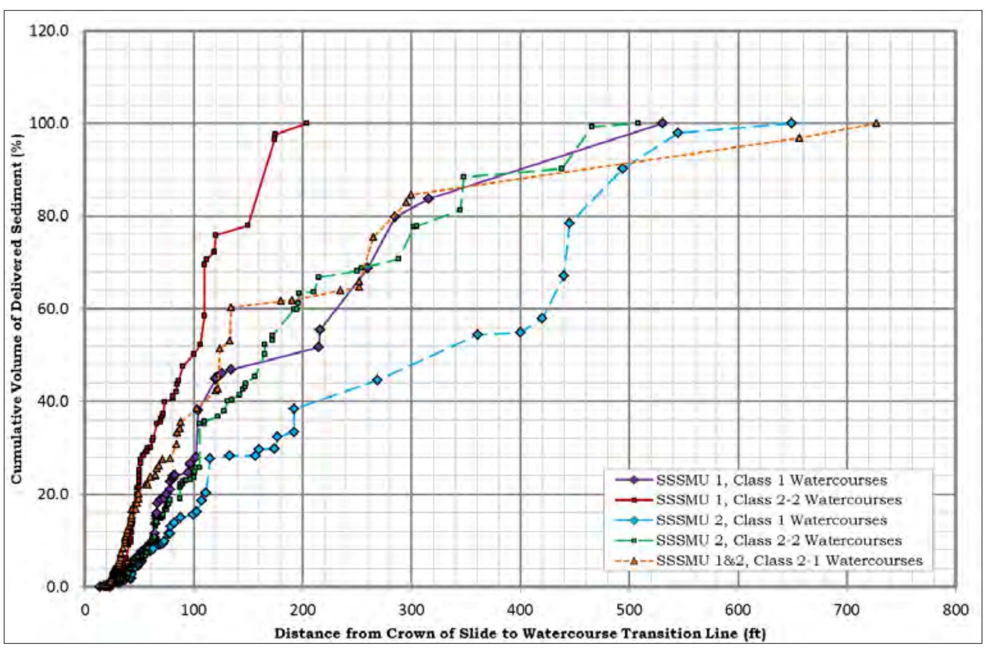

Figure 3—Cumulative volume of delivered sediment versus landslide distances from crown of slide to watercourse transition line for Class I, Class 2-1 and Class 2-2 watercourses.

Table 2—*New SSS thresholds & a comparison of the revised Coastal Klamath SSS buffers with initial default buffers.*

Coastal Klamath SSS Maximum Slope Distances (feet) & Minimum Slope Gradient (%) Thresholds			
SSSMU	**Class I**	**Class 2-2**	**Class 2-1**
1	240' & 65%	110' & 70%	135' & 75%
2	425' & 75%	195' & 85%	
Initial Default Buffers	475' & 70%	200' & 70%	100' & 70%

Slope Gradient thresholds were based on a cumulative volume of delivered sediment of 80 percent. With a significantly larger data set than the initial study, it was our judgment that we had sufficient data to characterize the SSS gradient thresholds by watercourse type as well as SSSMU. As was the case for the SSS slope distances we combined the data for the Class 2-1 watercourses to come up with an HPA wide slope threshold for this watercourse class. The distribution of cumulative volume of sediment delivered versus landslide slope gradients are shown on *figure 4* and a summary table of the slope gradient thresholds is shown in *table 2*.

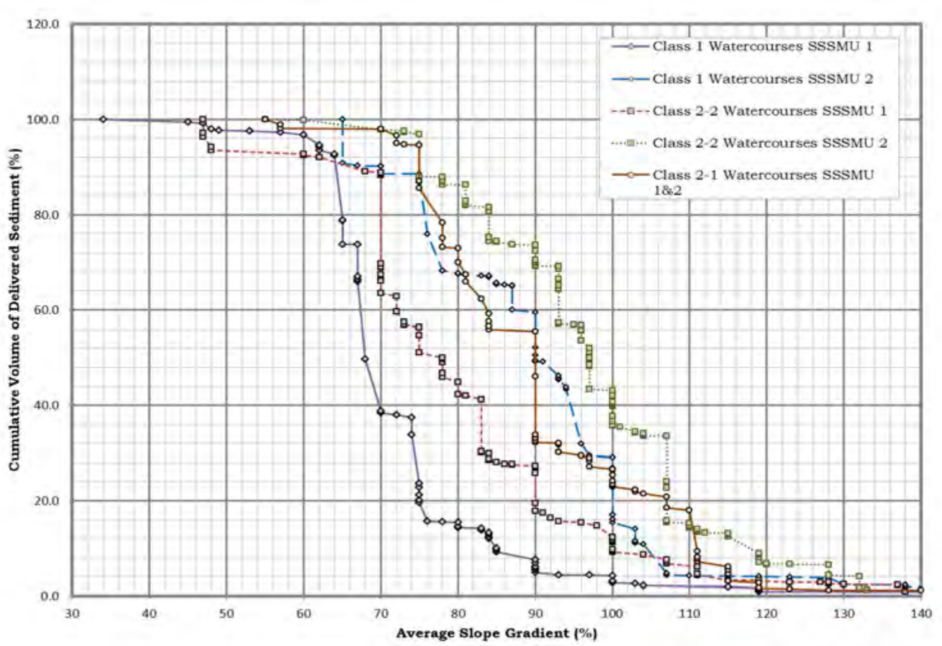

Figure 4—Cumulative volume of sediment delivered versus average slope gradient from SSSMU 1& 2 for Class I, Class 2-1 and Class 2-2 watercourses.

Discussion

Looking at the different slope distances and slope gradient thresholds we can see justification for the rationale behind the development of the SSSMUs. *Table 2* also outlines a comparison of the Initial Default SSS distance and slope thresholds as well as the new slope distances and associated slope thresholds.

Both slope distances and slope gradient thresholds vary significantly between SSSMUs. Within SSSMU 2 we can see the effects of the more competent sandstone and shale of the broken formation of the Franciscan complex. In these areas, we observed landslides occurring on steeper slopes with longer slide run out distances. In contrast within SSSMU 1 we observed more fragmented earth materials of the broken formation of the Franciscan complex as well as uplifted alluvial sediments, which resulted in slacker slopes and shorter run out distances with the associated landslides. In the future, our observations from this study coupled with more detailed mapping of the local bedrock may lead to differentiation of multiple bedrock units within the broken formation of the Franciscan complex. In the meantime, these varying buffer criteria offer more site specific protection measures where they are warranted.

Conclusions

Our analysis of landslides in the Coastal Klamath HPA has resulted in changes to both the slope distance and slope gradient criteria associated with the default SSS prescriptions. These new criteria offer specific protection to morphologically discrete areas within the Coastal Klamath HPA which we termed SSSMUs. Our sampling methods reduced bias and spatially distributed the samples about the HPA 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 buffers was 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. It is our judgment that these new default buffers will allow GDRCo to meet this goal set forth in the AHCP.

Acknowledgments

We would like to thank our field crew; Scott Matheson, Scott Kirkman, Evan Saint-Pierre, Nick Graehl, Daniel Hadley, William Troxler, and Esther Stokes for producing quality field data through inclement weather and the steep and challenging terrain of the interior Klamath River region.

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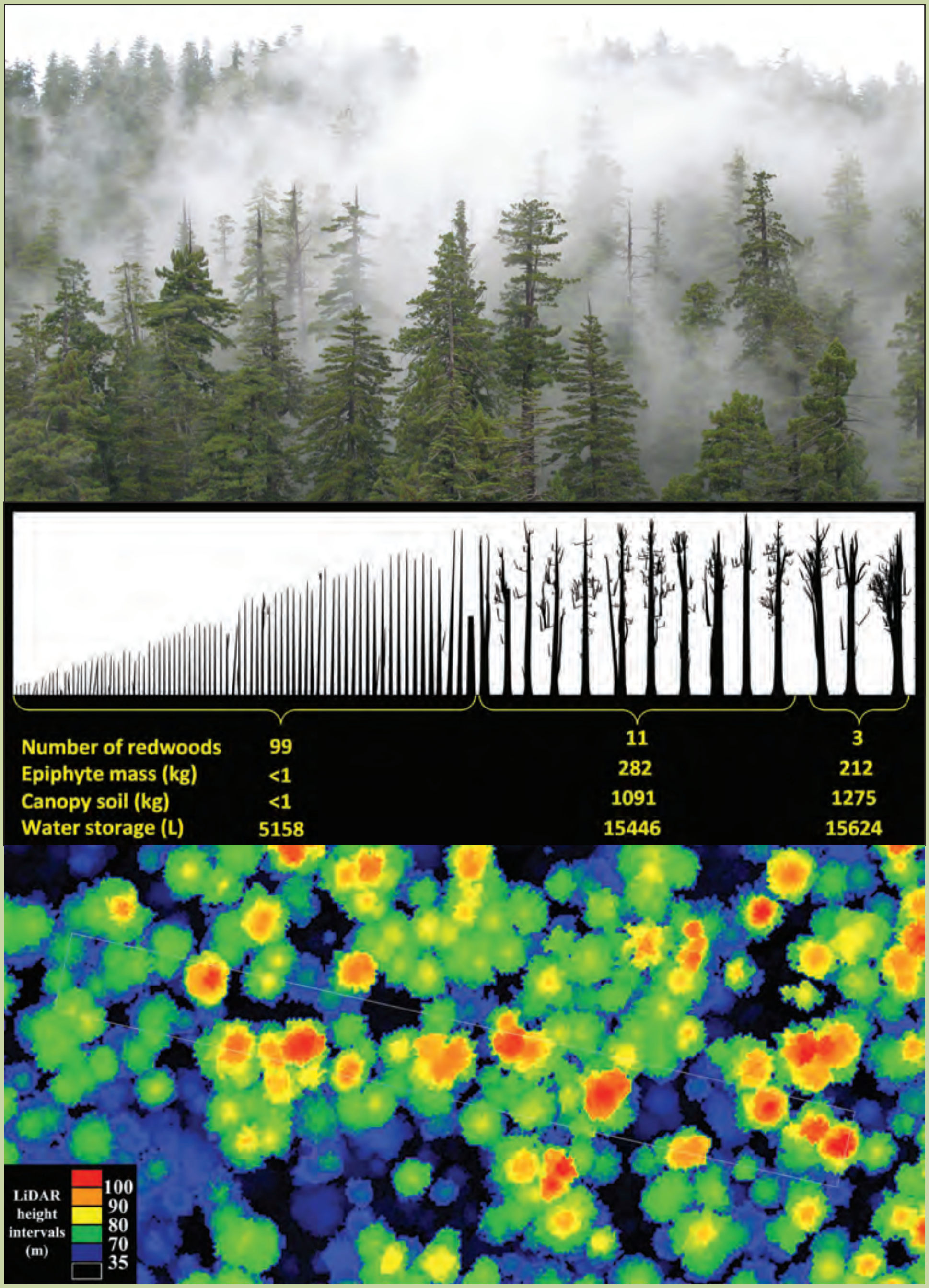
Ecology



Number of redwoods 99 11 3
Epiphyte mass (kg) <1 282 212
Canopy soil (kg) <1 1091 1275
Water storage (L) 5158 15446 15624
LiDAR height intervals (m)
100
90
80
70
35

Responses of Redwood Soil Microbial Community Structure and N Transformations to Climate Change

Damon C. Bradbury[1] and Mary K. Firestone[1]

Abstract
Soil microorganisms perform critical ecosystem functions, including decomposition, nitrogen (N) mineralization and nitrification. Soil temperature and water availability can be critical determinants of the rates of these processes as well as microbial community composition and structure. This research examined how changes in climate affect bacterial and fungal community structures and rates of N mineralization and nitrification in coast redwood forest soils. Soils were reciprocally transplanted between three redwood sites located across a latitudinal climate gradient, from near the southern extent of redwoods to near their northern extent and collected one year later at the end of the summer. A molecular community fingerprinting technique was used to examine changes in fungal and bacterial community structures, and ^{15}N-isotope pool dilution was used to measure gross rates of N mineralization and nitrification. After one year, soil fungal and bacterial community structures in transplanted soils had changed to become more similar, but not identical, to those native to their new destination sites. Both climatic and edaphic variables were correlated with the variability in microbial community structure. While there were few significant differences in gross N mineralization rates between soil-climate combinations, gross nitrification rates were influenced by a change in climate. Rates of gross nitrification were highest in soils when located in the wetter, most northern site. While rates of gross nitrification varied widely in soils with water potentials above -0.05 MPa, rates were low in soils below -0.05 MPa. Changes in redwood climate, fog frequency and summer water availability will likely alter soil microbial community structure and rates of gross nitrification. Greater magnitude changes in climate or more than one year of exposure may be necessary to cause alterations in rates of gross N mineralization.

Key words: bacteria, climate change, fungi, microbial community structure, nitrogen, redwood

Introduction

Global climate models predict alterations in regional patterns of precipitation and temperature (IPCC 2007). An understanding of how soil microbes respond to alterations in climate is important because these organisms determine many aspects of ecosystem structure, function and services. While patterns have been proposed for the manner in which macrobiota can respond to shifts in climate (e.g., shifts in species range with latitude or altitude; Colwell et al. 2008, Parmesan and Yohe 2003), we have limited knowledge of microbial responses to climate change.

Due to the influence of temperature and moisture on microbial activity, changes in gross nitrogen (N) mineralization and nitrification are likely to occur in response to climate change. Soil moisture, in particular, has been suggested to be one of the main

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factors that will influence rates of soil N mineralization and nitrification in response to changes in climate, especially in seasonally dry ecosystems (Gomez-Rey et al. 2010, Jamieson et al. 1999). It is possible that N mineralization may not be as readily affected by changes in moisture as gross nitrification because N mineralization reactions are carried out by a wide variety of prokaryotic organisms and fungi while nitrification is carried out by only a limited number of archaeal and bacterial groups (Gleeson et al. 2010, Paul and Clark 1996).

This study examined the impact of climate change on rates of gross N mineralization, gross nitrification and soil fungal and bacterial community structures in coast redwood forests. Soils were transplanted across the latitudinal range of coast redwood forests in northern California to examine how interactions among native soil characteristics and changes in climatic exposure affect gross rates of N mineralization and nitrification and bacterial and fungal community structures. Soils were sampled at the end of the dry, Mediterranean-climate summer, a time of year when the presence of fog is a defining characteristic of coast redwood forests (Azevedo and Morgan 1974) and differences in fog frequency result in the most dramatic differences in temperature and soil water availability north to south. The results elucidate the potential of climate change to alter rates of these communities and important N-cycling processes, while also considering how soil characteristics can modulate the impacts of climate change.

Methods

Study sites

The three study sites were located in old-growth coast redwood forests in Prairie Creek State Park, CA (Humboldt County), the Grove of the Old Trees, Occidental, CA (Sonoma County) and Big Basin State Park, CA (Santa Cruz County), referred to as the North, Middle and South sites, respectively. The mean annual average precipitation increases with latitude and temperature decreases with latitude (*table 1*). The aboveground biomass is much greater in the North site and the rates of primary productivity and decomposition are higher there as well. Soil characteristics (e.g., soil C and N, pH, percent silt and sand, and water retention curves) also differ between the sites (*table 1*).

Experimental design: three-way reciprocal transplant

In August 2004, two types of intact soil cores (either 10 cm diameter x 15 cm deep, solid schedule 40 PVC cylinders or flexible 2 mm mesh shade cloth sown into cylinders of the same dimensions with nylon thread) were transplanted between the three sites in a full factorial design (soil from each site transplanted into every site). Five (3 m x 3 m) plots located from 30 to 400 m apart were established at each site. Plot pairs between sites were chosen randomly, and cores were randomly located within a plot. Soil cores that were returned to their plots of origin served as controls. Ninety transplanted cores were harvested after one year in early September 2005 (45 mesh and 45 PVC; five of each from three sites of origin x three transplant sites), as well as 15 previously undisturbed cores (five cores x three sites).

Table 1—*Site characteristics. See text for weather station locations for mean annual rainfall and temperature data (1970 to2000). All soil variables were measured for 0 to 15 cm, except percent C and percent N were measured for 0 to 10 cm and soil temperature was measured at 7.5 cm.*

Location	**North** **Prairie Creek S.P., Orick, CA**	**Middle** **Grove of the Old Trees, Occidental, CA**	**South** **Big Basin S.P., Boulder Creek, CA**
Latitude	41° 41' N	38° 24' N	37° 10' N
Distance to coast, elevation	6.8 km, 54 m.a.s.l.	6.5 km, 148 m.a.s.l.	6.4 km, 128 m.a.s.l.
Mean annual rainfall	1,677 mm	1,419 mm	1,211 mm
Air temperature, mean (range)	11 °C (2 °C – 21 °C)	14 °C (4 °C – 29 °C)	15 °C (2 °C – 30 °C)
Soil texture	Sandy Loam	Sandy Loam	Sandy Loam
Soil pH (mean +/- s.e., n=5)	4.9 +/- 0.1	5.3 +/- 0.2	5.7 +/- 0.2
Soil %C (mean +/- s.e., n=5)	10.7 +/- 0.9	4.6 +/- 0.4	4.5 +/- 0.5
Soil %N (mean +/- s.e., n=5)	0.58 +/- 0.04	0.27 +/- 0.07	0.25 +/- 0.03

Microbial community analyses

Bulk soil DNA was extracted from 500 mg of soil using the Bio101 Fast DNA Spin Kit for Soils (Q Biogene, Carlsbad, CA) according to the manufacturer's instructions. The DNA extracts were then diluted (7.5X or 10X) to approximately 40 ng / µl. The PCR primers (from Sigma-Genosys, The Woodlands, TX) used for bacteria were 27F and 1492R (Giovanni 1991), and the ITS1F and ITS4 primers, targeting the internal transcribed spacer (ITS) region of the rDNA gene, were used for fungi (Gardes and Bruns 1993). The forward primers were labeled with 6-FAM for PCR for terminal restriction fragment length polymorphism (T-RFLP; Liu et al. 1997). Three PCR reactions of each sample were bulked together before PCR clean up and digestion. All of the PCR products were purified using the MoBio UltraPure Clean Up Kit and eluted in 50 µl of elution buffer. All 10 replicates of transplanted samples and controls and five replicates of fresh undisturbed samples were analyzed by T-RFLP with bacterial primers, and four replicates (two mesh and two PVC cores for each origin-transplant combination) were analyzed for fungal T-RFLP. After PCR clean up, approximately 400 ng of bacterial 16S rDNA PCR product was digested with *MspI*. For fungi, 100 ng of fungal ITS PCR product was digested with *HhaI*. The tubes were incubated at 37 °C for 18 hours and then ethanol precipitated and washed twice before resuspension in 10 µl of fresh formamide. The abundance of different terminal restriction fragments (TRFs) was measured using an ABI 3100 capillary sequencer (Applied Biosystems, Foster City, CA) run in the GeneScan mode. Each TRF was converted into its relative abundance (percent abundance) by dividing each individual TRF peak height by the total sum of peak heights for all of the TRFs found in a sample and multiplying by 100.

Rates of gross N mineralization and nitrification

Gross rates of N mineralization and nitrification were measured by ^{15}N-isotope pool dilution (Herman et al. 1995). Soils were labeled with ^{15}N as either $(^{15}NH_4)_2SO_4$

for gross N mineralization or $K^{15}NO_3$ for gross nitrification; 1 ml of labeled solution was added to 50 g of soil to increase the appropriate pool to approximately 50 or 25 atom percent ^{15}N, respectively. Initial samples were taken after approximately 2 hours and final samples 16 hours later. Concentrations of nitrate (NO_3) and ammonium (NH_4) were determined after a one-hour extraction in 1 M KCl. Atom percent^{15}N was determined using an automated N and C analyzer coupled to an isotope-ratio mass spectrometer (ANCA-IRMS; PDZ Europa Limited, Crewe, UK) after diffusing out the NH_4 or NO_3 onto filter disks (Herman et. al. 1995). Gross rates of N mineralization and nitrification were calculated according to Kirkham and Bartholomew (1954). Soils were stored at 12 °C rates were measured at 12 °C to determine rates that were reasonably representative of field conditions (14 to 16 °C).

Climate and soil characteristics

Monthly precipitation data was obtained from NOAA National Climatic Data Center records for weather stations near each site; the stations near the North and Middle sites, respectively, were Orick Prairie Creek Park (Station 046498, 41° 22′ N and 124° 01′ W) and Occidental (Station 046370, 38° 23′ N and 122° 58′ W), and for the South, the average of Ben Lomond #4 and Felton (Station 040673, 37° 05′ N and 122° 05′ W; Station 043004, 37° 03′ N and 122° 05′ W) was used. Soil temperatures were recorded at the time of sampling and in February, May and November as well as to determine the annual ranges in soil temperature. Gravimetric water content (GWC) was determined for soils collected at the same times. Determinations of bulk density, soil particle size distributions (sand, silt, clay; Sheldrick and Wang 1993), water retention curves (Klute 1986), pH (1:1 soil:0.01M $CaCl_2$), and concentrations of soluble organic C and N (extracted in 0.05M K_2SO_4) and microbial biomass C and N (Brookes et al. 1985, Cabrera and Beare 1993) were made using standard methods. Bulk density and water retention curves were used to transform GWC into volumetric water content (VWC) and soil water potential, respectively.

Statistical analyses

To examine changes in community structure, non-metric multidimensional scaling (NMDS) was used to visualize the (dis)similarity in community structure between samples (McCune and Grace 2002). In NMDS ordinations, the farther apart samples are in space, the more they differ in structure. Permutational multivariate analysis of variance (PERMANOVA) was performed to determine if the differences in structure between soil-climate groups were significant (Anderson 2001, 2005). Pair-wise differences between groups were determined a posteriori. The Bray Curtis distance was used, and the tests were done using the distance ranks. The results from the NMDS ordinations and PERMANOVA tests were compared to hypothesized general response scenarios (*fig. 1*) to determine if the transplanted redwood microbial communities had exhibited a strong response, intermediate response or no response to the change in climate. Transplanted samples were always compared to the control samples of both the site of origin and the new transplant destination.

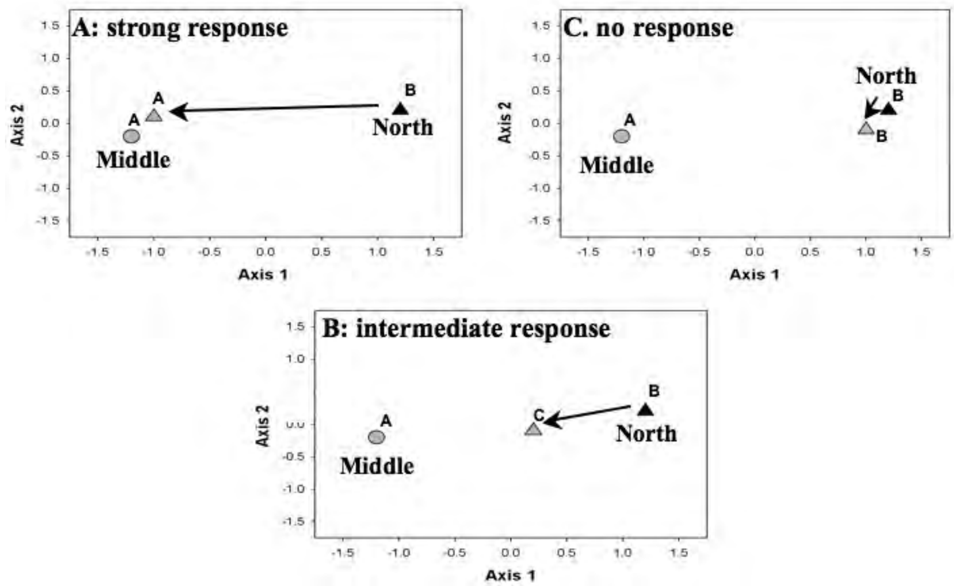

Figure 1—Hypothetical responses of soil microbial communities to climate change shown in theoretical NMDS ordinations. The distance between samples is indicative of the magnitude of their dissimilarity in community composition. Different letters indicate significant differences, as determined by PERMANOVA. The theoretical communities from the North and Middle controls are labeled and that of the North soil transplanted into the Middle climate is shown as a gray triangle.

The variability in fungal and bacterial community structure was correlated with environmental variables by performing Mantel tests. The Bray-Curtis distance was used for the taxa abundance matrix and the Euclidean distance for the environmental variable matrix. The standardized Mantel statistic (r) is equivalent to the Pearson's correlation (r) statistic. Mantel tests were performed to examine correlations between fungal and bacterial community structures and several climatic and edaphic variables, including: gravimetric water content, volumetric water content, water-filled pore space, water potential, total annual rainfall, spring rainfall (MJ), summer rainfall (JA), temperature, maximum mean monthly temperature, soluble organic carbon, total soluble nitrogen, microbial biomass carbon, microbial biomass nitrogen, pH, percent sand, percent silt and percent clay and pH.

Two-way analysis of variance was performed to determine if there were differences in gross rates of N mineralization and nitrification between sites of origin, transplant sites, or the interaction between the two (using R version 2.10.0). Because the results for the two types of cores (PVC and mesh) did not differ significantly, they were analyzed together. The N mineralization and nitrification results are presented as the averages by either the soil origins or the sites of transplantation/incubation because the interaction terms were not significant for either process rate.

Results

Effect of transplant on soil microbial community structure

In general, after 1 year, the fungal and bacterial community structures of the transplanted samples shifted to more closely resemble those of the control soils native to the new transplant site (climate), but they did not become indistinguishable from them. All fungal community responses were classified as "intermediate" by the NMDS ordinations and PERMANOVA results (*table 2*). For the bacteria, five of the six transplanted soils changed in structure but still differed from the control samples of the new site (*table 2).* One transplant scenario did not cause a bacterial community response, North soil into the South climate.

Table 2—*Summary of changes in fungal (F) bacterial (B) and community structures in response to transplanting. Responses were determined from NMDS ordinations and PERMANOVA results, as shown in fig. 1.*

Transplant	Strong response	Intermediate response	No response
North into South	-	F	B
North into Middle	-	F , B	-
Middle into North	-	F , B	-
Middle into South	-	F , B	-
South into North	-	F , B	-
South into Middle	-	F , B	-

Correlations between community structures and environmental variables

Fungal community structure was significantly correlated with temperature, precipitation, pH and soil texture values. The strongest correlations were found with the maximum mean daily temperature ($r = 0.62$), the mean annual temperature ($r = 0.56$), the amount of summer rainfall ($r = 0.56$) and the total annual rainfall of the previous year ($r = 0.37$*; table 3*). When examining the correlations between bacterial community structure and environmental variables, measures of precipitation and soil water availability had weaker correlations ($r = 0.15$ to 0.20). Measures of soil texture and soil pH had correlations between 0.28 and 0.39, and total dissolved nitrogen ($r = 0.18$), microbial biomass nitrogen ($r = 0.20$) and microbial biomass carbon ($r = 0.23$) all had lower but significant correlations with bacterial T-RFLP community structure.

Table 3—*Significant standardized Mantel correlations (Mantel r) between fungal and bacterial community structures and environmental variables. Only significant correlations are shown, p < 0.01 for all.*

Environmental variable	Fungi Mantel r	Bacteria Mantel r
Max. temperature	0.62	-
Temperature	0.56	-
Summer rain	0.56	-
Annual rainfall	0.37	0.20
Late spring rain	0.32	0.15
pH	0.20	0.39
% sand, % silt, % clay	0.12 – 0.14	all 0.28
Total dissolved N	-	0.18
Gravimetric water content	-	0.10

Gross N mineralization and nitrification: soil and climate controls

Rates of gross N mineralization did not differ significantly by transplant site ($p = 0.15$) or the interaction of transplant site and origin ($p = 0.10$), but they did differ by site of origin ($p = 0.02$). The rates in soils with a South origin (4.8 ± 0.6 μg N g soil^{-1}

day^{-1}) were significantly greater than in soils of Middle origin (2.6 ± 0.5 μg N g soil^{-1} day^{-1}), and soils of North origin were intermediate (3.1 ± 0.7 μg N g soil^{-1} day^{-1}). Among all soil-climate combinations, rates of gross N mineralization ranged from (1.5 ± 0.4 to 6.4 ± 1.2 μg N g soil^{-1} day^{-1}). Rates of gross nitrification differed significantly among soils harvested from different sites (of transplantation), but not among soils from different origins or the interaction of transplant site and origin (*fig. 2*). Gross nitrification rates were higher when transplanted soils had been located in the North site compared to the other two sites (*fig. 2A*). In terms of soil origin, rates tended to increase from the South to the Middle to the North, but the differences were not significant (*fig. 2B*). There appeared to be a minimum soil water potential below which nitrification was seriously impaired (*fig. 3*). Below -0.05 MPa, rates of gross nitrification were always below 0.8 μg N g soil^{-1} day^{-1}. Above this water potential, rates varied widely, ranging between 0.0 and 9.5 μg N g soil^{-1} day^{-1}.

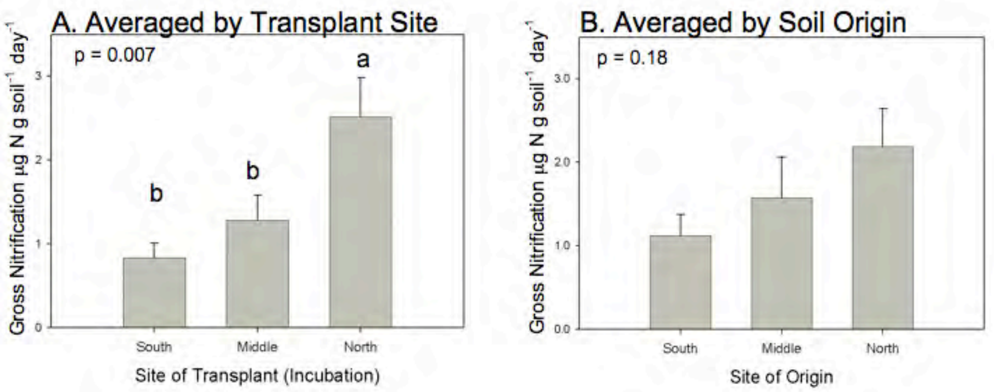

Figure 2—Gross nitrification graphed by (A) site of transplantation (incubation) and (B) site of origin for samples collected one year after transplanting. ANOVA p-values are in the upper left-hand corner and significant differences are denoted by lowercase letters. The interaction (not shown) was not significant (p = 0.32); thus, only the results averaged by site of transplantation and site of origin are shown to display the significant effect of transplantation.

Discussion

Response of bacterial and fungal community structure to a change in climate

Climate change resulting from transplanting soil cores across a 500-km latitudinal gradient in climate caused detectable changes in soil bacterial and fungal community structures within one year. Under all of the transplant scenarios, except one, the community structures of transplanted soils shifted to more closely resemble those of the sites into which they were transplanted. However, the community structures of the transplanted samples were still distinguishable from those native to their new sites. Given more time, the structure of a microbial community may

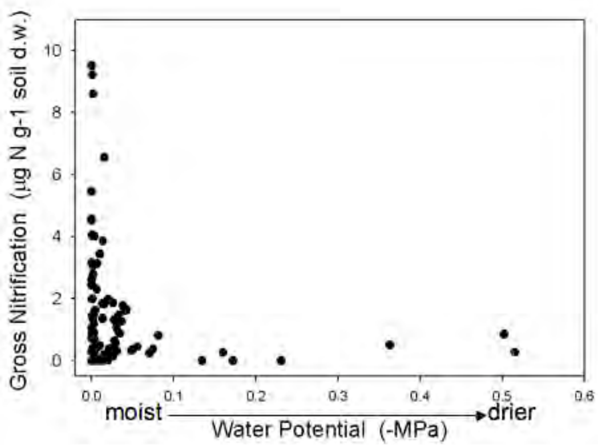

Figure 3—Rates of gross nitrification graphed against soil water potential for transplanted samples collected after 1 year.

continue to change, but it may also never fully resemble that native to the new site. While climate is clearly an important determinant of microbial community structure, it is certainly not the only significant controller.

The varying impacts of climate change on microbial community composition and structure that have been found across studies could be related to the strength of the change in climate to which soils are exposed. For example, in California, Waldrop and Firestone (2006) also found that soil microbial community structure changed when soils were transplanted from beneath oak canopies into an open grassland environment, but the reverse was not true (grassland soil microbial community structure did not change when transplanted beneath an oak canopy). They suggested that the exposure of communities to climatic conditions outside of their recently experienced range in historical climate caused a rapid change in community structure. In contrast, bacterial community composition differed little in response to increased rainfall after five years of rainfall manipulations in a coastal California grassland (Cruz-Martinez et al. 2009). However, in that study, all of the communities had experienced the same historical range in climate, one that likely encompassed the conditions experienced under the imposed rainfall manipulations. Additionally, Castro et al. (2010) observed changes in fungal and bacterial abundances and community structure after three years in response to manipulations of climatic drivers (temperature, precipitation and CO_2 concentration) in old-field ecosystems. They found that precipitation had the largest impact on community structure.

The climatic conditions of our three redwood forest sites overlap. The mean values for temperature and precipitation at all sites are within the ranges historically experienced by the indigenous communities. However, the extremes in precipitation and temperature (maxima and minima) occurring in the "new" sites post-transplanting may have provided additional stimulation for a change in community structure to occur. The warmest summer temperatures and driest soil conditions of the South site are outside of the range experienced in the North site, and the coolest winter temperatures in the North are slightly cooler than those experienced in the South. The relative importance of mean temperature and rainfall values compared to maxima and minima (and their duration) in determining differences in community composition and structure is an important but largely unexplored question.

While climate clearly influences community structure in this study, it is not the sole determinant. The predominance of intermediate changes in microbial community structure suggests that edaphic and/or biological factors also influence the trajectory and magnitude of the community response to climate. The structure of some communities may be slowly changing toward that of the new site, but in other instances, the structure of a transplanted community may never closely approach that of the new site's native community. Native soil characteristics can buffer against the effects of climate; in addition, biological interactions or environmental stochasticity could also cause the change in community structure to take a different trajectory. In this study of coast redwood soils, the strongest correlations between environmental variables and fungal community structure were found with temperature and rainfall, while bacterial community structure correlated most strongly with pH. The association of fungi with temperature and rainfall could indicate the importance of climate as a regulator of fungal decomposition, while soil pH has been found to be a strong environmental driver of bacterial community composition and structure (Chu et al. 2010, Fierer and Jackson 2006).

Impact of a change in climate on gross rates of N mineralization and nitrification

The responses of gross N mineralization and gross nitrification to imposed changes in climate differed. While rates of gross N mineralization differed somewhat among soils with different sites of origin, rates of gross nitrification differed significantly among transplant sites, that is, the sites at which the soil cores spent the preceding year (regardless of origin). Consequently, rates of gross nitrification were not correlated with rates of gross N mineralization. Instead, the impact of transplanting on gross nitrification seems to be due to the relationship between gross nitrification and water availability.

Rates of gross nitrification in coast redwood forests are clearly affected by soil water availability at the end of the summer, when north-south differences in fog frequency can cause substantial differences in climate between sites. The impacts of soil water availability on nitrification result from two primary factors: the effects of soil water content on the diffusion of NH_4 to ammonia oxidizers and the physiological impacts of the energetic availability of water (Stark and Firestone 1995). In this field study, there was a striking relationship between gross nitrification and soil water potential. Rates of nitrification were always low (< 0.8 μg N g soil^{-1} day^{-1}) below a water potential of -0.05 MPa. This water potential, however, is much greater (wetter) than the -0.6 MPa at which Stark and Firestone (1995) found physiological stress to become more important than diffusion in terms of limiting nitrification. Thus, the lower rates of nitrification below -0.05 MPa could be due to diffusional limitation of ammonium, or the ammonia oxidizers in these soils could be more sensitive to water stress since they are from a moister summer environment than the grassland soils studied by Stark and Firestone (1995). Ammonia-oxidizing bacteria are known to be able to carry out ammonia oxidation down to below -1.0 MPa in soils (Chen et al. 2011, Paul and Clark 1996), but little is known about the physiology of archaeal nitrifiers because few have been cultured (de la Torre et al. 2008, Konneke et al. 2005, Tourna et al. 2011). Soils in coast redwood forests are well suited to maintain relatively high water availability throughout the summer. The lowest water potential in this study was -0.5 MPa. The high organic matter contents (and associated high water holding capacity) and the presence of fog (which reduces evapotranspiration) can both contribute to this characteristic. Decreases in fog water inputs could, however, lead to drier soils (Ewing et al. 2009)

In contrast to gross nitrification, rates of gross N mineralization in coast redwood forests were relatively insensitive to any variable measured, edaphic or climatic. Other studies have reported a direct relationship between gross N mineralization and substrate availability (Booth et al. 2005), and a significant relationship between gross N mineralization and concentrations of soluble organic nitrogen was previously found for two northern redwood sites (Bradbury 2011). However, that study was conducted during a very wet time of year in late winter, March 2003, while this transplant study concentrated on a drier time of year at the end of summer. At the end of the dry summer period, it may be impossible to separate substrate and water controls of gross N mineralization. Mean rates of gross N mineralization varied between ~2 and ~6 μg N g soil^{-1} day^{-1} among the different origin-transplant site combinations in this study, but there was no consistent pattern in the variation of gross N mineralization among groups. Other studies conducted in pasture and grassland soils have found significant decreases in gross N mineralization at water potentials lower than those measured in these coast redwood soils (-1.5 MPa in Murphy et al. 1997 and -1.5 MPa in Jamieson et al. 1998), but a decrease in water potential from -0.1 MPa to -1.0 MPa did not cause a substantial decrease in gross N mineralization in the EA horizon of an acid coniferous forest soil (Chen et al. 2011). Thus, rates of gross N mineralization in forest mineral soils may not be highly influenced by decreases in soil water availability down to at least -1.0 MPa. Hence, greater magnitude changes in climate than those experienced in this study, or more than one year of exposure to a new climate, may be necessary to cause alterations in rates of gross N mineralization.

Implications for climate change in redwoods

Changes in summer fog frequency could have important effects on coast redwood forests. The importance of fog to coast redwood ecosystems in terms of supplying water and reducing transpiration during the otherwise dry summer has been demonstrated for coast redwood trees and their understory species (Dawson 1998, Limm et al. 2009). In this study, a regional-scale transplant-induced change in climate caused changes in fungal and bacterial community structure within one year. These changes occurred within the framework of the native soil characteristics, possibly as a result of the exposure of indigenous soil microbial communities to conditions outside of their historical range in climate. The relative importance of climate maxima and minima compared to climatic means requires further exploration. By examining the impact of differences in climate on gross N mineralization and nitrification, this study has helped to elucidate the importance of water availability for N-cycling processes that can control plant N availability at this critical time of year. While rates of gross N mineralization seem to be relatively insensitive to differences in summer climate across coast redwood forests, rates of gross nitrification are significantly affected by differences in summer climate and water availability. Therefore, continued reductions in fog frequency or amount, as have been observed over the last 100 years (Johnstone and Dawson 2010), may affect populations of ammonia-oxidizing bacteria and archaea and cause significant reductions in rates of gross nitrification and nitrate availability to redwoods and their understory species.

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Rangewide Genetic Variation in Coast Redwood Populations at a Chloroplast Microsatellite Locus[1]

Chris Brinegar[2]

Abstract

Old growth and second growth populations of coast redwood (*Sequoia sempervirens*) were sampled at 10 locations throughout its range and analyzed at a highly variable chloroplast microsatellite locus. Very low F_{ST} values indicated that there was no significant genetic differentiation between adjacent old growth and second growth populations at each location. Genetic diversity was moderate to high in populations north of the San Francisco Bay but low to very low in more southerly populations. Phylogenetic analysis produced a neighbor-joining tree with one clade composed of all populations north of San Francisco Bay and the other composed of the remaining southern populations. Differentiation of these two groups is consistent with the loss of rare alleles in the smaller, more fragmented southern populations by genetic drift.

Key words: chloroplast microsatellite, coast redwood, genetic drift, *Sequoia sempervirens*

Introduction

Ancestors of the modern coast redwood (*Sequoia sempervirens*) were once dominant forest species of the temperate northern latitudes. Glaciation ultimately forced the ancient redwood forests to retreat into the coastal fog belts of Oregon and California where they reached their extreme southern range limit near Santa Barbara in the late Pleistocene (Sawyer and others 2000). Genetic analysis of extant populations throughout the current 450 mile range could provide insights into the nature of historical expansions and contractions of redwood forests while shedding light on the evolutionary forces that will shape the composition of redwood forests in the future.

Three rangewide studies have provided data on the diversity and genetic structure of coast redwood populations. Hall and Langenheim (1987) detected a statistically significant geographic variation in leaf monoterpene composition. Cluster analysis revealed three major groups with the more divergent southern group separated from the more related central and northern groups by the San Francisco Bay. Growth performance differences in common garden experiments (Anekonda 1992) also supported a division between redwoods north and south of the San Francisco Bay. Douhovnikoff and Dodd (2011) detected a weak divergence between redwood lineages near the Sonoma/Mendocino border using nuclear microsatellite markers.

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[1] This paper was presented at coast redwood forests in a changing California, June 21-23, 2011, University of California, Santa Cruz.

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Molecular markers in haploid genomes have smaller effective population sizes compared to nuclear markers (Weising et al. 2005). Therefore, chloroplast microsatellites are typically more sensitive in detecting founder effects, bottlenecks, and differentiation by genetic drift (Provan et al. 2001). Among a set of coast redwood microsatellite markers developed by Bruno and Brinegar (2004) was the chloroplast *Seq*21E5 locus, a highly variable tetranucleotide repeat. Preliminary data from the *Seq*21E5 locus suggested a greater genetic diversity in northern coast redwood populations compared to populations in Santa Cruz County (Brinegar et al. 2007). However, the northern and southern populations were sampled from old growth and second growth forests, respectively. Whether the observed differences represented a north-south divergence or were due to regeneration artifacts could not be determined in that limited study.

The major goal of the current study was to determine whether a chloroplast microsatellite analysis at the *Seq*21E5 locus could provide independent confirmation of a north-south split in the genetic structure of coast redwood populations. A secondary objective was to ascertain if, in fact, there are detectable differences in genetic composition between old growth and second growth populations.

Methods

Sampling sites are shown in *figure 1*. At most sites, old growth and second growth populations were sampled in different areas within the boundaries of the same park or reserve. Exceptions were in the old growth/second growth collections at Armstrong Redwoods State Reserve/Austin Creek State Recreation Area, Big Basin Redwoods State Park/Castle Rock State Park, and Miller Creek/Lime Kiln State Park. In all cases, paired old growth/second growth populations were within five miles of each other. Redwood branchlets were collected from 40 trees spaced at least 10 m apart along transects. Tissue was dried in desiccant and stored at room temperature.

DNA extraction and PCR conditions were performed as described by Bruno and Brinegar (2004). PCR products were separated on 6.5 percent polyacrylamide gels (29:1 acrylamide:bisacrylamide, 40 mM Tris-acetate [pH 8.3], 1 mM EDTA) supplemented with 1X Spreadex Polymer NAB (Elchrom Scientific, Cham, Switzerland). Samples were electrophoresed for 3 h at 200 V and stained with 0.5 mg/L ethidium bromide. PCR product sizes were estimated using M3 markers (Elchrom Scientific). *Seq*21E5 size variants are referred to in this study as “alleles.”

Allele frequencies and Wright’s F_{ST} values, which represent the variance of allele frequencies among populations (Avise 2004), were calculated using HAPLOTYPE ANALYSIS (Eliades and Eliades 2009). Shannon diversity indices were calculated for populations from the formula: $S = -\sum p_i \log_2 p_i$ (Allnut and others 1999) where p_i is the frequency of the *i*th allele. Standardized Shannon indices where calculated by dividing *S* by its maximum value for a given sample size. A neighbor-joining tree was constructed in POPTREE2 (Takezaki and others 2010) using Nei’s D_A distance (Nei et al. 1983).

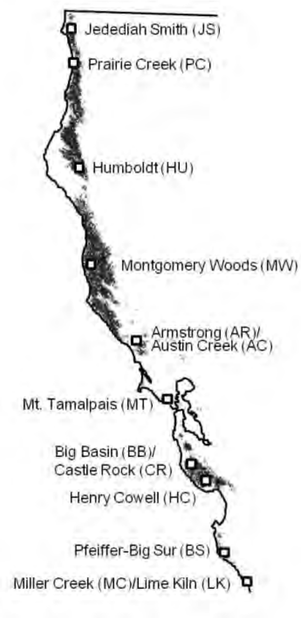

Figure 1—Map of coast redwood distribution in California with sampling locations.

Results

Comparison of old growth and second growth populations

The numbers of individuals with a given allele in the paired old growth and second growth populations are shown in *table 1*. Histograms of allele distributions based either on old growth or second growth allele frequencies are compared in *fig. 2*.

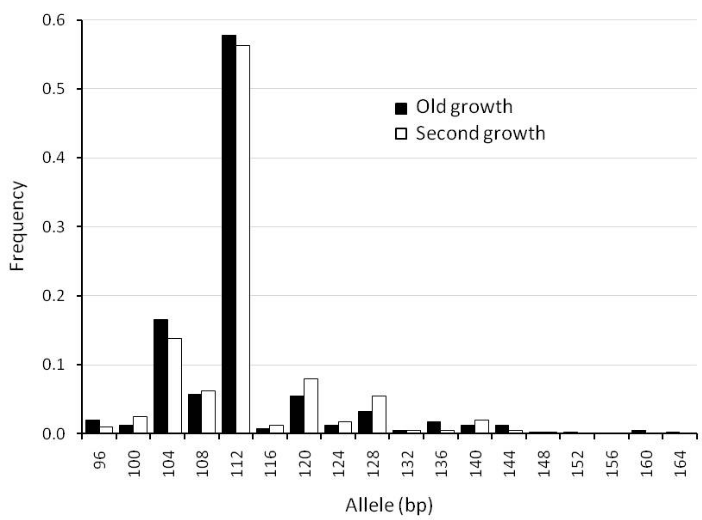

Figure 2—Allele frequencies in combined old growth populations and combined second growth populations.

Table 1—*Allele distributions (number of individuals) in paired old growth and second growth redwood populations.*[a]

	Allele size (bp)																	
Population[b]	**96**	**100**	**104**	**108**	**112**	**116**	**120**	**124**	**128**	**132**	**136**	**140**	**144**	**148**	**152**	**156**	**160**	**164**
JS-OG	4	0	6	7	11	1	3	1	2	1	0	3	1	0	0	0	0	0
JS-SG	1	0	6	4	19	1	2	0	4	0	0	3	0	0	0	0	0	0
PC-OG	1	1	8	4	15	0	5	1	2	0	3	0	0	0	0	0	0	0
PC-SG	1	0	7	1	14	0	8	1	5	1	0	1	0	1	0	0	0	0
HU-OG	1	0	1	2	28	0	1	1	2	0	0	1	1	1	1	0	0	0
HU-SG	0	2	6	6	18	1	3	1	2	0	0	1	0	0	0	0	0	0
MW-OG	0	1	6	6	11	2	5	1	3	1	1	0	0	0	0	0	2	1
MW-SG	2	2	5	5	8	1	8	2	1	1	0	3	2	0	0	0	0	0
AR-OG	2	1	6	4	16	0	7	0	2	0	1	0	1	0	0	0	0	0
AC-SG	0	3	5	3	17	1	6	0	3	0	2	0	0	0	0	0	0	0
MT-OG	0	2	6	0	26	0	1	1	2	0	0	1	1	0	0	0	0	0
MT-SG	0	3	2	4	21	0	1	3	6	0	0	0	0	0	0	0	0	0
BB-OG	0	0	15	0	22	0	0	0	0	0	2	0	1	0	0	0	0	0
CR-SG	0	0	8	0	29	0	2	0	1	0	0	0	0	0	0	0	0	0
HC-OG	0	0	12	0	28	0	0	0	0	0	0	0	0	0	0	0	0	0
HC-2G	0	0	10	2	26	0	2	0	0	0	0	0	0	0	0	0	0	0
BS-OG	0	0	5	0	35	0	0	0	0	0	0	0	0	0	0	0	0	0
BS-SG	0	0	4	0	36	0	0	0	0	0	0	0	0	0	0	0	0	0
MC-OG	0	0	1	0	39	0	0	0	0	0	0	0	0	0	0	0	0	0
LK-SG	0	0	2	0	37	1	0	0	0	0	0	0	0	0	0	0	0	0

[a] n = 40 for each population.
[b] Population abbreviations are defined in *figure 1*. OG = old growth; SG = second growth.

The two distributions, each based on data from 400 individuals, are extremely similar and include alleles ranging from 96 to 164 bp (corresponding to 4 to 21 repeat units).

Allele number and F_{ST} comparisons between paired old growth and second growth populations are displayed in *table 2*. Allele number ranged from two in some of the southernmost populations to 12 in the MW populations. In general, allele numbers in old growth populations are mirrored in their second growth counterparts. When differences in allele numbers occur, they are due to rare alleles.

Table 2—*Allele number and F_{ST} comparisons of paired old growth and second growth redwood populations.*

	Population[a]									
	JS	**PC**	**HU**	**MW**	**AR/ AC**	**MT**	**BB/ CR**	**HC**	**BS**	**MC/ LK**
OG[b] alleles	11	9	11	12	9	8	4	2	2	2
SG alleles	8	10	9	12	8	7	4	4	2	3
Normalized percent F_{ST}	0.7	0.3	1.6	0.4	0.1	0.8	1.3	0.2	0.0	0.2

[a] Population abbreviations are from *fig. 1*.
[b] OG = old growth; SG = second growth.

Wright's F_{ST}, when normalized to the maximum value possible, can be expressed as the percent of genetic diversity due to allele frequency differences among populations. Pairwise F_{ST} values for each old growth/second growth pair (*table 2*) are all extremely low (0.0 to 1.6 percent) indicating that there is negligible genetic differentiation between adjacent old growth and second growth populations throughout the coast redwood range. Therefore, for subsequent analyses, allele frequency data from the adjacent old growth and second growth populations at each location were combined in order to provide a larger population sample size (n = 80).

Rangewide variation in genetic diversity

The standardized Shannon diversity index is a measure of genetic diversity that takes both allele number and frequency into account. Values of 0 and 1 indicate no genetic diversity and maximum genetic diversity, respectively, within populations. *Figure 3* shows how genetic diversity at the ten sampling sites (combined old growth and second growth data) varies with distance from the northern range limit of coast redwood (approximately eight miles north of the Oregon/California border).

Populations north of the San Francisco Bay have standardized Shannon indices of 0.51 to 0.81 with the MW population being the most diverse. North of MW, diversity drops at the HU population owing to its high frequency of the 112 bp allele relative to the other populations in Mendocino, Humboldt and Del Norte Counties. Genetic diversity decreases almost linearly with distance in populations south of MW, falling to 0.08 in the southernmost population (MC/LK) where the 112 bp allele is nearly fixed (95 percent).

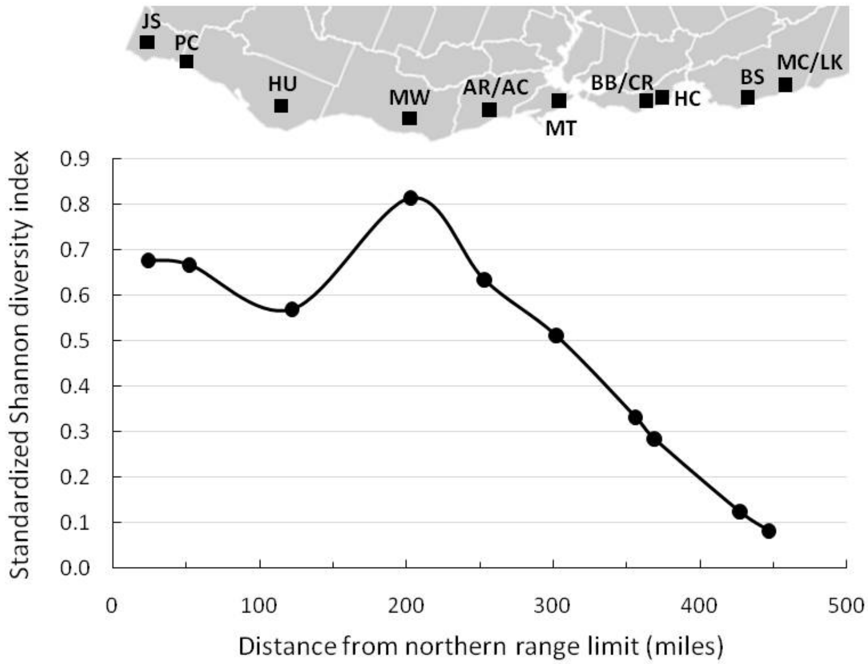

Figure 3—Standardized Shannon diversity index of combined old growth and second growth populations at each sampling location (indicated on map inset) vs. distance from the northern range limit.

Interpopulation genetic structure

Using the allele frequency data from combined old growth/second growth paired populations, an unrooted neighbor-joining tree based on Nei's D_A distances was constructed (*fig. 4*). The populations segregated into two clades: one containing the six populations north of the San Francisco Bay and one containing the four southernmost populations. Resampling of data by bootstrapping (Felsenstein 1985) is typically performed in order to determine the degree of support for the topography of phylogenetic trees. Unfortunately, bootstrapping values cannot be assigned to this tree which is based on data from a single locus.

Trees were also constructed using data from the old growth and second growth populations separately (not shown). The tree from second growth population data was essentially the same as the combined population tree. The tree from old growth population data differed only in the placement of the MT population which appeared on an interior branch of the southern clade. This suggests that the allele composition of the MT population, in Marin County, is teetering on the "borderline" between the northern and southern clades, indicating that it, too, may be succumbing to the effects of genetic drift.

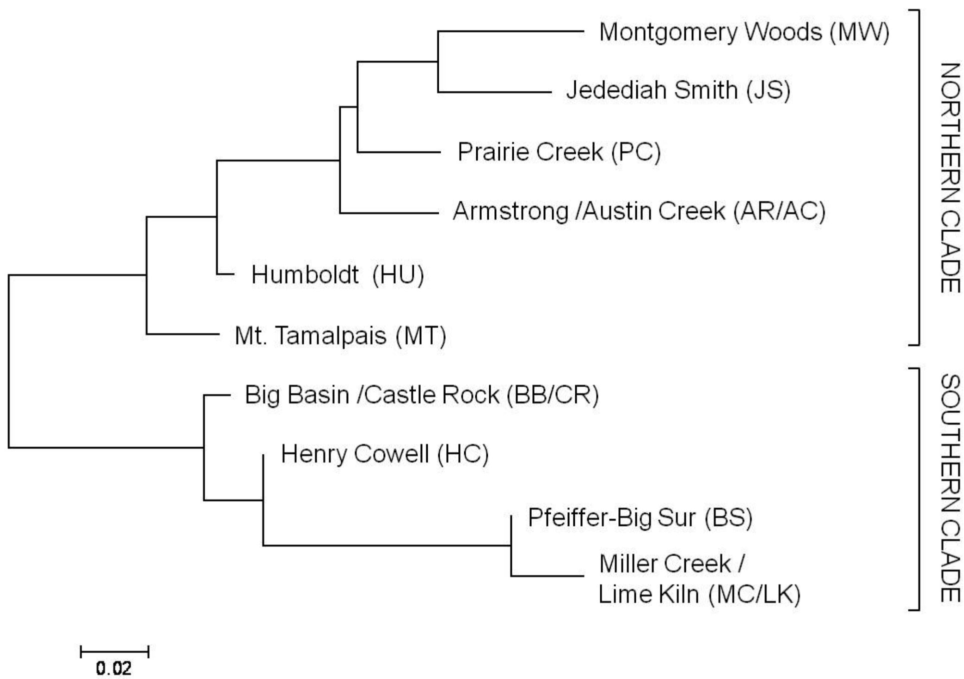

Figure 4—Neighbor-joining tree based on Nei's D_A distances calculated from combined old growth and second growth populations at each sampling location.

Discussion

In all 10 locations sampled throughout the range, second growth populations were not significantly different in their allelic compositions from adjacent old growth populations. Apparently, neither the decreased rate of stump sprouting in redwoods over the age of 400 years (Powers and Wiant 1970) nor the higher ratio of ramets per genet in second growth forests (Douhovnikoff et al. 2004) altered the genetic architecture of naturally regenerated forests to any significant extent.

Genetic diversity of coast redwood populations at the *Seq*21E5 locus varied considerably, with higher diversity levels in the northern populations and lower levels in the southern populations. The MW population, the most genetically diverse, is centrally located in the large tract of continuous coast redwood forest ranging from central Sonoma County to just north of the Mendocino County-Humboldt County border. It is likely that long distance gene flow via pollen would be substantial in this region, leading to higher diversity levels than would be present in smaller, more fragmented forests. An alternative explanation for the high diversity in the MW population is that this region could be closer to the center of expansion of the modern range's founding population and, being older, would be more diverse.

The two southernmost populations along the Monterey County coast, BS and MC/LK, are the least genetically diverse and are classic examples of how fragmented populations are susceptible to fixation through genetic drift. These and other populations along the Big Sur coastline are separated from each other by a severe canyon topography and therefore have little or no exchange of pollen with each other or with the more diverse populations to the north. Furthermore, Viers (1996) proposed that the warmer and drier conditions in the southern part of the range would cause greater difficulty in seedling establishment resulting in even higher rates of clonal reproduction than observed in northern forests. The high incidence of wildfires along the Big Sur coast would also lower diversity by promoting clonal over sexual reproduction. One can assume that the same evolutionary forces are at work on nuclear loci as well, although fixation will occur more slowly in a polyploid genome.

Although the Santa Cruz County populations (BB/CR and HC) had lower genetic diversity values and allele numbers compared to the northern populations, a larger sample study (n = 530) in Big Basin Redwoods State Park (Glavas 2006) detected 10 *Seq*21E5 alleles compared to six in this study's combined BB/CR population (n = 80). Between the two studies, 11 total alleles were observed in this general location. Although many of these alleles were found in only one or a few individuals, their presence raises the possibility that this region may have once harbored a forest as diverse as its neighbors to the north but has since become less diverse through a reduction in rare allele frequency through genetic drift.

The division of populations in the neighbor-joining tree (*fig. 4*) into northern and southern clades with a boundary at or near the San Francisco Bay is in keeping with results from previous biochemical, physiological and genetic studies (Anekonda 1992, Douhovnikoff and Dodd 2011, Hall and Langenheim 1987). Certainly, there is no substitute for using multiple loci in phylogenetic studies, but the large population sample sizes used in this study, the high variability of the *Seq*21E5 locus, and the low effective population size of the chloroplast genome all contributed to the detection of a north-south genetic division despite the use of only one locus.

While the major genetic transition zone between the northern and southern redwood populations has been identified by this and previous studies, the evolutionary relationships between redwood populations north of the San Francisco Bay remain unclear. The characterization and use of more chloroplast microsatellite loci would allow for a much more powerful phylogenetic analysis of that very important part of the redwood range.

Acknowledgments

I would like to thank the USDA Forest Service, the National Park Service, and the California Department of Parks and Recreation for issuing collecting permits; Rueben Bolton, Greg Brinegar, Michele Conrad and Steve Glavas for collecting assistance; Dr. Robert Fowler (San Jose State University) for the use of his laboratory for DNA extractions; and Bonnie Brinegar for generous financial support.

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Reference Conditions for Old-Growth Redwood Restoration on Alluvial Flats

Christa M. Dagley[1] and John-Pascal Berrill[1]

Abstract

We quantified structural attributes in three alluvial flat old-growth coast redwood stands. Tree size parameters and occurrences of distinctive features (e.g., burls, goose pens) were similar between stands. Occurrence of distinctive features was greater among larger trees. Tree size-frequency distributions conformed to a reverse-J diameter distribution. The range of tree sizes was similar between study sites. Redwood density ranged from 118 to 148 trees ha^{-1} and upper canopy tree density ranged from 45 to 74 trees ha^{-1}. Crown ratio was similar across study sites with an overall mean of 64.3 percent, except that crown ratio of the largest trees was lower at the site with the highest growing space occupancy. The percentage of plot area in canopy gaps ranged from 17 to 25 percent. Seedling regeneration was no more frequent beneath canopy gaps. These and other results describe structure in old-growth redwood forests and can serve as reference conditions for old forest restoration on alluvial flats.

Key words: canopy gaps, regeneration, *Sequoia sempervirens*, stand structure

Introduction

Old-growth redwood (*Sequoia sempervirens*) forests are structurally diverse, having been shaped over centuries by a wide range of forces (Lorimer et al.2009). Trees with diameters of 2 to 4 m and heights of 60 to 100 m are common. The desire to restore some of these features in managed forests has prompted application of various silvicultural practices, yet to date, little attention has been directed at developing a detailed description of structural parameters and distinctive features found in old-growth redwood forests that describe the historic condition and could serve as reference conditions for restoration.

Quantification of old-growth forest structure supports restoration efforts by providing reference conditions that could serve as targets for management activities (Harrod et al. 1999, SERISPWG 2004). The objective of this study was to identify structural parameters and distinctive features that were characteristic of alluvial flat old-growth redwood stands and may thus serve as targets for restoration of old forest structures. We hypothesized that: (1) multivariate analysis would not successfully discriminate between study sites on the basis of tree size and crown parameters; (2) seedling regeneration was no more frequent directly beneath canopy gaps than beneath tree crowns; and (3) the occurrence of distinctive features (e.g., epicormic sprouting) on live trees differed between sites, and was greater among larger trees. To test these hypotheses, calculate canopy gap areas and canopy volume, and summarize total live stem biomass, coarse woody debris, and regeneration data, we made detailed measurements of various tree size and crown size parameters, mapped tree locations, and recorded instances of distinctive features on individual redwood trees.

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Methods

Three study sites were selected—from the few larger remaining old growth groves on alluvial flats in northern California—to each represent different conditions and locations. Two of the study sites, named Children's and Rockefeller, were located at Humboldt Redwoods State Park in the southern part of Humboldt County (N 40°18.5', W 123°54.4'). The study site at Children's Forest in HRSP was located along the south fork of the Eel River adjacent to Myers Flat. Children's differed from the other two study sites in that a wildfire occurred there 1.5 years prior to study installation. The Rockefeller site in HRSP was approximately 14 km NW of the Children's site and located along the small perennial Bull Creek. Mean annual temperature for the HRSP area is 12.6 °C and mean annual precipitation is 123 cm. The third study site, named Armstrong, was located alongside a seasonally dry creek at Armstrong Redwoods State Preserve (ARSP) in the Russian River region of Sonoma County (N 38°53', W 123°0.6'). Mean annual temperature for ARSP is 13.9 °C and mean annual precipitation totals 104 cm.

Plants common to these redwood-dominated sites included redwood sorrel (*Oxalis oregano*) and western sword fern (*Polystichum munitum*). Hardwoods such as tanoak (*Notholithocarpus densiflorus*), bigleaf maple (*Acer macrophyllum*), and California bay (*Umbellularia californica*) were found in small numbers in openings or near streams.

At each site, a 1 ha sample plot (100 m x 100 m) was established with the objective of quantifying the horizontal and vertical structure. All trees ≥ 15 cm diameter at 1.37 m breast height (dbh) were mapped using a survey laser mounted on a tripod. Species, dbh, and location (e.g., azimuth and horizontal distance to the middle of each tree from reference points) were recorded. To assure mapping was accurate, the angle and distance from plot center to each reference point was recorded and cross-checked with tree locations taken from both plot center and the respective reference point. Additionally, all trees ≥ 15 cm dbh were measured for total height, live crown base height, and crown radius. Live crown base height was defined as the height of the lowest major branch that formed part of the main canopy. The number of crown radius measurements varied with complexity of crown shape. A minimum of four crown radius measurements were taken, each in a cardinal direction, for small trees with circular crowns. A maximum of eight measurements were taken for large trees with irregular crowns. Crown class, crown shape, and crown fullness estimates were recorded for each tree. The three-dimensional shape of individual tree crowns was most often described as a parabola; however, some were described as a cone, cylinder, or diamond. Crown fullness was an ocular estimation of the percentage of the live crown occupied by branches and foliage (e.g., 50, 66, 75 percent). Because older trees often have discontinuous crowns, incorporating the "fullness" variable into the crown volume equation provided a more accurate estimate of actual space occupied by live tree crowns.

For each redwood tree, we noted presence or absence of the following distinctive features: burls, epicormic sprouting, goose pens, and reiterations. A goose pen was defined as a large fire-created basal hollow greater than 30 or 60 cm tall for trees less than or greater than 1 m dbh, respectively. A reiteration was defined as an erect sprout or branch found on an existing tree but supporting its own network of horizontally oriented branches. Three 100 m north-south transects were established in each 1 ha plot with the objective of quantifying regeneration, understory vegetation, and groundcover. Height, location, species, and regeneration source (i.e., sprout versus seedling) were recorded for all trees < 15 cm dbh in a continuous 4 m wide belt centered on each transect line. Percent ground cover values were obtained by dividing the entire length of each transect line into one centimeter sections and recording species of shrubs, herbaceous vegetation, bareground, and litter covering ground directly below the transect line. Down logs ≥ 30 cm diameter and ≥ 2 m length were mapped and measured to determine the volume, mass, and percent cover of large down wood at Children's and Rockefeller. Diameter at the midpoint of each length of down log, log length, and location were recorded. Only the portions of logs falling inside the plot were measured. Down logs at Armstrong were not measured due to a history of removal of down wood from the site. Standing dead trees (snags) ≥ 15 cm dbh were mapped and measured for dbh at the three study sites.

Data for live trees with dbh ≥ 15 cm were summarized to give per-hectare stem density, basal area, and stem volume, stand density index (Reineke 1933), and maximum tree height for each species at the three study sites. Stem volume (V) was calculated using a simple conic shape: $V=\pi(0.5\ dbh)^2h/3$ where dbh = dbh in m and h = total tree height in m. Stand density index (SDI) was calculated as a summation of individual tree values because the dbh data were not normally distributed: $SDI=\sum(0.04 dbh_i)^a$ where dbh_i = dbh in cm of the i^{th} tree in the plot, and a = 1.605 (Long and Daniel 1990, Shaw 2000). Trees were grouped into 30 cm dbh size classes to give size-frequency distributions for each site. Individual tree size parameters of dbh, height, crown radius, crown volume, and crown ratio (ratio of crown length to total tree height) were analyzed using canonical discriminant analysis. The objective was to identify individual parameters or groups of parameters that differed least between sites, assuming these 'commonalities' would be more useful targets for restoration than parameters that differed between sites. Prior to analysis, a logarithmic transformation of dbh was applied to achieve normality of the residual variances.

After arc-sine transformation, average crown ratio was calculated for redwood trees in each 30 cm dbh size class. Average crown ratios for each size class were subjected to an ANOVA to test for significant differences (α = 0.05) between study sites. The length of clear bole was defined as the portion of the tree stem not occupied by the canopy (equivalent to live crown base height). Clear bole length was summarized by 30 cm dbh size classes to further describe the vertical structure of standing trees at each site. Crown volume for each tree was calculated using crown length, radius, shape, and fullness. Summing crown volume data for all trees in the 1-ha plots allowed for a site-by-site comparison of total canopy volume. To determine the vertical distribution of crowns and crown volume at each site, individual tree crown volume was separated into layers at 10-m height intervals and summed for each 10 m layer.

ArcGIS ArcMap (ESRI) was used to delineate and calculate plot area not covered by the downward projection of tree crowns. Crown canopy maps of each site were created using stem locations and crown radius measurements. Spatial coordinates of crown extent were created for each crown radius measurement and then connected for each tree to create polygons representing tree crown extent. Polygon boundaries were smoothed using a t-spline. To obtain an accurate estimate of gap area within the 1 ha plots, portions of the crown of trees surrounding the plot that extended inside plot boundaries were included in the crown canopy maps. This necessitated mapping and measurement of crown radii for trees with crowns encroaching on the plot area. Crown canopy maps were used to examine the relationship between canopy gaps and regeneration.

Location coordinate data for regeneration in each 4-m wide belt transect were imported into the crown canopy maps for each site. Each record of regeneration (all trees <15 cm dbh) was coded as occurring directly beneath a canopy gap or beneath an opening in the canopy. Binomial proportions tests were used to test for differences in occurrence of regeneration beneath tree crowns versus regeneration under canopy gaps, by species type (i.e., redwood or hardwood) and source of regeneration (i.e., sprout or seedling). Count data for regeneration under canopy gaps or tree crowns were each divided by the proportions of transect area either under canopy gaps or beneath tree crowns, respectively, making count data comparable when different proportions of transect area were located directly beneath the canopy and under canopy gaps.

Logistic models were developed to predict probability of occurrence of burls, epicormic sprouting, goose pens, and reiterations on individual redwood trees as a function of tree-size parameters. Dummy variables for study site were also included to test for significant differences in presence-absence of these distinctive features between the three study sites. Model goodness-of-fit in terms of -2 Log Likelihood was compared between models to identify the tree size parameter (i.e., dbh, height, live crown base height, crown length, crown radius) most strongly associated with probability of occurrence of distinctive features on redwood trees. Data were analyzed using SAS statistical analysis software (SAS Institute 2004).

Results

Redwood density ranged from 118 to 148 trees ha^{-1} and density of all live stems ranged from 118 to 183 trees ha^{-1} for the three study sites (*table 1*). The two sites at Humboldt Redwoods State Park (Children's and Rockefeller) were almost completely comprised of redwood. In contrast, the site at Armstrong Redwood State Park (Armstrong) contained a mix of hardwoods. Hardwood crowns had not attained upper canopy status. The distribution of stems by 30 cm dbh class indicated that Armstrong's higher stem density was mostly in the small size classes and because of the hardwood presence. The frequency distribution of redwood tree diameters was a reverse-J shape at the three study sites. The two northern sites (Children's and Rockefeller) contained larger diameter trees while a narrower distribution of diameters at Armstrong was observed (*fig. 1*). Mean dbh was similar at Armstrong and Children's (1.10 and 1.08 m, respectively). Mean dbh at Rockefeller was 1.45 m.

Table 1—*Species composition and stem density per hectare by crown class for all live trees ≥ 15 cm dbh in the 1-ha plot at the three alluvial flat old-growth redwood study sites.*

Site	Species	Dominant	Codom.	Intermediate	Suppressed	Total
Armstrong	*S. sempervirens*	47	27	27	47	148
	A. macrophyllum	--	--	4	2	6
	N. densiflorus	--	--	5	2	7
	U. californica	--	--	19	3	22
	Total	56	27	46	54	183
Children's	*S. sempervirens*	13	33	45	37	128
	U. californica	--	--	2	1	3
	Total	14	33	46	38	131
Rockefeller	*S. sempervirens*	29	16	44	29	118

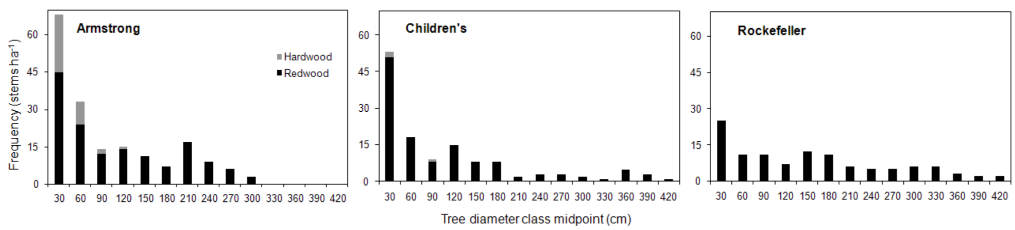

Figure 1—Diameter distribution of live trees ≥ 15 cm dbh for 1-ha plots at the three alluvial flat old-growth redwood study sites.

Structural attributes of redwood basal area, canopy volume, and total standing stem volume tended to be similar at Children's and Armstrong study sites, and highest at Rockefeller (*table 2*). Among these structural attributes, total standing stem volume differed most between study sites. Basal area and total standing stem volume at Rockefeller were at least 30 and 37 percent greater, respectively, than at Children's and Armstrong. Similarly, redwood SDI was greater at Rockefeller than at Children's and Armstrong (metric SDI: 2529, 1884, 2057, respectively). Maximum height was similar at the two northern sites and taller compared to Armstrong. Total canopy volume for the three sites was similar in overall value but was distributed differently along the vertical profile (*table 2*, *fig. 2*). Peak canopy volume values differed between the three study sites; Armstrong peaked at a lower height (40 m) than the two northern sites, Children's peaked at 50 m, and Rockefeller had the most crown volume at 60 m above ground.

Table 2—*Basal area (BA), canopy volume, total standing stem volume, and height of the tallest tree (Max. Ht.) for live trees ≥ 15 cm dbh in the 1-ha plot at the three alluvial flat old-growth redwood study sites.*

Site	Species	BA (m^2 ha^{-1})	Canopy volume (m^3 ha^{-1})	Stem volume (m^3 ha^{-1})	Max. Ht. (m)
Armstrong	*S. sempervirens*	228.3	168,281	5,542	95
	A. macrophyllum	0.3	838	1.7	24
	N. densiflorus	0.3	588	1.3	18
	U. californica	4.8	15,363	60.2	49
	Total	233.7	185,070	5,605	--
Children's	*S. sempervirens*	225.1	171,806	6,732	105
	U. californica	0.8	1,186	15.9	59
	Total	225.9	172,992	6,748	--
Rockefeller	*S. sempervirens*	307.8	174,588	9,272	107

Canonical discriminant analysis revealed significant structural differences between the three sites. Armstrong and Children's sites were found to be the most different ($P < 0.0001$) while the two northern sites, Children's and Rockefeller, were found to be the least different ($P = 0.007$). The analysis discriminated between the sites predominantly on the basis of dbh and height, suggesting that height to diameter ratio was the most important structural difference between sites. The analysis also revealed different tree diameter – crown volume relations between sites. However, modest allocation rates (42 to 57 percent) indicated that the sites could not be separated easily by tree size parameters.

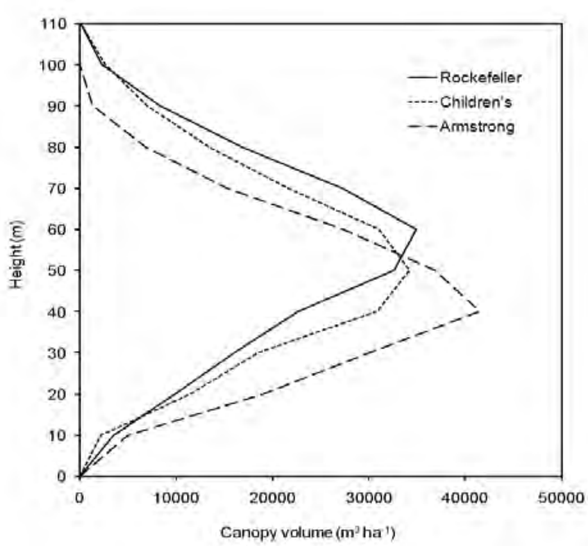

Figure 2— Canopy volume distribution by 10-m height interval along vertical profile for each alluvial flat old-growth redwood study site, based on tree crown volume estimates for redwood stems ≥ 15 cm dbh.

Long crowns were a common feature at the three study sites (*table 3*). The overall mean crown ratio for the three study sites was 64.6 percent. An F-test failed to detect significant differences in mean crown ratio for each 30-cm dbh size class between the three study sites ($P = 0.14$). However, differences in crown ratio between sites were detected in an F-test of tree crown ratio data for large trees (i.e., trees > 1.5 m dbh; $P = 0.001$). Further analysis of data for large trees detected no significant difference in crown ratio between Children's and Armstrong sites ($P = 0.92$), whereas crown ratios at these two sites differed significantly from the Rockefeller site ($P \leq 0.005$) where the larger trees had longer clear boles and shorter crowns (*table 3*).

Table 3—*Average crown ratio (CR) and length of clear bole (CB) by 30 cm dbh size class for all redwood trees ≥ 15 cm dbh at the three alluvial flat old-growth redwood study sites.*

Dbh class	Armstrong			Children's			Rockefeller		
midpoint (cm)	**n**	**CR**	**CB (m)**	**n**	**CR**	**CB (m)**	**n**	**CR**	**CB (m)**
30	45	.61	6	51	.62	8	31	.68	6
60	24	.71	9	18	.70	13	11	.76	9
90	12	.69	16	8	.69	21	11	.69	17
120	14	.65	22	15	.66	24	7	.67	23
150	11	.64	24	8	.66	27	12	.69	22
180	7	.74	21	8	.61	34	11	.58	34
210	15	.67	23	2	.74	22	6	.64	32
240	11	.55	35	3	.68	30	5	.50	44
270	6	.61	32	3	.64	34	5	.59	40
300	3	.65	30	2	.65	35	6	.58	41
330	0	--	--	1	.47	47	6	.50	47
360	0	--	--	5	.61	39	3	.50	47
390	0	--	--	3	.73	27	2	.62	38
420	0	--	--	1	.84	15	2	.54	46
Mean	--	.65	--	--	.65	--	--	.64	--

The sum of canopy gap areas as a percentage of total plot area was similar at the three sites, ranging from 17 to 25 percent. Gap shape and size were variable (*fig. 3*). For gaps contained within plot boundaries there was an approximate reverse-J distribution in gap sizes. Rockefeller had one large gap (>1272 m^2) which extended beyond the plot boundary and accounted for 50 percent of the total gap area within the plot.

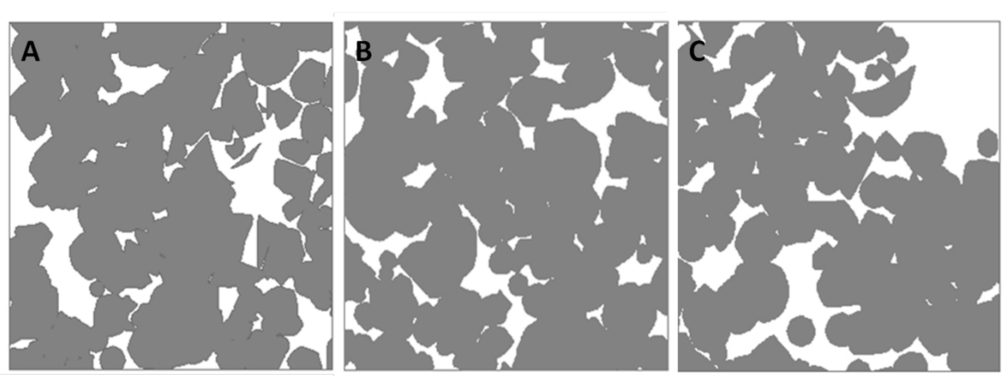

Figure 3—Canopy map of tree crown projections in 1-ha plots at (a) Armstrong; (b) Children's; and (c) Rockefeller alluvial flat old-growth redwood study sites.

Regeneration density and species composition differed between study sites (*table 4*). Hardwood trees < 15 cm dbh were found at Armstrong and Rockefeller. Armstrong contained a mix of hardwoods including tanoak, California bay, and red alder (*Alnus rubra*). In contrast, Rockefeller contained only one hardwood species, tanoak. Redwood sprouts and seedlings were present at all three sites. Binomial proportions tests did not detect differences between the occurrence of redwood sprouts or seedlings occurring in canopy gaps and redwood regeneration beneath the canopy ($P \geq 0.31$), nor between hardwood sprouts or individuals occurring in gaps or beneath the canopy ($P \geq 0.12$). Most redwood regeneration found along transects was in the form of basal sprouts associated with an existing root system.

Table 4—*Regeneration density per hectare for all trees < 15 cm dbh in three 400 m^2 belt transects at each alluvial flat old-growth redwood study site.*

Site	Species	Sprouts	Seedlings	Total
Armstrong	*Sequoia sempervirens*	1125	48	1173
	Hardwoods	404	538	942
Children's	*Sequoia sempervirens*	1538	25	1563
Rockefeller	*Sequoia sempervirens*	158	150	308
	Hardwoods	483	517	1000

The main constituents of the forest floor were similar among sites, with redwood sorrel, sword fern, and litter most prevalent. However, the proportions of these elements differed between sites. Rockefeller contained almost an even mix of litter, redwood sorrel, and sword fern (35, 21, and 32 percent, respectively). Children's groundcover was almost a complete blanket of redwood sorrel (76 percent), probably as a result of the fire. Armstrong's ground cover was dominated by litter (54 percent) followed by redwood sorrel (30 percent), and sword fern (6 percent). Consistent with the higher tree species diversity found at Armstrong, the understory was also most diverse and included fragrant bedstraw (*Galium triflorum*), trail marker (*Adenocaulon bicolor*), fairybell (*Disporum hookeri*), and wood rose (*Rosa gymnocarpa*).

Epicormic sprouting was the most common distinctive feature at each site. Burls, epicormic sprouting, goose pens, and reiterations were more common among larger redwood trees (*fig. 4*). Most redwood trees had one type of distinctive feature and very few trees had all of these features. The natural logarithm of tree dbh was a stronger predictor of the probability of occurrence of distinctive features than either tree height, live crown base height, crown length, or crown radius. The probability of occurrence of epicormic sprouting was greater at Children's and lower at Armstrong when compared with incidence of epicormic sprouting for any given tree size at Rockefeller. Logistic model coefficients indicated that redwood trees at Children's were less likely to have a goose pen than trees of equivalent dbh at either Armstrong or Rockefeller (*table 5*).

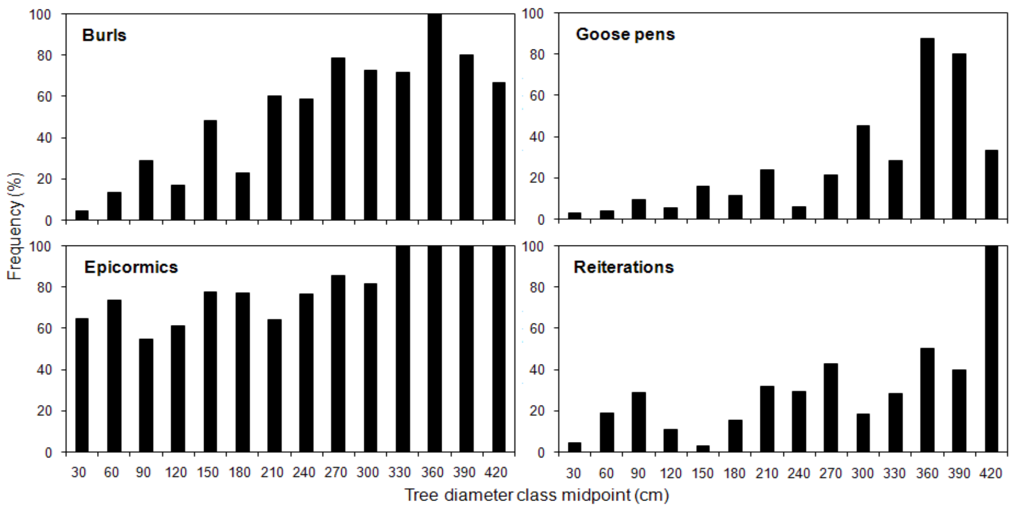

Figure 4—Percentage frequency distribution of redwood trees having a distinctive feature by 30 cm dbh class, based on pooled data from the three alluvial flat old-growth redwood study sites.

Table 5—*Coefficients and fit statistics for logistic model of the presence of burls, epicormic sprouts, goose pens, and reiterations as a function of the natural logarithm of dbh for alluvial flat old-growth redwood trees ≥15 cm dbh (n=394). Dummy variables for study sites were included when differences detected between sites (α=0.05).*

Distinctive Feature	Parameter	Coefficient	s.e.[a]	Wald's χ^2	Pr > χ^2
Burls	Intercept	1.0617	0.15	52.70	<0.0001
(Model -2LL = 346.5)	*Ln* dbh	-1.7086	0.20	76.08	<0.0001
Epicormic sprouts	Intercept	-1.0899	0.13	70.94	<0.0001
(Model -2LL = 438.9)	*Ln* dbh	-0.3871	0.13	9.43	0.0021
	Armstrong	-0.7871	0.16	25.48	<0.0001
	Children's	0.7660	0.19	16.47	<0.0001
	Rockefeller	0.0000	--	--	--
Goose pens	Intercept	2.4330	0.23	107.86	<0.0001
(Model -2LL = 225.8)	*Ln* dbh	-1.7091	0.28	36.91	<0.0001
	Armstrong	0.2623	0.25	1.12	0.2894
	Children's	-0.8022	0.24	11.31	0.0008
	Rockefeller	0.0000	--	--	--
Reiterations	Intercept	1.6237	0.15	123.64	<0.0001
(Model -2LL = 327.1)	*Ln* dbh	-0.8489	0.17	24.51	<0.0001

-2LL = -2 Log Likelihood, measure of goodness of fit.
[a] s.e. = standard error for coefficient.

The number of large fallen logs was similar at Children's and Rockefeller study sites. Logs at Rockefeller were slightly longer and larger in diameter which resulted in almost twice the amount of volume and area covered by logs at Rockefeller (*table 6*). There appeared to be no trend in the direction of tree fall and most of the logs were found to be scattered widely across each site. One exception occurred at Rockefeller. The eastern side of the plot contained a large gap which was created by three large fallen trees that fell in the easterly direction.

Table 6—*Coarse woody debris characteristics for Children's and Rockefeller alluvial flat old-growth redwood study sites. Minimum dimensions of logs sampled were 2 m length and 30 cm midpoint diameter. Standard error for mean log diameter shown in parentheses.*

Site	Density (logs ha^{-1})	Mean diameter (m)	Volume (m^3 ha^{-1})	Cover [a] (%)	Mass [b] (tons ha^{-1})
Children's	14	1.09 (0.12)	570	3.7	114
Rockefeller	19	1.25 (0.17)	1072	6.8	214

[a] Cover represents percent ground area covered by downward projection of logs.
[b] Mass estimated as volume by an averaged wood density from Bingham and Sawyer (1988).

Standing dead trees (snags) ≥ 15 cm dbh appeared to be exclusively redwood at the three sites. The two snags within the plot at Armstrong were 1.5 and 1.6 m dbh. The two snags at Rockefeller were 0.7 and 3.5 m dbh. The eight snags at Children's included four small dead-standing redwood trees (dbh < 40 cm) that appeared to have sustained severe fire damage and died recently. The other four snags at Children's ranged from 0.9 to 2.2 m dbh.

Discussion

Our work provides an important description of structural characteristics in alluvial flat old-growth redwood forests. Density, diameter distribution, crown ratio, and total canopy gap area were most similar among study sites. As such, they may serve as key reference conditions for future restoration treatments (*table 7*).

Table 7—*Reference conditions for alluvial flat old-growth redwood forests.*

Condition	Description
Stand density[a]	Redwood 118-148 trees ha^{-1}
Upper canopy tree density[b]	45-74 trees ha^{-1}; mean dbh 2.1 m; range 0.55-4.27 m dbh
Crown ratio	Trees > 1.5 m dbh: 0.56-0.64; trees ≤ 1.5 m dbh: 0.65
Canopy gap area	17-25% total area in gaps
Coarse woody debris[c]	Density 14-19 logs ha^{-1}; mean midpoint diameter 1.2 m; mean length 26.6 m; cover 3.7-6.8%; mass 114-214 tons ha^{-1}
Snags	2-4 dead standing trees ha^{-1}; range 0.9-3.5 m dbh
Spatial pattern	Random pattern for redwood > 1.5m dbh (Dagley 2008)

[a] Trees ≥ 15 cm dbh.
[b] Dominant and codominant trees.
[c] Logs ≥ 30 cm midpoint diameter and 2 m length.

Similarities between our data and published density data suggest both overall redwood density and upper canopy density can be useful reference points in old forest restoration efforts. Van Pelt and Franklin (2000) reported a main canopy density of 46 trees ha^{-1}. Sugihara (1992) reported an overall redwood density of 182 trees ha^{-1} for trees > 10 cm dbh and a density of 67 trees ha^{-1} for redwood trees > 1.5 m dbh. A redwood density of 107 trees ha^{-1} for trees > 10 cm dbh was reported for a stand located in Prairie Creek Redwoods State Park (Sawyer and others 2000). Fujimori (1977) reported 66 trees ha^{-1} for trees > 1 m dbh.

Stand density index (SDI) is a measure of growing space occupancy, and is useful for quantifying relative density across a wide variety of stand conditions (Long and Daniel 1990). Reineke (1933) reported that the maximum SDI for redwood is approximately 2500 (equivalent to an SDI of 1000 in English units) based on data from even-aged second growth stands. The SDI was highest at Rockefeller, and slightly exceeded the maximum reported by Reineke (1933). This could explain the lower crown ratios among larger trees (*table 3*) and lower regeneration density (*table 4*) at Rockefeller.

Old-growth redwood tree crowns occupy an immense amount of space. Van Pelt and Franklin (2000) reported a canopy volume of 230,100 m^3 ha^{-1} with the maximum occurring at a height of 50 m for a stand on the alluvial flats in Humboldt Redwoods State Park. This value exceeded our estimates (172,992 to 185,070 m^3 ha^{-1}; *table 2*) but was based on a conic representation of tree crown volume. We generated more realistic estimates by describing the shape and fullness of individual tree crowns. Crown shape impacts volume estimation and placement of crown volume in the vertical profile (*fig. 2*). For example, the volume of a parabola is 50 percent greater than that of a cone. When compared with crown volume estimates obtained using a cone shape, canopy volume estimates from this study were 10 m higher in the vertical profile when shape was taken into account. The shape was most often described as a parabola and the average crown fullness was 60 percent.

Total canopy gap area ranged from 17 to 25 percent between sites. These values closely resembled reported values from other old-growth stands ranging from 18 to 20 percent (Busing and Fujimori 2002, Sugihara 1992, Van Pelt and Franklin 2000). Gap size and shape were variable, suggesting that treefall events involved individual trees or groups of trees.

A preference to establish under canopy gaps was not detected among redwood regeneration. Hunter (1995) found the influence of canopy gaps to only account for 4.6 percent of the variation in understory light levels in a mixed evergreen forest (containing redwood) in Northern California. He proposed that several factors limit the effects of canopy gaps on the understory: temperate latitude placing the sun at lower angles, a dry summer season restricting growth when light is at its highest angle in the sky, a tall canopy, and small gap diameters. Van Pelt and Franklin (2000) found no relationship between understory tree location and canopy gaps.

Exposure to wind and windstorm events likely affects stem form. Results from the multivariate discrimant analysis revealed differences in tree height-diameter relations between study sites. Our use of a simple conic shape to estimate stem volume is repeatable, but highlights need for reliable predictive volume and taper equations. Existing equations predict merchantable volume above a tall stump to large top diameters, not total stem volume. Differences in stem taper and incidences of top breakage and reiteration will complicate the task of modeling stem volume and whole-tree biomass: a priority for future research.

Our characterization of coarse woody debris highlighted differences between sites (*table 6*). These differences may in part be due to stand history. Over time, down logs may have been removed for use or aesthetics. Yet, the lowest published estimates remain two to five times greater than those reported for most other temperate or tropical forests (Franklin and Waring 1980). Our estimates of 114 and 214 metric tons ha^{-1} for Children's and Rockefeller, respectively, fall within the range of values reported for alluvial flats (Busing and Fujimori 2005, Sugihara 1992).

The study sites at Armstrong, Children's, and Rockefeller were selected to represent alluvial flat old-growth redwood forests with different site characteristics and/or recent disturbance histories. Our data and analyses revealed that each stand had unique attributes, but also shared features with the other study sites, suggesting that these commonalities could represent general reference conditions or “targets” for future restoration efforts (*table 7*). For example, managers seeking to accelerate development of old-growth characteristics in a young stand might begin by identifying approximately 50 to 80 overstory trees ha^{-1} in a random spatial arrangement, and releasing these trees from competition. Retaining more trees would allow for artificial creation of 2 to 4 snags ha^{-1} in the absence of natural mortality. Concurrently identifying one or more open areas to serve as large canopy gaps between overstory tree crowns might preclude future conflict between overstory tree density and canopy gap area requirements. Less regard, at least initially, might be paid to structural parameters or distinctive characteristics that differed between the three reference sites or that correlated with tree size. When combined with descriptions of the spatial pattern of tree locations within stands (Dagley 2008), this work represents a detailed quantitative description of the three dimensional structure and complexity found in old-growth redwood forests on alluvial flats.

Acknowledgments

The project was funded by Save-the-Redwoods League and The Center for Forest Disturbance Science, Southern Research Station, USDA Forest Service.

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Fog and Soil Weathering as Sources of Nutrients in a California Redwood Forest

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Abstract
Fog water deposition is thought to influence the ecological function of many coastal ecosystems, including coast redwood forests. We examined cation and anion inputs from fog and rain, as well as the fate of these inputs, within a Sonoma County, California, coast redwood forest to elucidate the availability of these ions and some of the biotic and abiotic processes that may influence their relative abundance. At this site, the patterns of water and chemical inputs via fog and rain and their movement through the soil-plant ecosystem differed between the summer fog and winter rain seasons. Most (98 percent) of the annual water and more than three quarters of the total ionic load was delivered to the forest during the rain season. Soil water patterns followed those of throughfall. Water for plant use was most available in the rain season; however, after large fog events (fog season) plant-available soil water was also present at the forest edge. Differences between soil water and throughfall chemistry were a function of the mobility of each ion, whether or not an ion was a soil weathering product, and the likely biological demand for the ion. The impact of redwoods as fog catchers, transformers, and redistributors of both water and nutrients may extend all the way into the soil profile: in our plots, organic-rich soil horizons were thicker at the forest edge than in the forest interior. Our data show that, although total fog water inputs were small compared to inputs from rain, fog carried a large proportion of the total aqueous ionic inputs—inputs that, presumably, continued to be biologically available until their loss during the rain season. Cross-seasonal, functional coupling of aboveground (canopy) and belowground (soil) processes are likely to be prevalent in this redwood and other fog-inundated forests.

Key words: canopy, fog, input-output, rain, redwoods, soil

Introduction

Fog water deposition is thought to influence the ecological function—from plant physiology (Burgess and Dawson 2004, Limm et al. 2009, Simonin et al. 2009, Williams et al. 2008) to ecosystem function (Ewing et al. 2009, Weathers et al. 2000)—of many coastal forests, including coast redwood forests. Within California, redwoods inhabit a narrow strip of land from approximately 42 to 35.8 degrees N latitude and a zone fewer than 40 km from the ocean, a region known for its summer fog and winter rain (Noss 2000). Research on soil water and understory plants (Dawson 1998, Limm and Dawson 2010, Limm et al. 2009) and trees (Ambrose et al. 2009, Burgess and Dawson 2004, Ewing et al. 2009, Ingraham and Matthews 1995,

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Limm et al. 2009, Simonin et al. 2009) has shown that fog water is taken up directly into plant leaves within fog-enshrouded coastal California forests.

Much less is known about the influence of fog on biogeochemical processes, especially in regard to the influence of fog chemistry on ecological function. Nutrients and chemicals are consistently more concentrated in fog than rain (Collett et al. 2002, Weathers et al. 1986, Weathers et al. 1988, Weathers et al. 2000), suggesting that fog could be an important vector of nutrients and pollutants even when it contributes a relatively small fraction of the total water input (Azevedo and Morgan 1974, Ewing et al. 2009, Weathers 1999, Weathers and Likens 1997, Weathers et al. 1988, Weathers et al. 2000). In the redwood forests of California where fog drip is high in the summer, fog brings a fifth (21 percent) of the total nitrogen delivered via atmospheric deposition and below-canopy drip to the forest floor even though it delivers as little as six percent of the water to the same location (Ewing, et al. 2009). Fog in California, like other coastal and inland regions, can carry substantial amounts of calcium, magnesium, potassium, and sodium (Ca^{+2}, Mg^{+2}, K^{+}, and Na^{+}; for example, Weathers et al. 1986), and it has been proposed that these elements might influence nutrient cycling, soil fertility, and understory plant growth (Azevedo and Morgan 1974, Weathers 1999, Weathers et al. 1986) by enhancing the availability of essential major or minor nutrients. However, no studies have examined delivery of these nutrients to redwood forests via fog and rain within the same forest nor their abundance in soil water.

We examined fog and rain inputs, as well as the fate of these inputs, within a Sonoma County, California, coast redwood stand to elucidate the potential role of fog in the cycling of major mineral cations and anions. Our objectives were (1) to elucidate the importance of fog as a vector for the delivery of major mineral cations and anions to a redwood forest, and (2) to trace the movement of these ions through soil water to better understand the interplay between inputs, transformations, and translocations within the atmosphere-plant-soil water system.

Methods

Our research was conducted at a coast redwood forest site in Occidental, Sonoma County, California (see Ewing, et al. 2009 for a more complete description). About 96 percent of the annual precipitation falls between October and May (hereafter rain season); the warm summer season (hereafter fog season) is a time of little rain.

Intensive measurements of fog, bulk precipitation, throughfall (TF), and soil water were made at the Sonoma site from 2003 to 2007 (see Ewing et al. 2009 for full details). Briefly, TF collectors were arrayed from the forest edge to interior in a five-band stratified random design. Two additional (bulk) collectors were placed in open grassy areas outside the forest to the southwest of the forest stand. Fog water was collected outside the forest using a passive fog collector with a plastic mesh collection surface (after Azevedo and Morgan 1974). Soil water was collected using two tension lysimeters (TL; Soil Moisture 1900 series), installed at a depth of 12 cm and set to -50 kPa, and two zero-tension lysimeters (ZTL; PVC trough installed in a soil pit face draining into a collection bottle) at a depth of 70 cm in each of the five bands inside the forest and the clearing. Soil profile descriptions were done in each band in the forest as well as in the field outside the forest at the time of lysimeter installation.

Aqueous TF, bulk, fog, and soil water samples were collected every week during the fog season and every two weeks during the rain season between July 2003 and April 2006. Sampling handling, including nitrogen analysis, is detailed in Ewing et al. (2009). Samples were analyzed at the Cary Institute (IES) for bromide, chloride, phosphate, and sulfate (Br^-, Cl^-, PO_4^{3-}, SO_4^{2-}) on a Dionex DX-500 Ion-Chromatograph and for Ca^{2+}, Mg^{2+}, K^+, and Na^+ on a Perkin-Elmer P400 inductively coupled plasma emission spectrometer, after Weathers et al. (2001). Samples with concentrations below the method detection limits were set at half the detection limit for data handling.

Seasonal mean, median, minimum, maximum, and first and third quartile ion concentrations for fog, bulk, TF, TL, and ZTL samples were calculated over all sampling periods for each season; TF and bulk ion concentrations were volume-weighted before statistical analysis. Season delineation follows Ewing et al. (2009).

Results

Nutrients and other chemicals were more concentrated in fog than rain, and fog delivered a substantial proportion of the ionic load to the forest floor even though fog constituted only two percent of the total water delivered via TF (Ewing et al. 2009). On average, ions in fog water collected outside the forest were about 10-fold (range three- to 46-fold) more concentrated than in rain water for all elements, and ions were likewise more concentrated in TF in the fog season (seven- to 22-fold) than in the rain season (*fig. 1*). However, the large amount of precipitation in the rain season meant that the highest per-day inputs of both water and ions were in the rain season rather than the fog season. However, the flux of ions to the forest floor via TF (on a per day basis, charge delivered per unit area; eq/ha) from fog was, on average, approximately 20 percent that of rain (data not shown; Ewing, et al. 2009). The dominant ions were similar for fog, rain, and TF. In all collections, Na^+, K^+, Mg^{2+}, and Ca^{2+} made up the majority of cations, while Cl^- and SO_4^{2-} dominated the anions. Bromide, PO_4^{3-}, NO_3^-, and NH_4^+ made up a relatively small proportion of the total ionic load (*fig. 1*). Nitrate was the only ion that was consistently less concentrated in TF relative to fog. Most other ions had more variable concentrations in TF than in bulk collections, and the non-acid cations in particular were more often more concentrated in TF than in collections outside the forest (*fig. 1*).

Nearly all ions delivered to the soil via TF were also found in lysimeters, but their concentrations differed as a function of lysimeter position within the soil and were variable across fog and rain seasons. As reported in Ewing et al. (2009), soil water was collected throughout the rain season in the near-surface soil (12 cm, TL), but it took approximately a month of rain before water began to drain freely and be collected in the ZTL at 70 cm. Only rain season data are reported here. The non-acid cations Ca^{2+}, Mg^{2+}, Na^+, and K^+ and the anions SO_4^{2-} and NO_3^- occurred in higher concentrations in TL than in TF in the rain season (*fig. 2*). Chloride and Br^- appeared in roughly the same concentration in soil water as in TF. Sodium was slightly greater in ZTL than TL samples (median concentration) while Ca^{2+} and K^+ were lower in ZTL than TL (*fig. 2*).

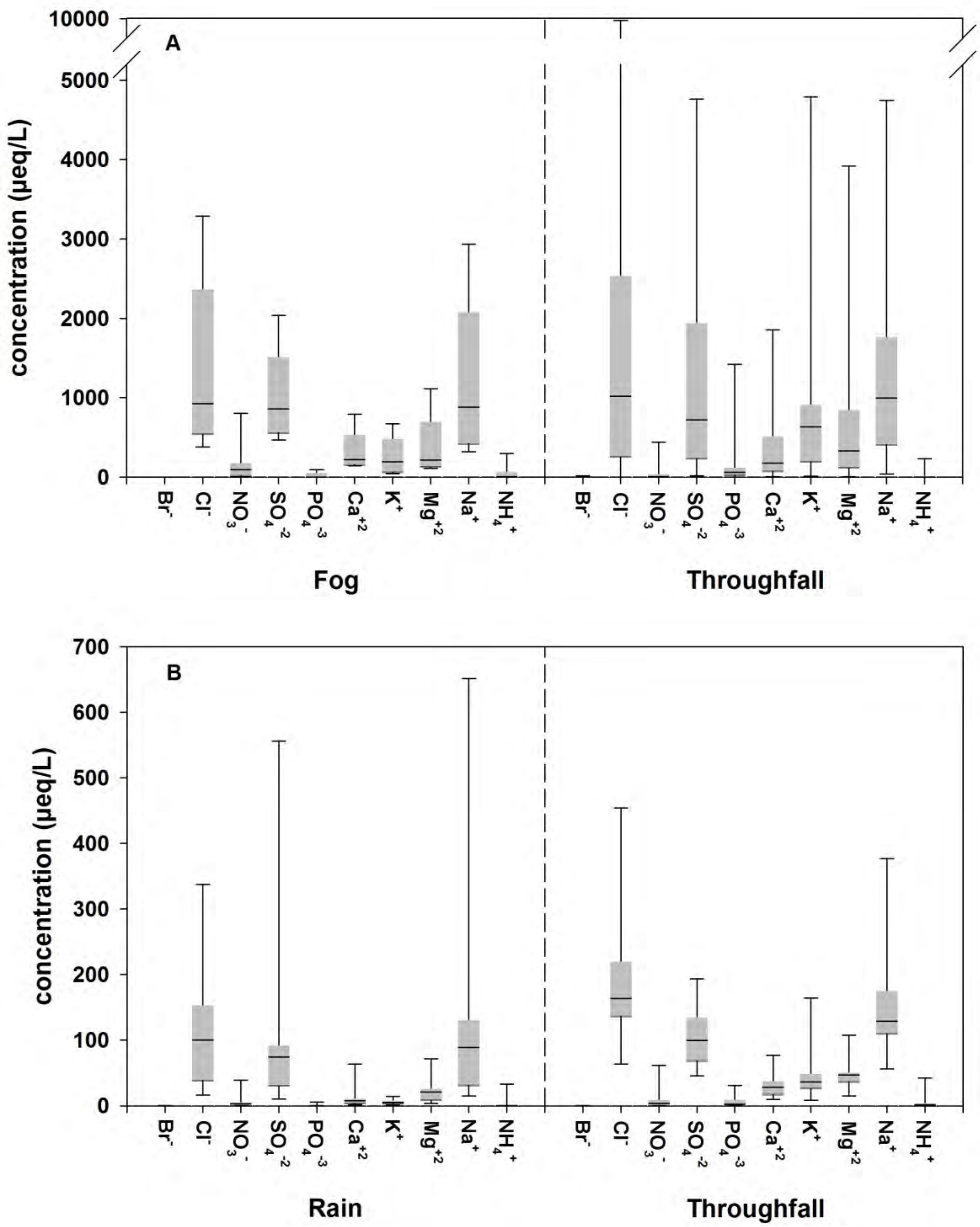

Figure 1—Ionic concentration (µeq/L) of (A) fog water and fog season throughfall, and (B) rain water and rain season throughfall collected in a redwood forest in Sonoma County, California. Values for collections at edge and interior sites were averaged before plotting; plots show variability among collections within a season. Note different concentration scales in (A) and (B).

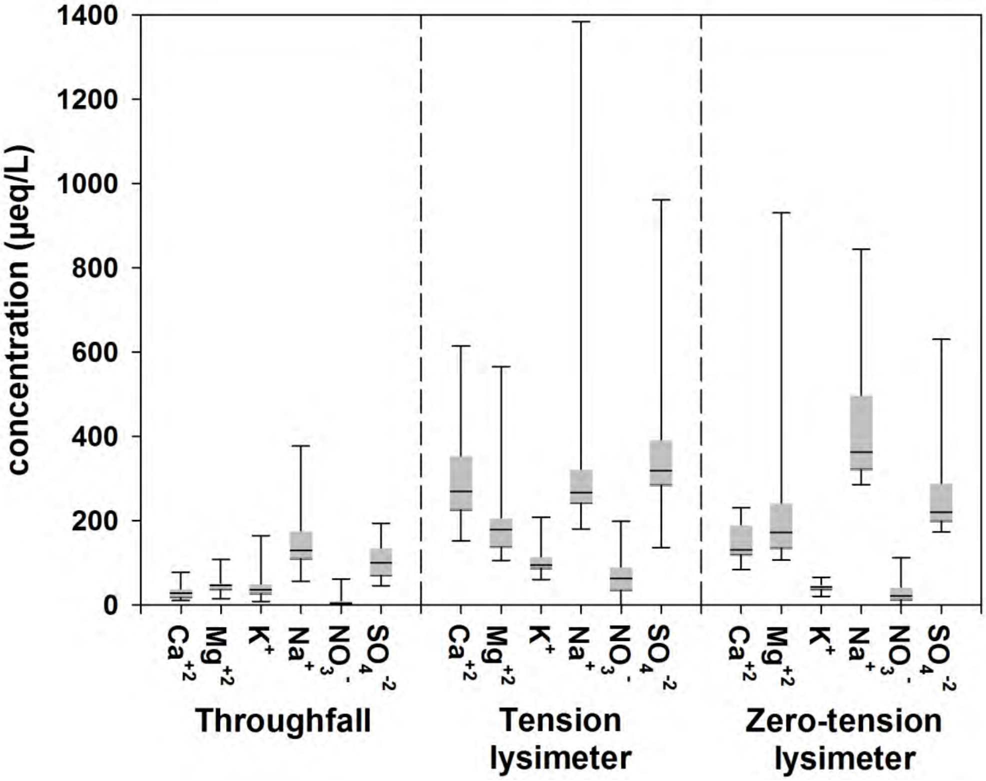

Figure 2—Concentration of ions (µeq/L) in throughfall, tension lysimeters and zero-tension lysimeters during the rain season, Sonoma County, CA. Values for collections at edge and interior sites were averaged before plotting; plots show variability among collections within a season.

Discussion

Fog is an important input vector for both water and nutrients for coast redwood forests; many cation and anion fluxes are influenced by fog despite small fog water fluxes. While water flux outside the forest in the fog season is less than one percent of the rain season flux, high redwood tree surface area results in TF flux at the forest edge that is more than five times greater than in the open, although almost no TF occurred in the forest interior (Ewing et al. 2009). Both the more concentrated nature of fog relative to rain and the change in concentration of water—usually an increase in concentration—as fog passed through the canopy supported observations made in previous studies (for example, Collett et al. 2002, Draaijers et al. 1997, Edmonds et al. 1991, Ewing et al. 2009, Weathers et al. 1988).

Where TF deposition via fog drip is high, fog may deliver a significant amount of ions to soil. Further, through root uptake of water and microbial processing, fog water input has implications for primary production and soil genesis. Root uptake would be possible most of the fog season at the edge of the forest where plant-available water, as evidenced by water found in TLs, was collected from surface soil throughout the fog season. Even in more leeward positions in the forest where fog inputs were less, trees could have access to these ions for the first quarter of the fog season when soil water potentials were high (Ewing et al. 2009). Nevertheless, tree moisture stress was lower at the windward edge of this forest (Ewing et al. 2009), and higher litterfall rates here compared to the interior of the forest indicate that primary production may be greater at the more fog-inundated windward edge of the forest (Ewing et al. 2009). These differences in inputs and tree production are reflected in soil characteristics insofar as soil organic matter concentrations and root densities remain higher deeper into the soil profile at the forest edge (data not shown).

Even in places where fog drip does not occur, fog interception by the canopy can still supply trees with a wide variety of cations and anions; trees have been shown to take in fog water through their needles (Burgess and Dawson 2004, Ewing et al. 2009, Limm et al. 2009, Simonin et al. 2009); they almost certainly are taking in ions in these highly concentrated solutions. Early work predicted that canopy interception of atmospherically-derived nutrients could be a significant source of plant nutrition in some crops (Breazeale et al. 1950, Ingham 1950). While no studies of redwood nutrition as a function of fog exist, our data suggest that redwoods have access to fog water with high concentrations of many nutrient ions.

As indicated by increasing concentrations of some ions as they pass through the soil system, some elements are clearly made available through soil weathering or mineralization in addition to deposition (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}), but their fate may depend upon their relative importance as limiting elements for plant growth and their mobility in soils. Of these, Na^{+}, which appears in greatest concentration in ZTL solutions (70 cm depth), is not needed for plant growth and may be in high concentration in the soil either as function of soil weathering or sea salt accumulation (Edmonds et al. 1991). Potassium and Ca^{2+}, on the other hand, appear in high concentration in TL (plant-available), but lower concentrations in the ZTL, suggesting that plant or microbial uptake is important. Sulfate and NO_3^-, like the non-acid cations, appear in higher concentrations in the soil solution than in TF suggesting a soil source for them as well. Since the parent material at our study site was primarily sandstone (Bradbury 2011), the most likely source for sulfur is mineralization of S from organic matter (Bailey et al. 2004). For NO_3^-, high concentrations in TL are also likely a result of organic matter decomposition and subsequent nitrification of NH_4^+. Since NO_3^- is an important nutrient for plants and microbes, it is unsurprising that nitrate is in much lower concentration in ZTL, suggesting strong conservation within the forest, as with Ca^{2+} and K^{+}.

Implications

Both fog water and the chemical constituents within fog are likely to influence biogeochemical cycling in fog- and cloud-dominated forests. Any environmental change that affects the frequency, chemistry, and height and depth of fog layers is likely to impact canopy, soil, and soil water biogeochemical transformation and fluxes (Cavelier and Goldstein 1989, Ingraham and Matthews 1995, Johnstone and Dawson 2010, Ponette-González et al. 2010). Microbial community structure and activity (Bradbury 2011) can also be directly and indirectly affected by fog inputs.

Acknowledgments

Authorship: We make no distinction in effort and contribution between the first and second authors. Co-authors are listed in order of relative contributions to this paper. S. Simkin assisted with study design, field set-up, and initial sampling; D. Herman contributed to field sampling. Funds were provided by a grant from the A.W. Mellon Foundation.

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Foliar Uptake of Fog in the Coast Redwood Ecosystem: a Novel Drought-Alleviation Strategy Shared by Most Redwood Forest Plants

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Key words: fog, understory, *Polystichum munitum*, leaf wetness, foliar uptake, drought, climate

Introduction

Fog inundates the coast redwood forests of northern California frequently during the summer months (May to September) when rainfall is largely absent (Azevedo and Morgan 1974, Byers 1953, Oberlander 1956). This maritime fog modifies otherwise warm and dry summer climate by increasing humidity, decreasing the air temperature, reducing solar radiation, and contributing water to the ecosystem through leaf wetness and fog drip (Fischer et al. 2008, Grubb and Whitmore 1966, Hutley et al. 1997, Williams et al. 2008). Fog water provides an important water subsidy for redwood forest plants that face high demand for water during the spring growing season and throughout the summer as soil moisture levels decrease (Dawson 1998, Ewing et al. 2009). In this study, we investigated how effectively fog water alleviates redwood forest plant drought stress during the summer and identified foliar uptake as a key mechanism promoting plant hydration following exposure to fog.

Foliar uptake is the absorption of water into plant crowns by leaves and stems (Rundel 1982). It occurs in diverse taxa around the world directly through the leaf cuticle (Gouvra and Gammaikopoulos 2003, Slatyer 1960, Suarez and Gloser 1982, Vaadia and Waidel 1963, Yates and Huxley 1995), specialized water-absorbing trichomes (Benzing et al. 1978, Franke 1967), or hydathode channels in the leaf epidermis (Martin and von Willert 2000). Fog, light rain, and dew may contribute little or no water to the soil profile, but may wet leaves for hours at a time (Boucher et al. 1995, Breshears et al. 2008, Ewing et al. 2009). Plants with foliar uptake capacity can absorb this water whenever their crowns are wet. In this way, foliar uptake provides an efficient mechanism to hydrate photosynthetic tissue when water is available aboveground because the water does not need to be absorbed from the soil first and transported from the roots to the plant crown second (Simonin et al. 2009).

Given the frequent occurrence of fog in the coast redwood forest during the summer and the known contribution of fog to the annual water budget of redwood forest plants (Dawson 1998), we hypothesized that foliar uptake was a common water acquisition strategy alleviating drought in the coast redwood ecosystem. Through a

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combination of glasshouse experiments using artificial fog chambers and field measurements, we studied (1) how many species in the redwood forest absorb fog water directly by foliar uptake, (2) geographical variation in the foliar uptake capacity of the dominant understory fern, *Polystichum munitum*, along the redwood forest range, (3) how foliar uptake influences the water status of *P. munitum*, and (4) how fog influences the seasonal timing and intensity of drought stress for understory plants.

Methods

Foliar uptake

To determine if foliar uptake is a shared mechanism for fog water absorption by redwood forest plants, we measured changes in leaf water content gravimetrically on cut leaves submerged in water for 3 hours to standardize water availability on the leaf surface. Single leaves were sampled from potted plants (n=7 per species) of two ferns species, *P. munitum* and *Polypodium californicum*; two shrub species, *Gaultheria shallon* and *Vaccinium ovatum*; five tree species, *Sequoia sempervirens, Pseudotsuga menziesii, Arbutus menziesii*, *Notholithocarpus densiflora*, and *Umbellularia californica*; and one herbaceous species, *Oxalis oregano*. Changes in leaf water content (LWC) were calculated as:

Equation 1

LWC = 100% × [((Mass2-(Mass4-Mass3)-Massdry))/((Mass1-Massdry))-1]

Where, Mass1 is the leaf fresh mass before submergence, Mass2 is the towel-dried leaf fresh mass after 3 hours of submergence, Mass3 is the leaf fresh mass after brief air-drying and before a method-control resubmergence, Mass4 is the towel-dried leaf fresh mass after a brief resubmergence, and Massdry is the oven-dried mass of each leaf. We used one-sample t-test to calculate if LWC increased significantly (α=0.05) after leaf wetting. Please see Limm et al. (2009) for further method detail.

To determine if the dominant redwood forest understory fern *P. munitum* exhibits intraspecific variation in foliar uptake capacity along the redwood forest range, in mid-summer we randomly sampled fronds from ten *P. munitum* crowns in Prairie Creek Redwoods State Park, Humboldt Redwoods State Park, the Angelo Coast Range Reserve, Hendy Woods State Park, the Grove of Old Trees, Roy's Redwood Preserve, and Big Basin State Park. Foliar uptake capacity was measured on the frond samples by calculating the change in LWC gravimetrically after the fronds were exposed to water misted on the adaxial leaf surface for ten hours overnight. LWC was calculated using equation 1, where the leaves were misted with water droplets instead of submerged. We used a linear regression to determine the relationship between LWC increase and foliar uptake among sampled redwood forest sites. Please see Limm and Dawson (2010) for further method detail.

Fog effects on plant water status

To determine if nocturnal fog exposure provides enough hydration to eliminate plant water deficit even when soil moisture availability is low, we exposed potted plants of *P. munitum* to four consecutive nights of artificial fog in glasshouse fog chambers and measured changes in plant water status. To isolate the effect of leaf wetting, we constrained fog exposure to the fern crowns by preventing fog drip into the soil and measured soil moisture volumetrically using sensors transecting 20 cm of the rooting zone. Over the course of the experiment, we did not add any water to the soil to allow the soil moisture to decline. We evaluated maximum plant water status at predawn after the fogging treatments ended by measuring the leaf water potential using a Scholander Pressure Chamber (PMS Instruments, Corvallis, OR) and compared fog-exposed fern hydration to control (no-fog exposure) fern hydration levels. We used non-linear regression to evaluate the relationship between leaf water potential and soil volumetric water content. Please see Limm (2009) for further method detail.

To determine if and when fog alleviates redwood forest plant drought stress, we measured seasonal trends in water status for three dominant understory species at the Grove of Old Trees, a central forest in the coast redwood ecosystem range. From 2007 to 2009, we measured the predawn leaf water potential of randomly selected plants of *S. sempervirens*, *U. californica*, and *P. munitum* distributed throughout the forest (n=15 per species). At each forest location where we measured plant water status, we quantified fog drip to the forest floor using throughfall collectors as described by Ewing et al. (2009). Please see Limm (2009) for further method detail.

Results

Foliar uptake

We found that 80 percent of the redwood forest species we studied exhibit capacity for foliar uptake. Only *U. californica* (a broadleaf tree) and *O. oregano* (a herbaceous understory plant) exhibited no measurable capacity for foliar uptake. We observed that LWC increased most (11 percent) in the dominant understory fern, *P. munitum* (*fig. 1*). Throughout the redwood forest ecosystem, the magnitude of *P. munitum* LWC increase following overnight leaf wetting varied significantly (*fig. 2*) among sites. While an 11 percent LWC increase for *P. munitum* was observed in one site (the Grove of Old Trees), *P. munitum* from most sites averaged LWC increases of approximately 5 percent. *Polystichum munitum* from one of the seven sites (Big Basin State Park) exhibited no measureable foliar uptake capacity.

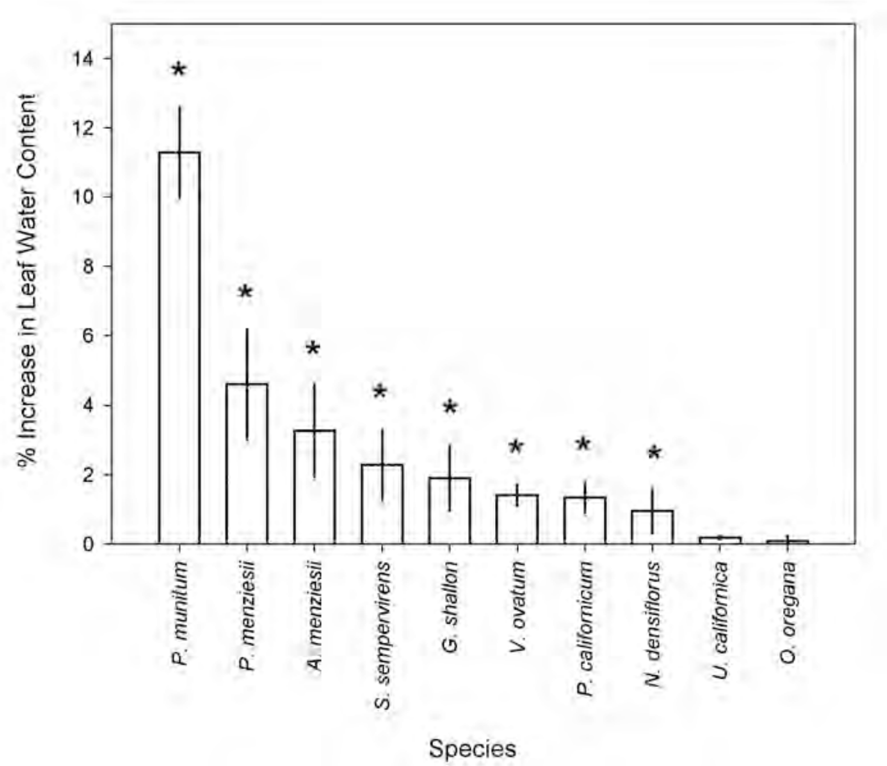

Figure 1—Foliar uptake significantly increased (*) mean (±SE) leaf water content (LWC) in the leaves of eight of the ten species studied following three hours of foliar submergence in water. This figure is modified from Limm et al. (2009).

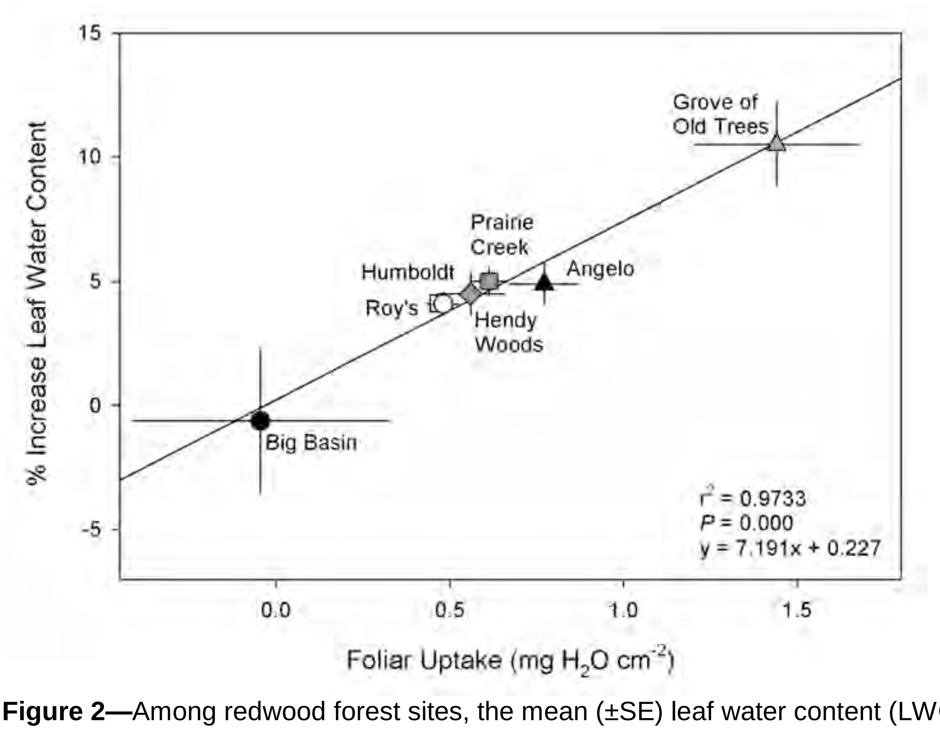

Figure 2—Among redwood forest sites, the mean (±SE) leaf water content (LWC) of *Polystichum munitum* increased with overnight foliar uptake of water misted on the frond adaxial surface. This figure is modified from Limm and Dawson (2010).

Fog effects on plant water status

In the glasshouse, both fog-exposed and control *P. munitum* plants maintained high predawn water potential (greater than -0.75 MPa) when the soil volumetric water content (VWC) was above 5 percent (*fig. 3*). The water potential of ferns exposed to four consecutive nights of leaf-wetting by artificial fog remained high even when their soil VWC dropped below 5 percent. In contrast, the water status of ferns that received no leaf wetting and only ambient, control conditions for four consecutive nights decreased sharply as the soil dried out. Without fog exposure, soil VWC below 5 percent resulted in *P. munitum* predawn water potentials decreasing to less than -2 MPa.

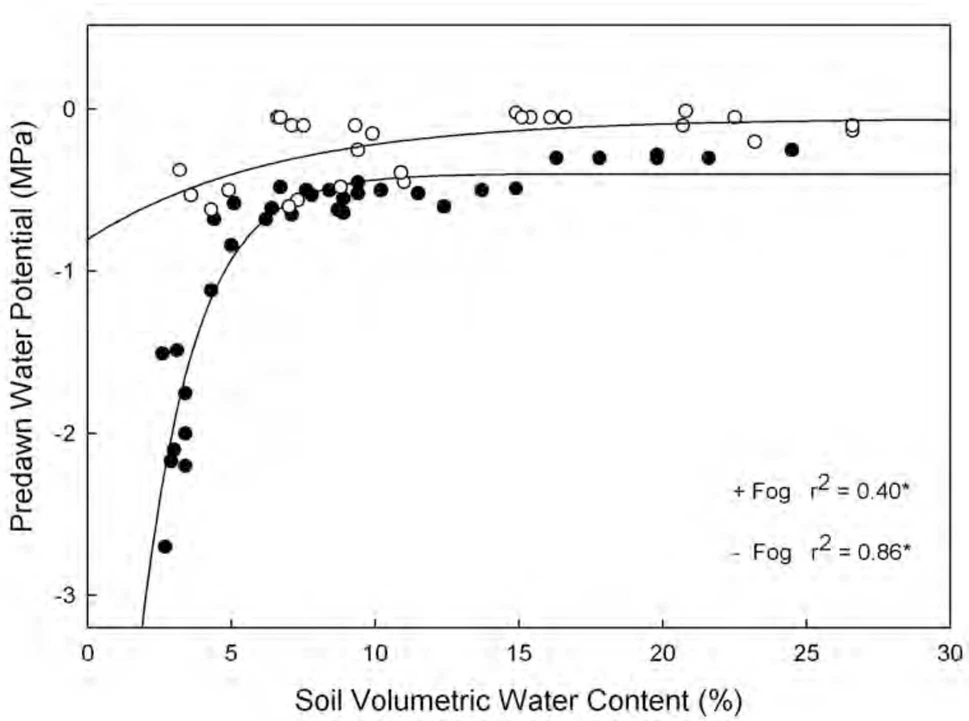

Figure 3—Predawn water potential decreased markedly in control ferns (- Fog; closed circles) that received no fog exposure during the experiment when the soil volumetric water content (VWC) dropped below 5 percent. In contrast, the predawn water potential stayed high in the fogged ferns (+ Fog; open circles) that received overnight fog exposure regardless of declining soil VWC.

At the Grove of Old Trees, the predawn water potential of all three species studied varied significantly, both seasonally and between years (*fig. 4*). In general, *S. sempervirens* maintained the highest water status in comparison to *U. californica* and *P. munitum* unless the measurements occurred during a fog event. When foggy, *P. munitum* hydration increased significantly, resulting in higher predawn water potentials than observed in either tree species. When not foggy, *P. munitum* predawn water potential was lowest. We measured significantly more water input to the forest floor as measured by the throughfall collectors during the rainy season (October to April) than during the fog season (May to September). In each year, the hydration levels of all three species increased at least once during the summer fog season when no rain fell. For all three species, the positive hydration response to fog occurred despite the disproportionately low volume of water delivered by fog drip.

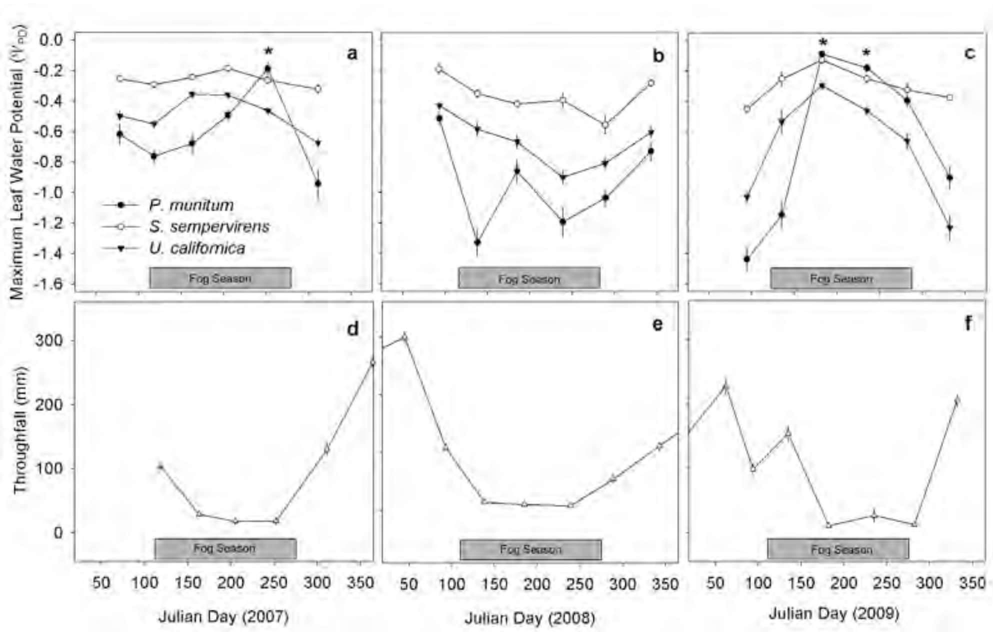

Figure 4—The predawn maximum water potentials of *S. sempervirens*, *U. californica,* and *P. munitum* changed throughout the summer fog season in (a) 2007, (b) 2008, and (c) 2009 at the Grove of Old Trees in Sonoma County. The understory received less water input during the rainless summer fog season relative to the winter rainy season in (d) 2007, (e) 2008, and (f) 2009. On foggy mornings (*), the hydration status of *P. munitum* was higher than the other species.

Discussion

Foliar uptake is a water acquisition strategy shared by most plants of the coast redwood forest (Burgess and Dawson 2004, Limm et al. 2009). While a common phenomenon among this taxonomically and morphologically diverse plant assemblage, the magnitude of potential hydration by foliar uptake varies significantly both between species and even within species. The species with foliar uptake capacity possess a wide range of leaf types, from evergreen broadleaves to fern fronds to coniferous needles and no one leaf form appears more adapted for foliar uptake than another. *Polystichum munitum* demonstrated double the capacity to increase its leaf water content following leaf wetting relative to all other species studied, but even this plant exhibits a range of foliar uptake capacity in the field. The mechanistic pathway for foliar uptake is unknown in *P. munitum* and the other redwood forest species, but we hypothesize that the leaf cuticle is the most likely pathway for water absorption as in most other species known to do foliar uptake (Limm and Dawson 2010). If the cuticle does facilitate water absorption in redwood forest plants, variation in chemical composition and architecture of the cuticle between species and populations in the ecosystem may explain the range of foliar uptake capacities we observed (Kersteins 1996, Riederer and Schreiber 2001, Shepherd and Griffiths 2006). Specifically for *P. munitum*, the greatest foliar uptake capacity was observed at the Grove of Old Trees, the windiest and foggiest site studied. Both wind and fog abrade the cuticle (Baker and Hunt 1986, Hoad et al. 1992, Pitcairn et al. 1986) and may have resulted in greater capacity of *P. munitum* to absorb water directly through its fronds in this forest.

Foliar uptake is physiologically important for coast redwood forest plants because it provides opportunities for hydration after the rainy season ends and soil water availability declines. Many fog events contribute little or no fog drip into the soil profile where plant roots access water (Ewing et al. 2009), but leaf wetting associated with short-duration or low-intensity fog events provides a subsidy that foliage can access. In the glasshouse, we observed that overnight fog exposure maintains a high *P. munitum* hydration status despite extremely low soil moisture. Simonin et al. (2009) also observed that water absorption by *S. sempervirens* crowns can decouple plants from soil water deficit. When foliar uptake readily occurs as observed in *P. munitum* and *S. sempervirens*, the potential for soil water deficit to cause plant drought stress diminishes significantly. If plants stay hydrated throughout the summer because of fog water subsidies as we observed at the Grove of Old Trees, plants can maintain higher photosynthetic rates and extend the length of their growing season.

Seasonal water deficit may increase in the coast redwood region as climate changes over the next decades (Hayhoe et al. 2004, Loarie et al. 2008). This will increase plant demand for water and dependence on fog water subsidies. Fog frequency is declining along the coast of California (Johnstone and Dawson 2010), reducing the number of days when redwood forest plants receive drought alleviation by fog exposure. The future availability of fog remains uncertain, but continued fog presence in the coast redwood forest would likely lessen the redwood forest drought stress imposed by climate change.

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A Chronosequence of Vegetation Change Following Timber Harvest in Naturally Recovering Coast Redwood (*Sequoia sempervirens*) Forests

Kristin K. Hageseth Michels[1] and Will Russell[1]

Abstract

The management of second-growth coast redwood (*Sequoia sempervirens*) forests for the purpose of restoration and ecological conservation is a growing trend. However, little is known about the long-term regenerative potential of this forest type in the absence of post-harvest management techniques such as thinning and planting. Data were collected on a chronosequence of second-growth sites (18 to 127 years) and three old-growth reference sites in order to characterize changes in stand structure and composition over time. A total of 360 plots on 18 sites with minimal post-harvest treatment were sampled in the central range of the coast redwood forest in California in order to compare stand conditions between post-harvest age groups.

One-way ANOVA with Bonferroni post-hoc analyses indicated that stand density, canopy cover, and species richness approached old-growth conditions within 40 to 80 years. Total basal area and the mean maximum diameter of *S. sempervirens* stems continued to increase up to 127 years. Cover of non-native species declined with stand age to the extent that no non-native species were recorded in stands older than 60 years. The cover of old-growth associated understory species was highest on the oldest second-growth stands and in some cases reached levels statistically equivalent to the old-growth references sites between 40 and 100 years. Results suggest that coast redwood forests are highly resilient to human disturbance and will recover naturally over time in the absence of post harvest management.

Key words: chronosequence, coast redwood, natural recovery, second-growth, *Sequoia sempervirens*

Introduction

Timber harvest practices have significant impacts on forest communities. The removal of canopy species exposes the forest floor to increased levels of solar radiation (Rivas-Ederer and Kjeldsen 1998, Russell and Jones 2001), facilitating the recruitment of opportunistic and exotic species (Rivas-Ederer and Kjeldsen 1998). Logging practices also alter soil conditions through compaction (Corns 1988, Stone and Wallace 1998), and reduce nitrogen levels in previous logged stands (Jussy et al. 2004), particularly near skid roads (Ebrecht and Schmidt 2003). Forest clear-cutting has been categorized as a major forest disturbance (Oliver 1981).

Oliver (1981) provides a framework for describing the recovery processes of overstory and understory species following stand level disturbance. Post-disturbance development involves four stages: stand initiation, the invasion of new stems on

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released sites; stem exclusion, the vertical stratification and competition among existing and new stems; understory reinitiation, the development of favorable conditions for understory species; and old-growth development, the increase in canopy variability and understory development (Oliver 1981). These processes have rarely been studied in naturally recovering understory and overstory vegetative communities in coast redwood (*Sequoia sempervirens* (D. Don) Endl.) forests after clear-cut harvest. The purpose of this study was to determine how coast redwood communities respond to this major disturbance over time.

Methods

This study was conducted in the central range of the coast redwood forest. Study sites were primarily located in the Big River watershed, consisting of more than 2,968 ha of regenerating coast redwood forest (California Department of Parks and Recreation 2006) in Mendocino County, California. Much of the Big River watershed was managed as industrial timberland prior to its purchase by the Mendocino Land Trust in 2002, and was subsequently transferred to the California State Park system as the Big River Unit of Mendocino Headlands State Park (*fig. 1*). The Big River watershed was an ideal location for this study due to the presence of recovering redwood stands between 15 and 127 years old that had received minimal post-harvest manipulation. Three unharvested old-growth reference sites, the only sizable remaining old growth stands in Mendocino County, were also included in this study: the 49 ha Russell Unit (the largest remaining old-growth redwood stand on the Mendocino coast), Montgomery Woods State Natural Reserve (462 ha), and Hendy Woods State Park (342 ha).

Within the three study sites selected in each of the five post-harvest age classes (0 to 20, 21 to 40, 41 to 60, 81 to 100, and 101 to 130 years) and the three unharvested old-growth reference sites, 60 plots were sampled in each site (360 plots total). Sites were selected using detailed timber harvest and land management history data on a GIS platform (Rutland 2002). Site selection criteria included stands that were dominated by *S. sempervirens*, previously clear-cut, large enough for adequate sampling without edge effects (Russell and Jones 2001), and received no post-harvest management such as seeding, thinning, or planting. The post-harvest age class of 61 to 80 years was not sampled as no sites in that age range existed that met these criteria. Physical variation between sites was minimized to the extent possible.

Twenty, 20 m diameter (0.031 ha), circular sample plots were randomly selected within each of the 18 study sites using ArcGIS™, and located using a handheld GPS receiver. Each sample plot was located a minimum of 20 m from adjacent plots, 10 m from special habitats such as riparian areas and rock outcroppings, and 200 m from adjacent age class boundaries and main access roads. Plot size and sampling intensity were determined through a pilot study using the species-area curve method (Cain 1938). Data were collected on each plot to describe the composition and structure of the existing stand as well as physical characteristics of the site. Slope, aspect, dominant canopy species, and percent canopy cover (measured with a spherical crown densiometer) were recorded at the center of each plot. The occurrence and abundance of each tree species was recorded within each 20 m diameter sample plot, as well as the diameter at breast height (dbh) of all individuals greater than 1 m in height. Percent cover was determined for all understory species using ocular estimates. All tree species and understory species were identified following nomenclature from the Jepson Manual (Hickman 1993).

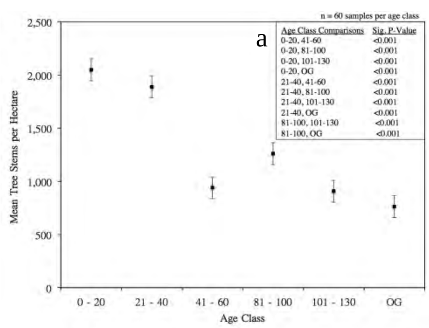
Statistical analyses were conducted in Aabel™. One-Way ANOVA were used to test differences between recovering stands and old-growth sites. Data were tested for normality using a skewness test and transformed using the natural log ($\log_e$) transformation if skewed. Additional post-hoc analyses were conducted using the Bonferroni test for pair-wise differences between groups.

Results

Tree species development

Stand structure and composition varied significantly between age-classes. One-Way ANOVA indicated that tree density ($F(5, 360)=27.599$, $p<0.001$) (*fig. 1a*) and richness of tree species ($F(5, 360)=17.121$, $p<0.001$) (*fig. 1b*) declined significantly from younger to older stands, but did not statistically differ from old growth stands (hereafter referred to as 'old-growth equivalence') at 41 to 60 years after disturbance. Approaching levels of old-growth density and richness within 41 to 60 years suggests stands of this age class have successfully reached the stem exclusion phase of stand recovery as described by Oliver (1981).

Canopy cover exhibited an inverse pattern than that of tree density and tree richness, as percent canopy cover increased across age classes ($F(5, 360)=8.470$, $p<0.001$) (*fig. 1c*). Canopy cover did not reach old-growth equivalence until stands were 81 to 100 years old. Although there was a significant increase over time, percent canopy cover exhibited minimal variation between age classes, ranging from approximately 76 percent in the youngest age class to approximately 86 percent in the old-growth stands. These results illustrate a dominant canopy is present throughout all stages of post-disturbance development in this ecosystem.



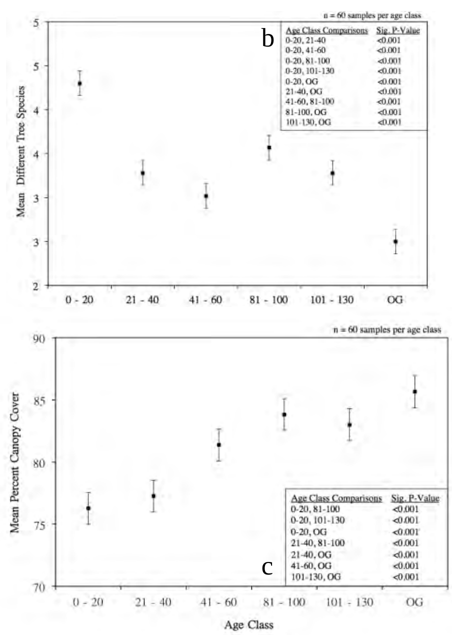


Figure 1—One-Way ANOVA between (a) tree density (stems per hectare), (b) tree richness, (c) canopy cover, and age class with Bonferroni post hoc analyses in recovering and old-growth stands.

Similar to canopy cover, the combined basal area of all tree species ($F(5, 360)=167.158$, $p<0.001$) and the basal area of the most dominant tree species, *S. sempervirens* ($F(5, 360)=81.207$, $p<0.001$), increased significantly with stand age (*fig. 2a*). Old-growth stands encompassed the highest combined basal area than any other age group. In contrast to canopy cover, however, total basal area and *S. sempervirens* basal area did not resemble areas typical of old-growth stands within the chronosequence of stands sampled. Observation of this pattern of development is limited by the ages of recovering stands sampled in the chronosequence (15 and 127 years) compared to the old-growth stands (>1,500 years). Although total basal area and *S. sempervirens* basal area did not statistically reach old-growth stands, basal area consistently increased throughout recovering stands and illustrated initial movement towards old-growth levels at 81 to 100 years. Basal area development, also indicated movement through the stem exclusion phase of recovery.

The variability in dominance among the three most predominant tree species, *Notholithocarpus densiflorus* (Hook. & Arn.) Rehder, *Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*, and *S. sempervirens*, fluctuated throughout the chronosequence and compared to old-growth stands (*fig. 2b*). Both *N. densiflorus* and *P. menziesii* var. *menziesii* were dominant in the younger sites, however after stands reached 41 to 60 years, this variability in dominance decreased, favoring *S. sempervirens* in the older age classes. Due to the regenerative capability and growth pattern of *S. sempervirens*, the older sites and old-growth stands exhibited higher levels of dominance by this critical old-growth species and less variability in canopy dominance.

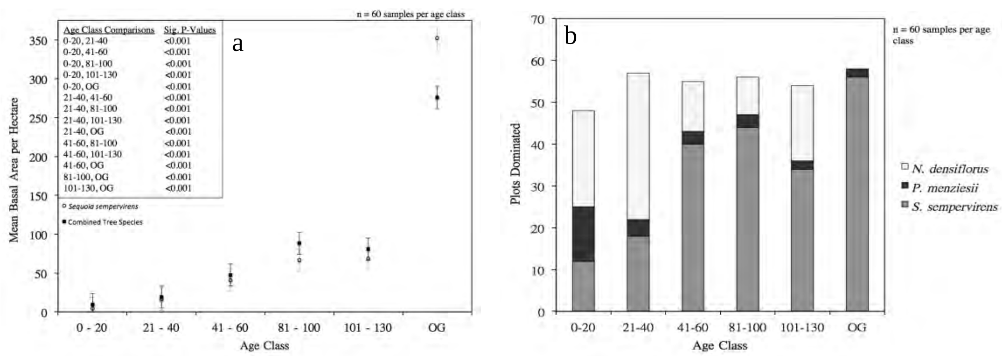


Figure 2—(a) One-Way ANOVA between total dominance and age class in recovering and old-growth forests and *Sequoia sempervirens* dominance and age class in recovering and old-growth forests and (b) number of plots dominated by the three most dominant tree species, *Notholithocarpus densiflorus, Pseudotsuga menziesii,* and *Sequoia sempervirens,* with Bonferroni post hoc analyses in each recovering age class and old-growth forests.

Understory species development

The understory community also significantly varied with stand age. Percent cover of understory species initially decreased immediately after harvest, reaching the lowest cover within 41 to 60 years, followed by a slight increase ($F(5, 360)=52.140$, $p<0.001$), illustrating a moderate bimodal pattern of development (*fig. 3a*). Understory dominance was closest to old-growth levels immediately after harvest, likely due to non-native species that easily colonize recently disturbed habitats. Understory percent cover did not achieve old-growth equivalence in the recovering stands. However, a subsequent increase of understory cover did occur in the 81 to 100 year age group, suggesting the understory reinitiation phase may be suitable after at least 80 years of recovery.

In contrast, the richness of understory species declined to old-growth levels after only a few decades of post-harvest recovery ($F(5, 360)=4.261$, $p<0.001$) (*fig. 3b*). As soon as 21 to 40 years after harvest, total understory species richness reached levels equivalent to old-growth stands. Compared to tree species richness, recovery of understory species richness progressed much quicker. Understory species richness achieved old-growth equivalence after at least 21 years of development, supporting the understory reinitiation that arises in subsequent age classes.

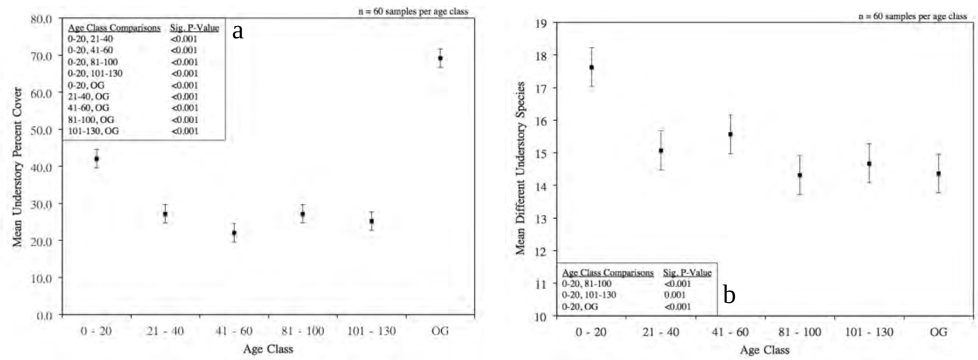


Figure 3—One-Way ANOVA between (a) understory species percent cover, (b) understory species richness, and age class with Bonferroni post hoc analyses in recovering and old-growth forests.

The recovery of individual associated understory species varied. The most dominant species in all age classes including old-growth, *Oxalis oregana* Nutt., steadily increased in second-growth stands, exhibiting the highest value within the recovering stands in the 101-130 age group, but did not statistically reach old-growth values ($F(5, 360)=11.937$, $p<0.001$) (*fig. 4a*). Although *O. oregana* did not statistically indicate full recovery during the chronosequence, maximum cover values approached old-growth within 101 to 130 years (*fig. 4b*).

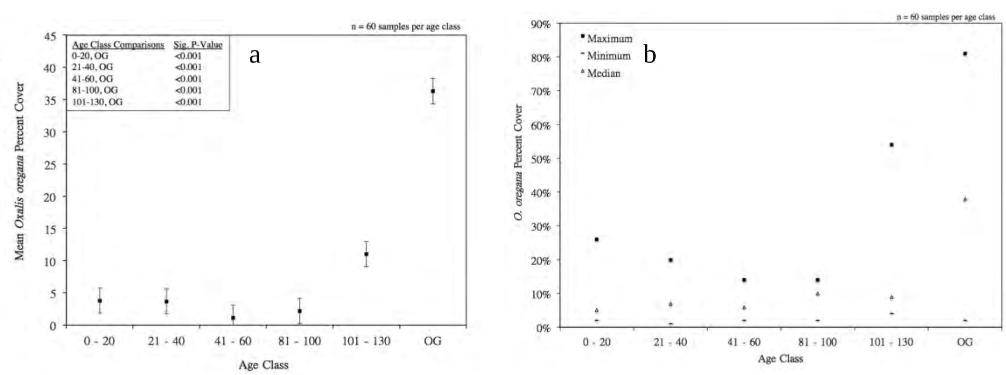


Figure 4—(a) One-Way ANOVA between *Oxalis oregana* percent cover and age class in recovering and old-growth forests and (b) maximum, minimum, and median values for *O. oregana* percent cover with Bonferroni post hoc analyses in each recovering age class and old-growth forests.

Trillium ovatum Pursh increased significantly from the youngest to the oldest age-classes ($F(5, 360)=16.247$, $p<0.001$) (*fig. 5a*). *T. ovatum* achieved old-growth equivalence during the 81 to 100 age class. Although a decrease subsequently occurred, this may in part be due to an increase in *O. oregana* dominance.

Non-native species cover was significantly higher in younger recovering stands compared to older age classes ($F(5, 360)=16.328$, $p<0.001$) (*fig. 5b*) and achieved old-growth equivalence in 81 to 100 years. Although exotic species were present in early age groups following harvest, non-native species were completely absent in second-growth stands older than 60 years and old-growth stands.

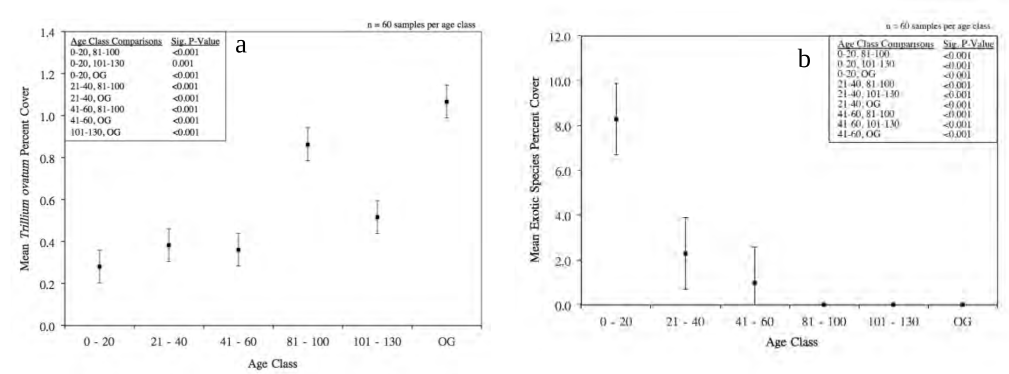


Figure 5—One-Way ANOVA between (a) *Trillium ovatum* percent cover, (b) exotic species percent cover, and age class with Bonferroni post hoc analyses in recovering and old-growth forests.

Discussion

Results from the unmanaged stands sampled may be described utilizing the Oliver (1981) recovery model following major stand-scale disturbances. Stand characteristics including tree species density, tree species richness, *S. sempervirens* basal area, and total combined basal area of all tree species indicate the stem exclusion phase was reached within 41 to 80 years, supporting understory species richness to approach old-growth status shortly after disturbance. Development during this phase allowed for further understory reinitiation within 81 to 130 years as illustrated by the increased cover of native redwood associate species, including *Trillium ovatum* and *Oxalis oregana*, and die out of non-native species. Understory development continued in the oldest age class demonstrated by the marked increase in cover and dominance of *O. oregana*, the most prevalent understory species in all age classes.

In this study, indicators of stand heterogeneity, such as tree species richness, were highest immediately after harvest, but declined in subsequent age groups, most likely due to the increased availability of resources immediately after harvest, such as greater soil fertility, solar radiation, and habitable landscapes (Fraterrigo et al. 2006, Tilman 1985). Fritz (1945) found that total tree density decreased after harvest within a permanently established naturally-recovering one-acre sample plot. Busing and Fujimori (2002, 2005) also illustrated high levels of *S. sempervirens* dominance in old-growth forests, which resulted in a decrease of tree species richness and diversity in stands. This research illustrates that tree species density and tree species richness are capable of stem exclusion, successfully reaching old-growth levels, without any active management of stands.

Increased basal area is characteristic of old-growth forests (Busing and Fujimori 2002, 2005) and can be attributed to the dramatic growth of the dominant species, *S. sempervirens* (Fritz 1945). As has been found in previous research, *S. sempervirens* in old-growth stands had the highest basal area (Busing and Fujimori 2002, 2005; Fujimori 1977, Russell and Jones 2001). In this study, basal area development of *S. sempervirens* contributed to the stem exclusion of other dominant tree species. Due to the regenerative capability, growth pattern, and longevity of *S. sempervirens*, additional time may be necessary for basal area to fully resemble old-growth levels in naturally recovering redwood communities. However, the older sites and old-growth stands exhibited higher levels of dominance by principal old-growth species, resulting in the understory initiation phase described by Oliver (1981).

Old-growth forests exhibited the highest levels of canopy cover, but remained isolated to patches, due to an increase in canopy complexity and insularity (Loya and Jules 2007). Although younger stands were highest in tree density, the intermediate stages illustrated old-growth levels of canopy cover. This suggests that canopy cover began to move toward old-growth levels in the intermediate stages of recovery, allowing for understory development to take place. These developing stages produced similar levels of understory species as old-growth forests. As with density, the cover of these naturally recovering stands approached that of old-growth stands without active management. Future research should consider the effect of actively removing canopy species within these redwood communities and the subsequent effect on the development of native plant communities.

Loya and Jules (2007) also found understory cover was highest in early stages of development following clearcut harvest, although the stem exclusion patterns occurring within the 41 to 80 year range in this study resulted in understory reinitiation and subsequent patch dominance of old-growth associate species, such as *Oxalis oregana*, in the 81 to 130 year stands. In addition, an important old-growth associate species, *Trillium ovatum* did reach old-growth equivalence during this understory reinitiation phase. Heavy soil compaction resulting from clear-cut harvest may not have allowed for adequate recovery of other understory species on the chronosequence sampled. Additional research, including soil analyses, is likely required to fully evaluate recovery patterns of understory species approaching old-growth levels. Although understory species richness was low in old-growth forests in this study and in previous research (Busing and Fujimori 2002, Loya and Jules 2007), richness reached old-growth equivalence within a relatively short time frame following disturbance. Richness of understory species was therefore also capable of recovery in second-growth stands without active management.

In the unmanaged, naturally recovering redwood forest communities sampled, non-native species may establish in younger second-growth stands, but could eventually give way to native species over time. This supports previous research by Loya and Jules (2007) that also found few exotic species in old-growth stands. However, the year of introduction, life history, distribution, and physiology of exotic species must also be considered (Russell and Hageseth Michels 2010). Nonetheless, the most dominant exotic species within the study area, *Cortaderia selloana* (Schult. & Schult. f.) Asch. & Graebn. (pampas grass), was first introduced into California in 1848 (Lambrinos 2001), before the initial harvest of the oldest stands sampled.

Recommendations

The results of this study illustrate that a variety of community parameters and individual species followed Oliver's (1981) model of natural forest recovery without active management. Even though sites were highly fragmented due to various harvest time frames and land use histories, older sites were relatively undisturbed, even from trail use or recreation activities, and existed in fairly remote areas distant from local populations. In addition, the Russell Unit is the least disturbed remaining old-growth stand on the Mendocino coast and is the most representative of old-growth communities of this region. These recovering stands have the potential to eventually resemble old-growth forests and retain developing old-growth characteristics.

While the results from this study indicate that second-growth coast redwood stands are capable of natural recovery, some small-scale actions could enhance these stands. Continued presence or use of prominent logging roads would likely increase erosion, negatively affect edaphic health, and damage adjacent aquatic habitats. Major roads should be further decommissioned to a natural setting to mitigate potential negative effects. Due to the sensitivity of these stands, any recovery activities should take place with minimal impact.

Limitations of this study could be clarified through additional research. This study focused on the vegetative change in redwood plant communities following clear-cut harvest on a chronosequence, although several other factors beyond the scope of this study likely contributed to the development of these forest communities. Future research on the stands sampled in this study could include assessing variables related to soil conditions, fungal associations, fauna biodiversity, or long-term vegetative monitoring. The recovering redwood stands of Mendocino County should continue as a focal point for forest ecologists.

A detailed understanding of community development is necessary to manage recovering coast redwood forests. A management paradigm that focuses on specific tree species does not adequately assess timber harvest effects on the entire redwood community, which may hinder recovery. The remaining redwood forests are largely fragmented, increasing the importance of the overall continuity of the redwood range. If allowed sufficient time to recover after harvest, developing second-growth coast redwood communities could increasingly resemble old-growth redwood stands.

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Accounting for Variation in Root Wood Density and Percent Carbon in Belowground Carbon Estimates

Brandon H. Namm[1] and John-Pascal Berrill[1]

Abstract

Little is known about belowground biomass and carbon in tanoak. Although tanoaks rarely provide merchantable wood, an assessment of belowground carbon loss due to tanoak removal and Sudden Oak Death will inform conservation and management decisions in redwood-tanoak ecosystems.

The carbon content of woody biomass is a function of density and the proportion of carbon in dry biomass. Whole-tree basic wood density or specific gravity estimates are often available to facilitate calculation of forest biomass, and contemporary carbon analyses generally assume that carbon comprises 50 percent of the dry biomass for the whole tree. Less is known about root wood density and carbon, or variations within root systems. Quantifying root wood density and carbon content changes along the length of a root will enable more accurate estimation of belowground carbon, and support development of equations predicting carbon from easily measurable aboveground variables, such as dbh.

To analyze root density and carbon content at different locations within root systems, tanoak trees were first removed using an excavator. Root wood samples were taken from four locations within the root system: within the stump (aboveground), within the lignotuber, at the start of the root (adjacent to the lignotuber) and at the end of the root (near tip).

We did not detect significant differences in root wood density between samples collected at different distances from the stem or between different sizes of roots. Percent carbon was highest at the sample farthest from the lignotuber, while samples from the other locations were not statistically different. Root carbon also varied among root systems sampled.

Key words: belowground, biomass, carbon, lignotuber, *Notholithocarpus densiflorus*, roots, tanoak, wood density

Introduction

Due to challenges in sampling and measurement of tree roots, little is known about root system biomass and carbon (C) content (Böhm 1979). The C content of roots is a function of root biomass and the proportion of carbon in dry root biomass. Accurate estimates of root biomass are difficult to obtain. Any variability in root wood density and C content within and between root systems will introduce uncertainty into estimates of belowground C. An understanding of patterns of belowground density and C within root systems is needed to guide belowground sampling methods and support calculations or predictions of belowground C.

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One method of sampling roots (Achat et al. 2008) involves digging a trench next to the root system of interest and mounting a grid frame on the trench wall to count or map roots. This method reveals gradients in root distribution with depth, but ignores root length. Extracting entire roots gives root volume and biomass, but requires care further from the stem where roots narrow, weaken, and become intertwined with roots from neighboring trees. Root volume equations can be fitted to diameter and length data. Volume estimates can be converted to biomass and carbon estimates once patterns of root density and carbon content have been described.

Aboveground biomass and C allocation patterns have been described for many species. Variations in light interception as well as interspecific differences in wood density are major factors determining variations in growth among trees (King et al. 2005). Wood density is positively correlated with crown area and depth and can be a reasonable predictor of light demand of some species (Aiba and Nakashizuka 2009, Nock et al. 2009). Wood density also generally decreases along the length of the stem (Correia et al. 2010, Espinoza et al. 2004, Park et al. 2009, Paul et al. 2008).

Radial patterns of growth (growth from pith to bark) are different in stems branches and roots, supporting the hypothesis that wood in different positions in a tree has different physiological or mechanical roles (Peterson et al. 2007). Tracheid length, which is correlated with wood density, in roots is relatively constant from pith to bark, while tracheid length varies within boles and branches (Peterson et al. 2007). Although branch specific gravity is not affected by its distance from the bole (Woodcock and Shier 2003), the pattern of root wood compression strength decreasing with distance from the stem (Stokes and Mattheck 1996) may also hold for root wood density.

Roots are subject to different conditions and stresses than tree stems and branches, and therefore may exhibit different trends in wood density and C content. The C content of woody biomass has been shown to range from approximately 47 to 59 percent (Arola 1976, Fryling 1966, Hollinger et al. 1993, Pingrey 1979, Ragland 1991, Tillman 1981), but is generally assumed to equal 50 percent for both wood and bark (Cooper 1983, Dewar and Cannell 1992, Hollinger et al. 1993, Matthews 1993, Sedjo 1989, Thuille et al. 2000, Wenzl 1970). However, bark typically contains less carbon than wood relative to its density (Quilho and Pereira 2001).

Accurate wood density estimates depend on accurate volume measurements. For branches, volume can be estimated from path distances between set diameter intervals and summed for each branch (Sillett et al. 2010). The same method can be applied to roots, giving root taper information. The alternative volume estimation method of displacement (immersion in liquid) does not provide root dimension and taper information, which can be used to develop root volume and taper equations. These models would allow for subsampling during excavation and prediction of the biomass in missing/broken roots from diameter at the point of breakage.

The objective of this study was to quantify wood density and percent C at different locations within tanoak (*Notholithocarpus densiflorus* (Hook. and Arn.) Manos, Cannon and S.H. Oh) root systems. In addition to measuring the percent C, accounting for wood density gradients is likely to improve the accuracy of belowground biomass and C estimates (Nock et al. 2009). We hypothesized that (1) roots of larger diameter have both denser wood and a greater proportion of C and (2) wood within the lignotuber is most dense and holds proportionally the most C while roots decrease in wood density and proportion of C with increasing distance from the lignotuber. Root bark volume and C content were also examined. Results will support efforts to quantify belowground C storage and sequestration in the redwood region.

Methods

Site description

Root systems were excavated at the L.W. Schatz Demonstration Tree Farm, near Maple Creek, Humboldt County, California. The 148 ha tree farm is located 40 km inland and extends from N40º46’49” to 40º45’56” and W123º52’21” to 123°51’32”. Average annual precipitation is approximately 120 cm with the majority falling as rain between the months of November and March.

Before clearcutting in the 1950s, the site was occupied predominantly by old-growth Douglas-fir (*Pseudotsuga menziesii*) forest with tanoak and California bay (*Umbellularia californica*). Species composition on the property now consists primarily of Douglas-fir, grand fir (*Abies grandis*), and tanoak mixtures, and pure stands of *Alnus rubra* (West 2007).

Excavation and measurements

Three second-growth tanoak trees were selected for sampling. Tree height, live crown base height, diameter at breast height (dbh), and diameter at the base of the tree were measured. An excavator was used to create a 30 cm wide trench at a distance of 1 m surrounding each tanoak to the depth where roots were no longer visible (approximately 1 m) severing the lateral roots traversing the trench. A rope was attached 5 to 10 m up the bole and each tree was pulled over using either a 4WD vehicle or a hand winch. We measured the diameter of the severed root ends and their distances from the lignotuber to allow for future prediction of mass lost during excavation (Niiyama et al. 2008).

Root samples were extracted at four locations within the root systems. Two pith-to bark wood cores were collected from each stump and lignotuber. Lateral roots oriented in the four cardinal directions around the three sample trees were excavated by hand for density and C analysis. Two short sections of root were cut from each lateral root: one adjacent to the excavated trench approximately 100 to150 cm from the stem, and a section of the same root at the narrow end farthest from the stem (*fig. 1*).

The green volume of these samples was estimated by measuring their diameter and length, and assuming the volume of each section was best represented by a truncated cone. Samples were dried at 65 °C and weighed to obtain dry mass. Much of the water was removed by drying biomass at 65 °C but drying at 100 °C was required to remove water bound within the cell walls (Wiemann and Williamson 2010). Different portions of the same roots used for C analysis were dried at 65 °C and weighed, and then dried further at 100 °C and re-weighed for comparison.

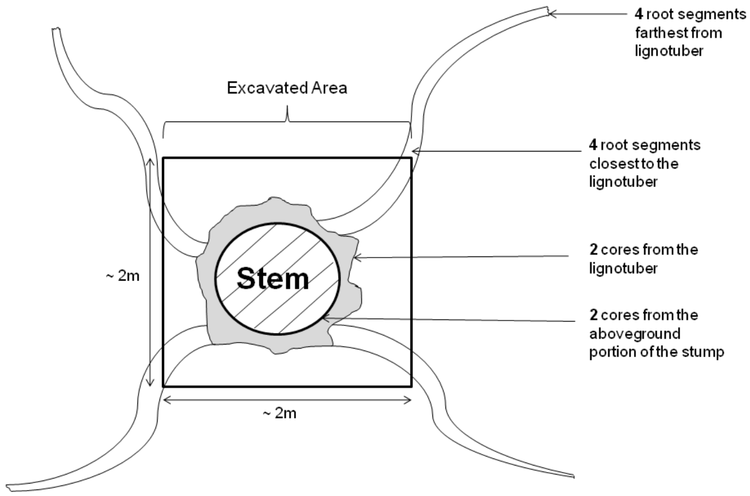

Figure 1—Locations and quantities of C samples within each of the three individual tanoaks excavated at the L.W. Schatz Demonstration Tree Farm.

Root bark samples collected throughout the three root systems were displaced in water to obtain estimates of bark volume. The estimates of volume and dried bark weight gave estimates of bark density. A subsample of root bark with known mass was removed and analyzed for percent C. A caliper was used to measure the outside diameter of roots including the bark and the diameter inside bark of roots with varying diameters giving estimates of the cross sectional area and volume of bark in lateral roots.

Carbon/nitrogen analysis

Percent C in each stump, lignotuber, root and bark sample was determined using a dynamic flash combustion system coupled with thermal conductivity/ IR detection (Thermo-Finnigan Flash EA 1112). Samples were oxidized by flash combustion which converted all organic and inorganic substances into combustion gases (N_2, NO_x, CO_2, and H_2O). The method has a detection limit of approximately 0.1 percent for C (http://anlab.ucdavis.edu/analyses/plant/sop522[2], AOAC Official Method 927.43).

Statistical analysis

We constructed a nested model to test for differences in wood density and percent C at the different locations within the root system across all sample trees. Tukey-Kramer’s multiple comparison test was used to identify differences among the sample locations and the acceptable level of error was adjusted for multiple comparisons ($\alpha = 0.008$). One-way ANOVA was used to identify differences in root wood density and percent C within and between trees. Additionally, density and percent C were regressed against the sample locations’ path lengths from the lignotuber using Minitab 16 Statistical Software (Minitab 2010).

[2] Total nitrogen and carbon – combustion method. University of California, Davis, analytical lab.

Results

The sample trees were similar in height, but had different stem diameters and crown ratios and occupied different positions within the canopy (*table 1*).

Table 1—*Aboveground sample tree data for tanoak at L.W. Schatz Tree Farm.*

Tree	Height (m)	Live crown ratio	Dbh (cm)	Diameter at base (cm)	Crown position
1	21.1	0.81	33.5	39.6	open grown/dominant
2	18.3	0.27	34.8	42.1	codominant
3	20.7	0.59	52.4	60.1	codominant

Mean wood density for belowground components (lignotuber, lateral roots) was 590.3 kg/m^3 and varied between 565 and 621 kg/m^3 (*table 2*). The nested model with trees as random factors indicated that no significant difference in root wood density among trees ($p = 0.508$), or among sample locations ($p = 0.334$) was detected (*fig. 2A*). Wood density did not change when regressed against root diameter ($F = 1.16$, $p = 0.260$) nor path distance from the lignotuber ($F = 0.04$, $p = 0.852$) but was highly variable among trees (coefficient of variation = 0.139).

The grand mean of the C samples across the three trees was 49.5 percent. The nested model indicated that no significant differences in percent C among trees were detected ($p = 0.097$), but percent C differed among sample locations ($p < 0.0001$). Kramer-Tukey multiple comparison tests indicated that sections of roots farthest from the lignotuber had a significantly greater C content than the other sample locations while the aboveground stump, lignotuber and the samples nearest the lignotuber were not significantly different in percent C (*fig. 2B*). Mean percent C ranged from 48.6 to 50.6 percent (*table 2*).

Percent C did not change when regressed against diameter ($p = 0.703$). A nested model of belowground percent C (stump C not included) with individual trees as a random effect indicated that percent C increased with path distance from the lignotuber ($p = 0.003$) (*fig. 3*). The model explained 45 percent of the variation in belowground percent C which varied within and between sample trees (coefficient of variation among trees = 0.02).

Mean root mass decreased 5.5 percent when the drying temperature increased from 65 °C to 100 °C. The proportion of root volume comprised of bark was consistent for all root diameters (mean proportion bark volume = 22.1 percent) and bark consisted of 48.6 percent C.

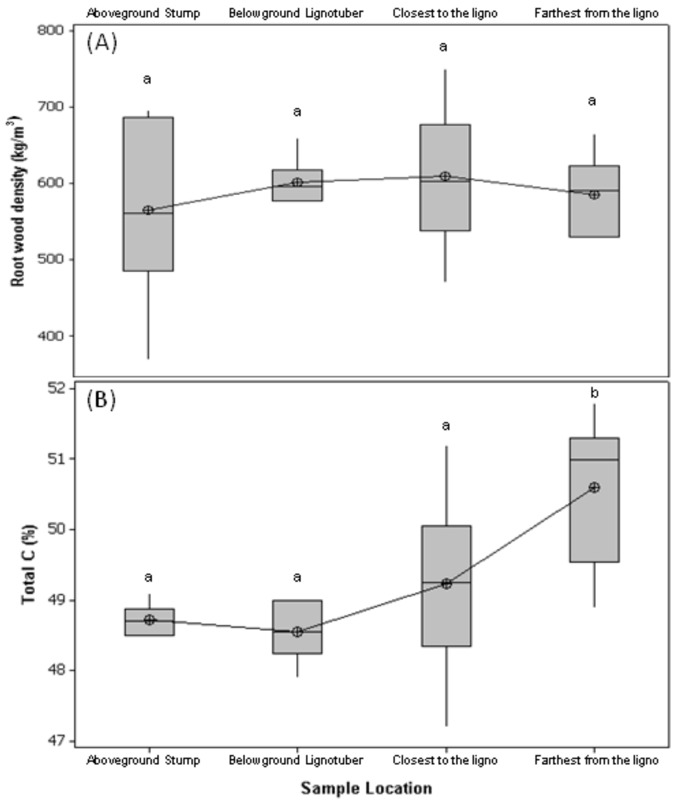

Figure 2—Box plot of (A) wood density and (B) percent C at the different sample locations in the root system. Different letters indicate significant differences in percent C and wood density. Lines connected by location means.

Table 2—*Root wood density and percent C for the three sample trees. Differences in location means detected by Tukey-Kramer multiple comparison tests indicated by different letters. Acceptable levels of error adjusted for multiple comparisons (α = 0.008).*

		Wood Density (kg/m^3)		
Tree	stump	lignotuber	near	far
1	635.7	628.3	648.7	507.8
2	529.5	590.8	614.2	526.5
3	592.3	585.7	564.9	662.9
mean	585.9[a]	601.6[a]	621.2[a]	565.7[a]
st. dev.	52.4	30.7	87.0	105.5
max	664.9	659.7	749.4	696.5
min	528.4	575.6	471.2	370.0
		Total C (%)		
Tree	stump	lignotuber	near	far
1	49.0	49.0	50.6	51.3
2	48.7	48.3	49.1	50.2
3	48.5	48.4	48.0	50.3
mean	48.7[a]	48.6[a]	49.2[a]	50.6[b]
st. dev.	0.2	1.4	1.2	1.1
max	47.3	51.1	51.8	51.4
min	48.5	47.3	47.2	48.5

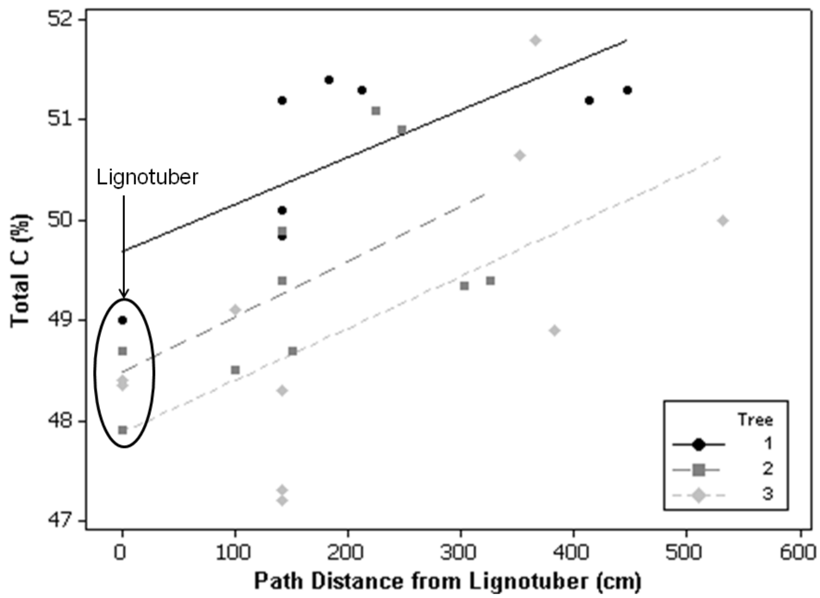

Figure 3—Change in C with increasing path length from the lignotuber. Lignotuber included as zero distance.

Discussion

The grand mean root wood density of 589.58 kg/m^3 was almost equal to the value of 580 kg/m^3 reported in the Global Wood Density Database for tanoak in North America (http://datadryad.org/handle/10255/dryad.235, Alden 1995) even though latitude, temperature and precipitation (Wiemann and Williamson 2002), successional stage (Henry et al. 2010, Wiemann and Williamson 1988, Woodcock and Shier 2002), stocking levels (Persson et al. 1995), and diameter distribution (Ruiz-Jaen and Potvin 2010) typically contribute to variation in wood density across forests. The differences in root wood density and percent C between trees was not surprising considering individuals of the same species often exhibit large within-stand variation due to genetic variation (Fries and Ericsson 2009, Wang et al. 2000, Zhang and Morgenstern 1995) and canopy position (Aiba and Nakashizuka 2009, King and et al. 2005). The variations in root wood density between different sample sections within each sample tree was a concern, and suggests that great care must be taken when measuring green root dimensions for volume estimates. The large differences in root wood density between different sample trees indicated that a large sample size is needed to obtain precise estimates of belowground biomass in tanoak at the stand level (*table 2*).

The increase in C and absence of detectable change in wood density with distance from the lignotuber is contrary to aboveground findings that wood density decreases with increasing stem height (Correia et al. 2010, Park et al. 2009, Paul et al. 2008). Larger diameter roots containing more biomass did not have greater wood density or percent C (*table 2*), suggesting that root systems allocate C differently than aboveground stems. Our finding that wood density does not increase with larger diameter roots also suggests that tanoak roots may not develop juvenile wood. Peterson et al. (2007) reported that juvenile wood was absent in roots and that roots have a relatively high density at a low cambial age (ring number from the pith). This contradicts the common assumption that the cambium is constrained to produce short cells when it is young and after a certain period of time produces denser wood. Because roots, branches and stems have different mechanical and physiological functions, it is not safe to assume that above- and belowground components allocate C in the same way.

Modeling percent C as a function of the continuous variable ‘path distance’ from the lignotuber (*fig. 3*) will allow us to make better predictions of belowground C mass in tanoak. Given that most of the belowground biomass is centrally located near the stem, summing C estimates for discrete sections of roots with different C content will give more accurate estimates at the tree level than if the grand mean of 49.5 percent C content was applied to estimates of total root system biomass.

Conclusions

Root wood density was highly variable within and between trees, but remained approximately constant on average with increasing distance from the lignotuber. Percent C did not remain constant throughout tanoak root systems. Accounting for density and C differences throughout the root system will allow for more accurate estimation of belowground C.

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Response of *Montia howellii* (Howell's montia) to Road Management in California Coastal Timberlands[1]

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Abstract

Howell's montia (*Montia howellii* S. Watson), a tiny annual plant with a California Rare Plant Rank of 2.2 (rare in California but more common elsewhere), is found throughout its range in seasonally wet, natural and disturbed habitats. On Humboldt Redwood Company timberlands it occurs on native surface or lightly rocked roads and turnouts, and in wet meadows used by cattle. We examined the spatial and temporal extent of Howell's montia on 10 road complexes over 6 years by counting the plants in consecutive fixed 10 meter segments of road. Using GIS event routing, we documented the changes in plant numbers in each segment, the movement of plants into new segments, and their decline and disappearance from segments over time. We found that opening and using seasonal, native surface roads was related to increases in plant numbers and spatial spread. After road use ceased, plants declined in both numbers and spatial extent. On rocked roads with heavy use and annual maintenance, populations were not well sustained even with a nearby seed source. We conclude that where Howell's montia occurs on roads, periodic seasonal use and road maintenance appears to maintain the local population.

Key words: GIS event routing, Howell's montia, *Montia howellii*, rare plant mitigation, road management, seed bank

Introduction

Howell's montia (*Montia howellii*, *fig. 1*) is a tiny annual of the Portulacaceae (purslane family). Germinating in late fall, it grows through the early spring, flowers from February to May, then sets seed and quickly disappears. Its range generally coincides with the maritime coastal forests from southern British Columbia to Northern California (CNPS 2011, Hickman 1996, Hitchcock and Cronquist 1973). The Global Status is G3G4, or vulnerable to secure (NatureServe 2011). In California it is ranked S3 (CNPS 2011), in Oregon and British Columbia it is ranked S3S4, (Oregon Biodiversity Information Center 2010), and in Washington it has been dropped to watch status because it has been found to be more abundant than previously assumed (Washington Natural Heritage Program 2011).

Throughout its range, Howell's montia typically occurs in sparsely-vegetated moist to seasonally wet lowland areas such as river and pond edges, cattle and game trails, open fields, vernal pools, seeps, and wet prairies, often on compacted soil (CNPS 2011, Hickman 1996, Kaye 1991, Wilson 1998). It is also found on human-

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[1] An abbreviated version of this paper was presented at the redwood science symposium: coast redwood forests in a changing California, June 21-23, 2011, Santa Cruz, California.

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Figure 1—A single plant of *Montia howellii,* a quarter, and redwood twigs.

disturbed habitats such as dirt roads, skid trails, landings, turnouts, parking areas, and lawns (CNPS 2011, Kaye 1991, personal observation), and it appears to need disturbance for survival (Kaye 1991, Wilson 1998). It occurs from near sea level to over 800 m (CNPS 2011, personal observation).

It was believed extirpated in California until rediscovered in 1999 near the Van Duzen River in Humboldt County during plant surveys conducted prior to timber harvesting. Since then, 69 populations have been located in California (CNPS 2011), and threats are believed to exist from logging, road construction and maintenance, vehicles, and competition. On Humboldt Redwood Company land (84,000 ha), we have documented 43 populations totaling over 300,000 plants.

From monitoring that we conducted from 1999 to 2004, we found wide site-level fluctuations in both plant numbers and spatial extent. Howell's montia numbers declined and the density of other herbaceous species increased where we avoided impacts to Howell's montia by closing roads, barricading turnouts with rebar stakes and flagging, and placing signs directing drivers to stay on the road running surface, (*fig. 2*). We found that plant numbers typically increased the following year after barricades were removed to allow road use in the summer while the populations existed as seeds. Plant numbers decreased where dirt road segments were upgraded to rocked segments, usually done to minimize sediment inputs to streams.

To better understand the changes that occur in Howell's montia populations, we developed this case study to document the extent of temporal fluctuations in the numbers and locations of Howell's montia on selected roads, and to correlate observed changes with the timing of road use and other disturbances.

a)

b)

Figure 2—Effects of mitigation applied 1999 to 2004 showing herbaceous overgrowth: a) Sign with overgrown turnout behind it, and b) a turnout barricaded for 6 years; both sites are no longer suitable habitat for Howell's montia.

Methods and materials

Study area selection

We selected 10 road complexes for this study (*table 1*) on Humboldt Redwood Company land. They include a full range of small to large populations, elevations from 45 m (150 ft) to over 800 m (2,625 ft), and are located in several major stream drainages representing the majority of the company's land base and road management practices. Both rocked and native-surfaced roads were present in the road complexes, except for Chadd and Larabee which had no rocked roads. Mainline rocked roads and some of the seasonal native-surface roads have regular use and maintenance each year, while most seasonal roads have disturbance only in years when they are used for timber harvesting operations. Prior to beginning the study, we were aware that some areas had periodic light-vehicle or foot-traffic disturbance from our watershed monitoring activities or from trespassers. Other areas had continuing disturbance from cattle.

Table 1—*Road complexes used in the study.*

Road complex	Cal Planning Watershed Sub-basins	Elevation range (m)	Topographic location	Km
Booth's Run	Booth's Run	524-610	Upper slopes	1.82
Chadd	Chadd Cr.	183-408	Lower/middle slopes	5.20
Cummings Cr.	Cummings Cr.	122-305	Lower/middle slopes	4.65
Jordan	Jordan Cr.	61-122	Lower slopes	6.50
Larabee	Scott Cr. Complex	701-884	Ridges, upper slopes	5.63
Monument A55	Monument Cr. and Kiler Cr.	183-427	Ridges, upper slopes	5.70
Stafford A51	Jordan Cr.	524-610	Ridges, upper slope	1.29
Van Duzen L35	Hely Cr.	61-110	Lower slopes	4.02
Van Duzen L64	Root Cr.	76-171	Lower slopes	4.02
Wrigley	Lower N. Fork Elk R.	30-104	Lower slopes	2.57
			Total Km in the study	41.40

The locations used for this study are shown in *figure 3*. We incorporated approximately 25 percent of the populations[4] on the property, collectively encompassing approximately 80 percent of the plants as of 2004.

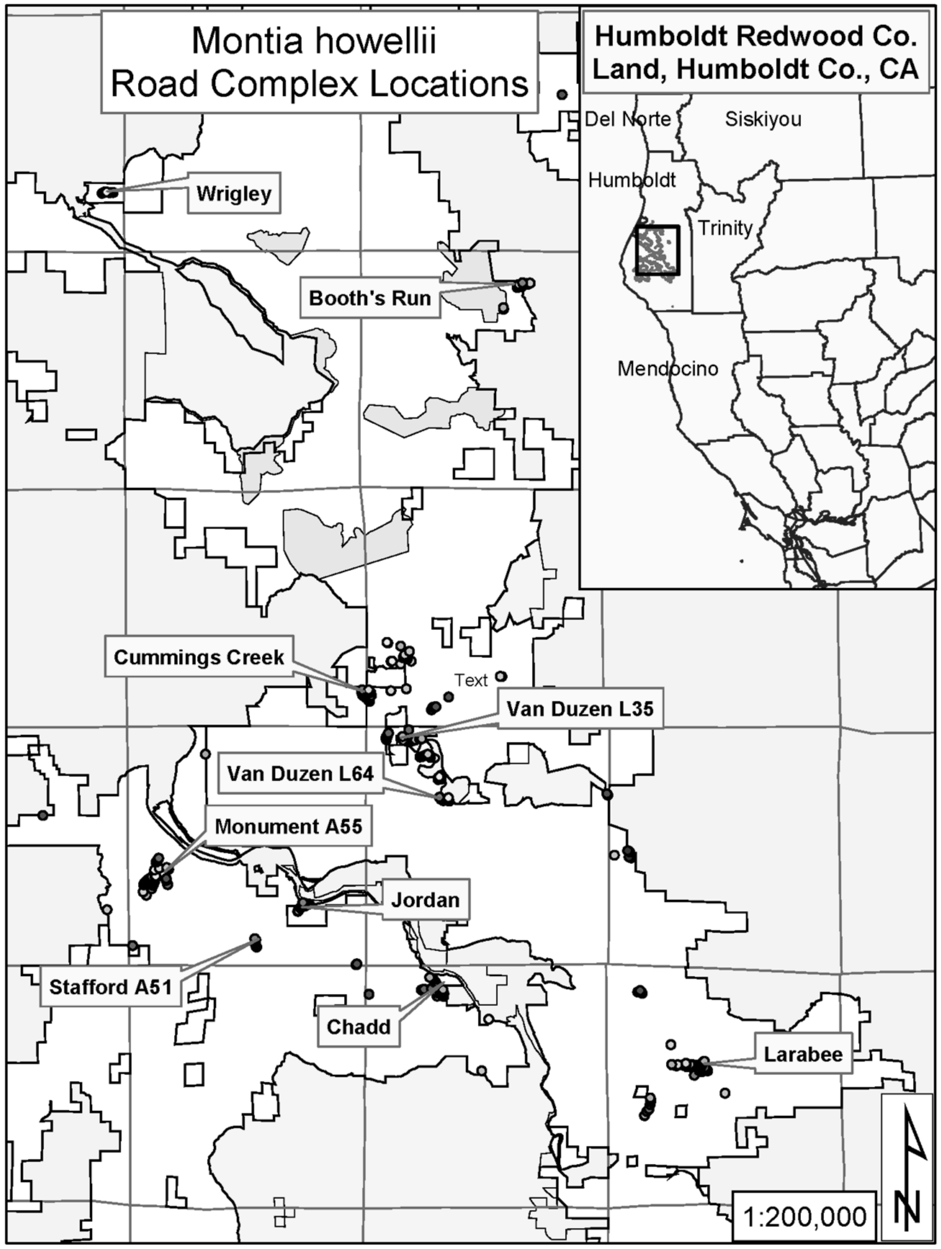

Figure 3—Location of study sites on Humboldt Redwood Company lands in Humboldt County, California. Map inset shows the property in relation to surrounding counties in California.

Data collection

Howell's montia typically occurs in sparse to dense clumps of plants often separated by long road segments with few or no plants. Given the miniscule size of

[4] Populations are defined as collections of occurrences separated by at least ¼ mile.

this plant and the variation in density, we knew that standard methods of sampling would result in non-normal, highly skewed, zero-rich data (McCune and Grace 2002). A random allocation of fixed-area quadrats, especially where there were unoccupied segments of road in a sparsely-occupied road complex, would be an inefficient design (Green 1979, Salzer and Willoughby 2004). It would result in an inflation of the error variation among quadrats because of the clumped differences in abundance, and small new groups would be easily missed. For the same reason, point intercept or line intercept sampling methods would also be ineffective (McCune and Grace 2002). Therefore, we elected to conduct an extensive survey of the entire road prism and associated turnouts and landings along all interconnected roads in each study area; in other words, we counted the entire population at each site in each visit. Even though we did not use a sampling process, this study is sufficiently inclusive to enable us to make inferences about the management of Howell's montia on roads in managed timberland settings in California (Elzinga et al. 1998, Elzinga et al. 2001, Mueller-Dombois and Ellenberg 2002, Quinn and Keough 2002).

We collected data from mid-February through mid-April (end of March in dry springs), starting with lower elevation roads before progressing to higher elevations. From a random start outside the area containing plants, we divided the road into fixed 10 m lengths and counted the plants in each segment. If the area was sparsely occupied we counted individual plants; where the plants were denser, we made a careful estimate of the number[5]. We monumented the random start so we could re-establish the location of the fixed 10 m segments in subsequent years. We continued for a fixed distance of 500 m (0.31 mi) beyond the last observed Howell's montia plant or until the habitat became unsuitable. This ensured that we were likely to find plants that had spread beyond the original extent of the occupied area.

Data analysis

We recorded the data in a Microsoft Access table and linked it to our Global Information System (GIS) roads layer. We added a conversion to account for discrepancies between road (slope) distance and GIS (planar) representation, although we had to accept a small level of error on roads with steep slopes. This process gave each 10 m segment, as measured on the ground, an address along the road system in our GIS. This type of analysis is known as "event routing," which characterizes each 10 m segment as an "event" containing the data while the GIS road layer is the "route" on which that data rides. We then used the event route tables to create a map set that very accurately displayed the plant locations and numbers for each year of the study.

We queried our road work database and project archives to create a management history for each road system that included the specific roads used in any year and the type of management that occurred each year starting in 2004.

Results

Plant number changes

Plant numbers fluctuated widely at most sites (*fig. 4*) and even within individual 10 m segments, similar to what we had documented prior to initiating this study. The

[5] All team members are experienced in counting and estimating Howell's montia numbers.

dates shown on the graphs in *figure 4* indicate when the greatest amount of road use from timber harvesting operations occurred. Operational activities took place in the summer after plant counts were taken; any changes in numbers that may have resulted from the disturbance from operations were recorded in the next scheduled survey year. Plant numbers increased in all populations following disturbance, and declined within 2 years after disturbance ended.

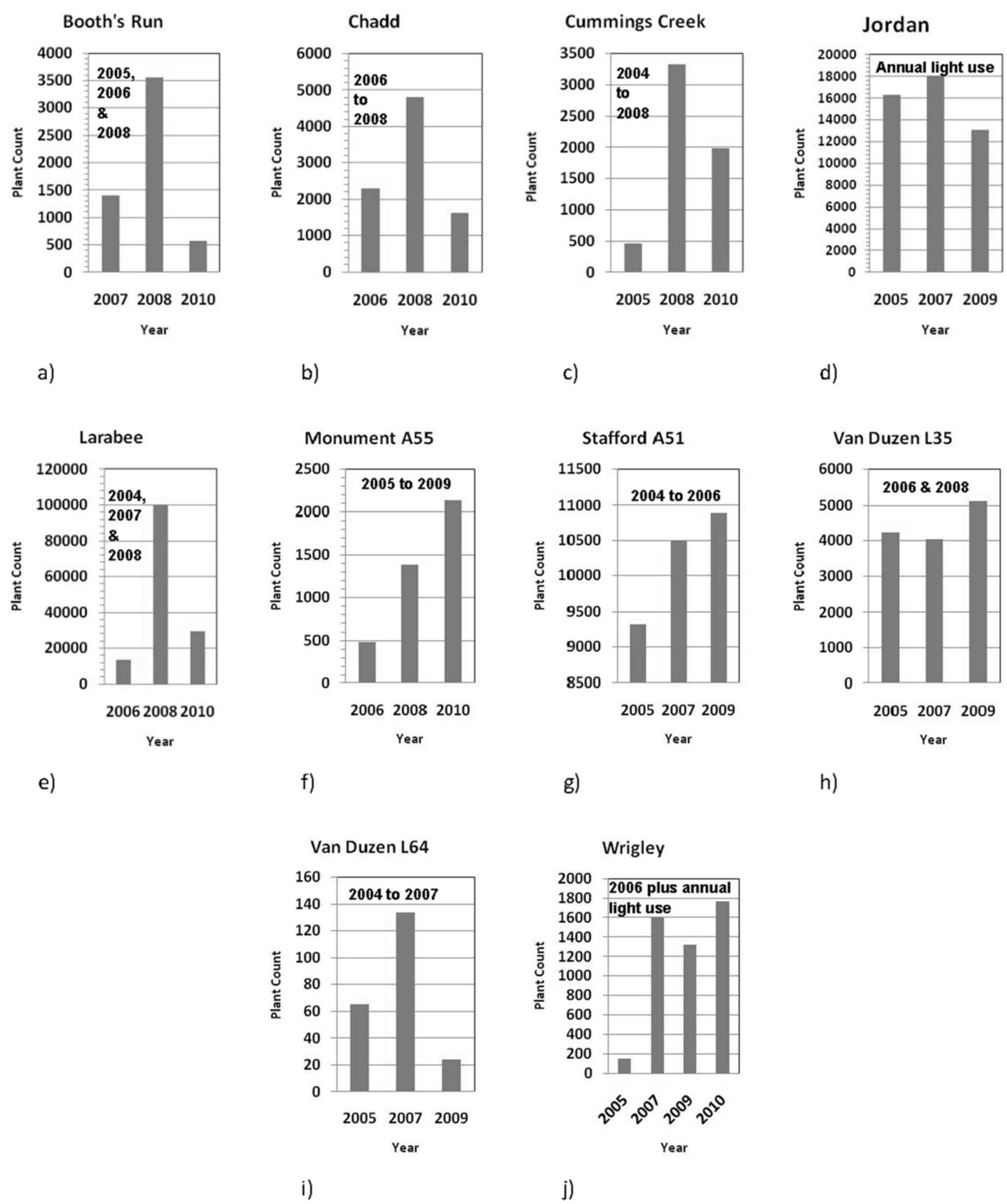

Figure 4—*Montia howellii* plant numbers, 2005 to 2010: a) Booth's Run, b) Chadd, c) Cummings Creek, d) Jordan, e) Larabee, f) Monument A55, g) Stafford A51, h) Van Duzen L35, i) Van Duzen L64, and j) Wrigley. Dates indicate years of maximum disturbance.

In addition to disturbance from road use associated with timber harvesting, there were other sources of disturbance documented in all road complexes except the Chadd and Van Duzen L64 road complexes. Disturbance from cattle was present at

Booth's Run, Larabee, Monument A55, and Stafford A51. At Cummings Creek, Jordan, Van Duzen L35, and Wrigley there was light vehicle use of the roads and turnouts by pickups and all-terrain vehicles. Where disturbance continued to maintain lightly-vegetated habitat conditions, plant numbers did not decline.

Changes in density and location

Two examples of temporal changes in plant density and location are shown in the following figures, generated from GIS event routing. Similar changes took place at the other road complexes.

The Chadd road complex (*fig. 5*) illustrates what we found when disturbance ended: competing vegetation became dominant while Howell's montia numbers declined. Timber harvesting and road maintenance in 2006 and 2007 required opening all but the most eastern road. The population increased in 2008 to 4,880 plants, more than double the 2,361 plants found in 2006, and plants were documented on two spur roads that had no plants at the beginning of the study. In 2008, road work occurred at two stream crossings; plant numbers in 2010 remained high near those locations but declined on all the other roads.

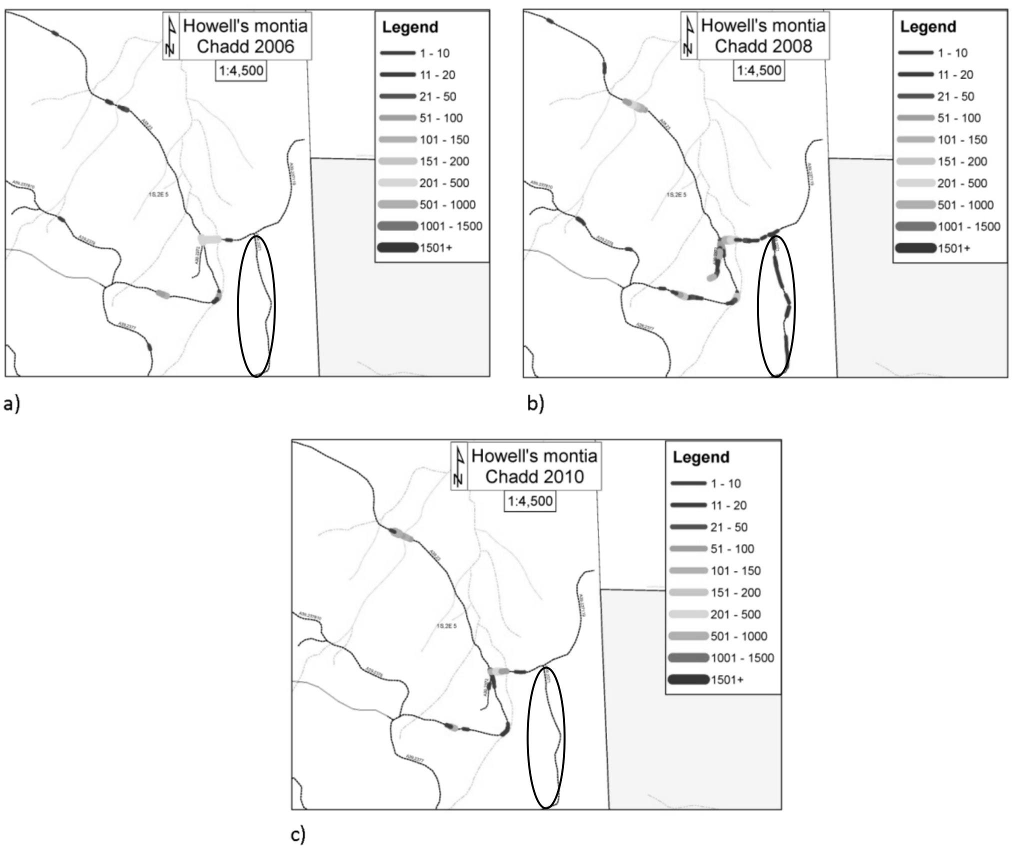

Figure 5—Spatial changes at the Chadd road complex generated from GIS event routing: a) 2006, b) 2008, c) 2010; width of the symbol indicates number of plants in the 10 m segment.

For example, the southeast spur road outlined in *figure 5* had no plants in 2006 before it was opened and used for logging. In 2008 it had a low density of up to 20 plants per 10 m, but no plants in 2010. The declining numbers documented on spur roads such as this one resulted in a decrease in total plant numbers throughout the road complex to just over 1,600 in 2010.

The Monument A55 road complex (*fig. 6*) is an example of what can happen when forest management operations open roads that are subsequently used by cattle. Here, timber harvesting and road maintenance took place 2005 to 2009 on the mainline road but the spur roads were opened and used only during 2005 to2006. Overall plant numbers in the entire road complex showed a steady increase from 486 plants in 2006 to 2,133 plants in 2010.

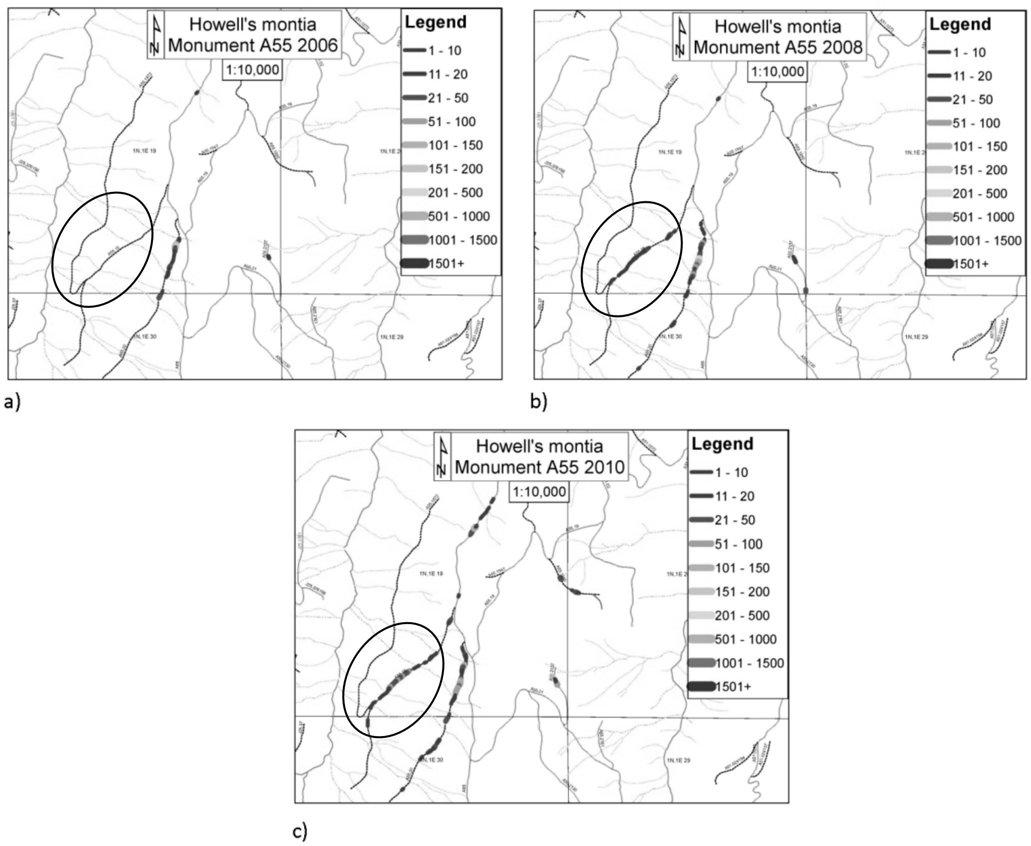

Figure 6—Spatial changes at the Monument A55 road complex generated from GIS event routing: a) 2006, b) 2008, and c) 2010; width of the symbol indicates number of plants in the 10 m segment.

On the southwest spur road outlined in *figure 6*, our previous monitoring from 2004 documented only 20 plants in a 670 m length. In the spring of 2006, the first time we counted the population for this study, we found no plants on this spur road. It was opened to allow harvesting in summer of 2006, and in 2008 and 2010 we recorded 171 and 673 plants respectively with plants present all along a previously unoccupied 400 m length of road. The increase through 2010 occurred even though the road was only used for harvesting operations in 2006 and closed when operations were completed. However, cattle are abundant in the area (from legal leases) and the increases in Howell's montia numbers and locations were associated with continuing use of the newly-opened roads by cattle.

In all areas, heavily used graveled mainline roads did not support plants except in some turnouts, even though large numbers of plants were present on nearby roads.

Discussion

When roads known to be occupied by Howell's montia were opened and used for timberland management, we found short-term and sometimes surprisingly large increases in plant abundance as well as local changes in spatial extent following disturbance from operational activities. Plants spread into adjacent, previously unoccupied road segments in all 10 study areas when these roads were opened for use. Howell's montia numbers typically declined within a year after the cessation of disturbance as the roads and turnouts became dominated by grasses and forbs. We speculate that new groups of Howell's montia that appeared after opening some of these roads came from an in-situ seed bank, as may be the case for the Monument A55 spur described above. Other new groups may have come from seed spread by road graders and other equipment from nearby sites containing active populations.

After roads were opened, in locations where the sparsely vegetated habitat on seasonal roads was maintained by cattle or light vehicle use, plant numbers either increased or declined more slowly than in the areas where there was no continued disturbance from cattle or vehicles. For example, at Stafford A51 (disturbance from cattle) and Wrigley (light vehicle use), plant numbers continued to increase after the roads were no longer used for harvesting operations. High numbers were also maintained at Jordan where no harvest-related use occurred on the roads occupied by Howell's montia, but where there was ongoing light vehicle use. This disturbance apparently maintained the conditions needed to sustain Howell's montia population numbers at these and other similar sites. Conversely, at Chadd and Van Duzen L64, where neither cattle nor light vehicle use were present to maintain an open, lightly-vegetated habitat, population numbers decreased sharply following the end of road use.

An unexpected benefit derived from this work has been to apply the GIS event routing mapping to operational needs. For example, work on a Van Duzen L35 road in 2008 was adjusted to avoid significant impacts to most of the 10 m segments containing Howell's montia. Plant counts more than doubled on that road the following year.

The study results indicate that maintaining populations of this species can be compatible with active forest management. Where ongoing disturbance to populations from summer road maintenance and use occurs, conditions favorable to Howell's montia have been preserved and population numbers remain fairly stable. As part of our Howell's montia management strategy, we avoid heavy road rocking, excavation, and deep grading where plants are known to occur, since these activities can alter the microsite conditions or bury the seed bank. We will also continue periodic surveys, though on a less intensive schedule.

The Global and California state status of Howell's montia are not likely to change as a result of this study. The southern range extent in California is limited to two counties, and while known populations there can have high numbers of plants, most are on managed timberlands. If more populations are recorded on non-commercial timberlands and submitted to the Biogeographic Data Branch of the California Department of Fish and Game, this plant will be eligible for status review. Until then, more surveys are needed.

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‘Pygmy’ Old-Growth Redwood Characteristics on an Edaphic Ecotone in Mendocino County, California

Will Russell[1] and Suzie Woolhouse[2]

Abstract

The ‘pygmy forest’ is a specialized community that is adapted to highly acidic, hydrophobic, nutrient deprived soils, and exists in pockets within the coast redwood forest in Mendocino County. While coast redwood is known as an exceptionally tall tree, stunted trees exhibit unusual growth-forms on pygmy soils. We used a stratified random sampling procedure to characterize forest composition, structure, and growth-form, across the pygmy/redwood ecotone. Results indicate that tree height, canopy cover, basal area, herbaceous cover, shrub height and cover, and the dominance of pygmy forest endemics decreased across strata, and were negatively correlated to soil pH. In addition, the structure of individual redwoods varied significantly across the ecotone, exhibiting three main growth-forms: 1) Stunted multi-stemmed (>100 stems) individuals >2 m in height; 2) Small diameter muti-stemmed (two tosix stems) individuals with an average height of 12 m growing as subcanopy under pygmy cypress and bolander pine; 3) Pygmy old-growth redwoods with a maximum height of 41 m growing as a co-dominant and exhibiting complex canopy structure. The results of this study indicate that while the growth-form of individual redwoods is significantly affected by pygmy soil conditions, stunted trees on the edge of the pygmy formation exhibit structural characteristics and canopy complexity similar to full stature redwoods.

Key words: Coast redwood, hydrophobic podisols, pygmy forest, *Sequoia sempervirens*

Introduction

Coast redwood (*Seqouia sempervirens*) is the tallest tree species in the world reaching heights exceeding 112 m (Preston 2007). Where site conditions are less than optimal, however, growth can be stunted (Noss 2000). In the most extreme cases, *S. sempervirens* exhibits a shrub-like growth-form with an abundance of clonal stems connected to a single root mass, with no single stem exerting apical dominance. In less extreme cases *S. sempervirens* will develop a growth-form similar to the giants for which the species is known, but on a smaller scale. The most dramatic transition from giant redwoods to stunted redwood shrubs occurs on the ecotone between the ‘pygmy forest’ formation and the coast redwood forest in Mendocino County, California.

The pygmy forest is found on a series of five coastal terraces that increase with age as they progress inland from the coast (Fox 1976, Yu et al. 1999); the oldest with soils of approximately one million years (Westman and Whittaker 1975). Variation in vegetation between terraces is related to soil conditions. The first terrace has well

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drained soils rich in organic matter and nutrients (Aitken and Libby 1994) supporting shore pine (*Pinus contorta* subsp. *contorta*), bishop pine (*Pinus muricata*), and sitka spruce (*Picea sitchensis*) (Sholars 1982). Coast redwood mixes with western hemlock (*Tsuga heterophylla*) and Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) on the second terrace, where podzolized soils begin to develop. The third, fourth, and fifth terraces where pygmy forest are found contain highly acidic, extremely podzolized, soils with a hydrophopic hardpan (Aitken and Libby 1994). Consequently, pygmy forest species, dominated by two endemic species, bolander pine (*Pinus contorta* subsp. *bolanderi*) and pygmy cypress (*Hesperocyparis pygmaea*), are well adapted to both high water table conditions (Sholars 1982, Wurzburger and Bledsoe 2001) and summer drought (Westman and Whittaker 1975).

Complex lateral branching patterns, reiterated trunks, arboreal soils, and epiphytic plants have been well documented on full stature redwoods, which commonly occur on productive alluvial soils (Sillett and Van Pelt 2000). These features have been shown to provide habitat for threatened species such as the marbled murrelet (*Brachyramphus marmoratus*) and northern spotted owl (*Strix occidentalis caurina*) (Sawyer et al. 2000), and for less well known species such as the wandering salamander (*Aneides vagrans*) (Spickler et al. 2006). The occurrence of these features has not been adequately studied on less productive sites, such as those occurring adjacent to the pygmy forest formation. The objective of our research was to characterize the structure and composition of vegetation across the pygmy/redwood ecotone with particular emphasis on the growth-form of *S. sempervirens.*

Methods

This study was conducted on the Russell Unit of the Mendocino Headlands State Park, a 50 ha property composed of mixed primary old-growth and mature second-growth redwood, with patches of riparian and pygmy forest habitat. Stratified random sampling was used to locate 10 (7 m diameter) sample plots within each stratum.

Six stratum radiating out from the center of the pygmy formation were defined loosely on Westman (1975) and included: ‘extreme pygmy’ dominated by stunted pygmy cypress and bolander pine; ‘short pygmy’ with similar species composition but less stunted trees; ‘tall pygmy’ dominated by tall Ericaceous shrubs and large specimens of bolander pine and pygmy cypress; ‘transitional’ dominated by tall pygmy cypress mixed with bishop pine and sub-canopy coast redwood; ‘pygmy redwood’ characterized by stunted old-growth redwoods growing as co-dominants, or sub-canopy species, with bishop pine; and ‘extreme redwood’ dominated by tall old-growth redwoods with western hemlock, grand fir (*Abies grandis*), and Douglas-fir associates.

Data recorded on each plot included biotic factors including diameter (dbh) of trees by species, canopy cover, average canopy height, percent cover and height of shrub species, percent cover herbaceous species; and abiotic factors including pH of the A horizon, depth to the hardpan, and distance to the center of the pygmy formation. In addition, the growth-form of individual *S. sempervirens* specimens sampled on each plot were described with the following characteristics: height, number of stems emerging from root bole, diameter of largest stem, and presence of old-growth canopy characteristics (large lateral branches, reiterated trunks, epiphytes, and fire hollows).

The relationship between stand characteristics and soil pH were analyzed using linear regression. Variation of response variables between strata was analyzed using ANOVA with Benferroni post-hoc analysis.

Results

The size and growth-form of trees, composition of vegetation, and soil pH varied significantly across the pygmy/redwood ecotone. Soil pH ranged from 4.0 to 6.6, and was found to be the best abiotic predictor of stand characteristics. 'Depth to hardpan' and 'distance to the pygmy center' were found to be poor predictors.

A positive linear relationship was found between canopy height and pH, suggesting a negative correlation between canopy height and pygmy soil conditions. ANOVA analysis confirmed this relationship with significant differences in canopy height between strata with the highest canopy found in the 'extreme redwood' stratum, and the lowest found in the 'extreme pygmy' stratum (*table 1*).

Table 1—*Relationship between soil pH and six dependant variables measured across a pygmy/redwood ecotone on six strata; (I) extreme pygmy, (II) short pygmy, (III) tall pygmy, (IV) transitional, (V) pygmy redwood, and (VI) extreme redwood. Reported R^2 and p-value result from linear regression analysis with soil pH. Strata sharing the same lower case letters did not exhibit significant differences based on ANOVA.*

	I	II	III	IV	V	VI	R^2	*p-value*
Median soil pH	4.8^a	4.95^b	4.95^b	5^b	5.25^c	5.95^c	-	-
Canopy Height (m)	2.09^a	5.64^b	11.95^c	21.5^d	39.6^e	58.3^f	*0.29*	*>0.001*
Canopy Cover (%)	37.1^a	49.2^b	59.2^b	83.9^c	91.7^d	97.7^d	*0.25*	*>0.001*
Shrub Cover (%)	70.8^a	89.3^b	78.1^b	60.4^c	34.1^d	46.3^d	*0.23*	*0.003*
Herb Cover (%)	1.8^a	1.3^a	3^a	0.4^a	2.35^a	16.4^b	*0.34*	*>0.001*
Understory richness	6^a	10^b	10^b	10^b	15^c	24^d	*0.27*	*>0.001*
Basal Area (m^2/ha)	142.9^a	186.6^a	66.8^b	109.4^a	371.3^c	479.8^c	*0.33*	*>0.001*
Basal Area of Redwood	0^a	0^a	0^a	101.9^b	335.1^c	371.3^c	*0.41*	*>0.001*
Basal Area of Pygmy Cypress	136.8^a	175.2^a	61.1^b	3.3^c	0^c	0^c	*0.35*	*>0.001*

Percent tree canopy cover, shrub cover, and herbaceous cover, also exhibited linear relationships with soil pH as well as significant differences between strata. Tree canopy cover was relatively low on pygmy soils, while shrub cover was relatively high. Herbaceous cover was uniformly low across the ecotone except for the 'extreme redwood' stratum, which was significantly higher.

Total basal area, and the basal area of coast redwood, exhibited positive linear relationships with soil pH, while the basal area of pygmy cypress exhibited a negative linear relationship (*table 1*). Pygmy cypress and bolander pine had the highest relative frequency on first three strata, and coast redwood and Douglas-fir on the last three strata (*fig. 1*). The presence of Douglas-fir in the 'tall pygmy' stratum and pygmy cypress in the 'transitional' stratum suggests a somewhat porous, boundary between the two communities.

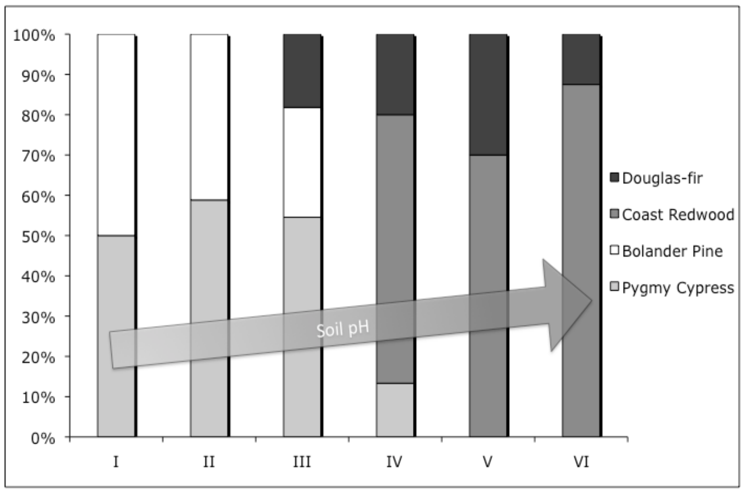

Figure 1—Relative frequency (n=60) of the four most commonly occurring tree species across six strata: (I) extreme pygmy, (II) short pygmy, (III) tall pygmy, (IV) transitional, (V) pygmy redwood, and (VI) extreme redwood, on a pygmy ecotone.

The maximum height, and growth-form, of coast redwood trees also varied significantly across the pygmy/redwood soil gradient (*fig. 2*). In the 'extreme pygmy' and 'short pygmy' strata the growth-form coast redwood was limited to rare multi-stemmed (>100 stems/plant) shrub-like forms. Still infrequent, but less stunted forms, were found in the 'tall pygmy' and 'transitional' strata. Individuals within these two strata tended to be vertical in form with fewer stems (two to six) and generally grew in a conical form with little lateral branching in the crown.

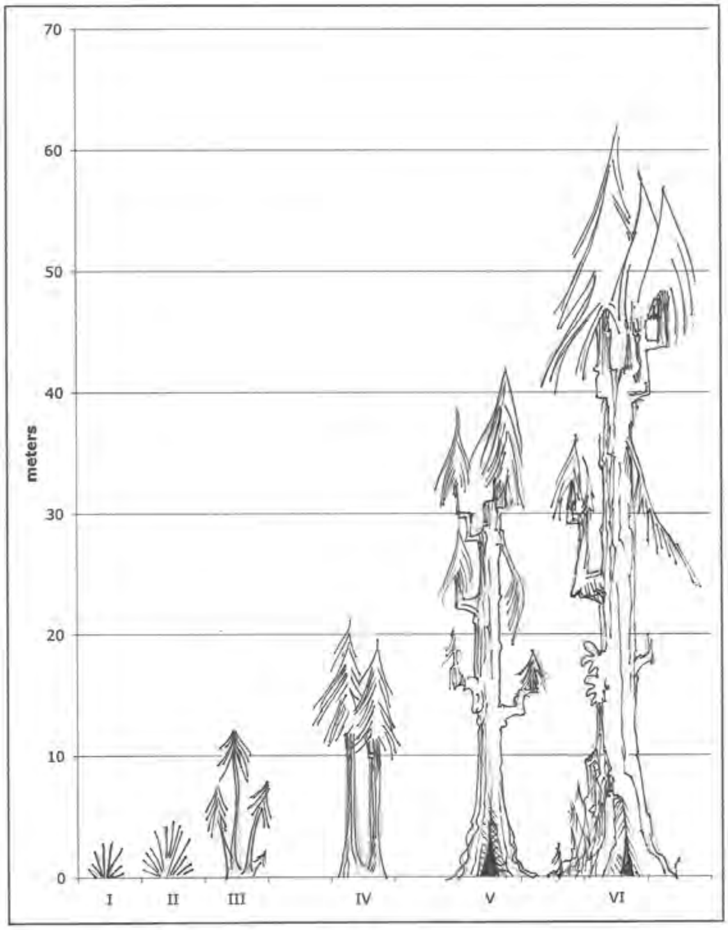

Figure 2—Canopy height of *S. sempervirens* across six strata (n=60); (I) extreme pygmy, (II) short pygmy, (III) tall pygmy, (IV) transitional, (V) pygmy redwood, and (VI) extreme redwood, on a pygmy forest ecotone in Mendocino, California (SE=0.26; 0.76; 1.54; 4.31; 5.26; 7.34).

Individual redwood trees in the 'pygmy redwood' and 'extreme redwood' strata grew in a vertical upright form, and exhibited complex canopy features including lateral branching patterns and multiple trunk reiterations. Well-developed fire hollows were common on older trees in both strata (*fig. 4*), and epiphytic vascular plants such as leather fern (*Polypodium scouleri*) huckleberry (*Vaccinium ovatum*), and billberry (*Vaccinium parvifolium)* were observed growing in the canopy. In many ways the growth-form of old-growth trees was comparable between the 'pygmy redwood' and 'extreme redwood' strata, except in regard to scale. The maximum tree diameter measured in the 'extreme redwood' was 750 cm compared to 244 cm in the 'pygmy redwood' and the maximum heights were 66 m and 41 m respectively.

Figure 4—The 'Owl Tree' found in the 'Pygmy Redwood' strata with insert illustrating complex branching pattern at the top of tree. A continuous fire hollow ran from top to bottom, and was a nesting site for barn owls (*Tyto alba*). Photo courtesy of Dylan Crutchfield, Department of Environmental Studies, San Jose State University.

Similarities between the 'pygmy redwood' and 'extreme redwood' strata were less pronounced in regard to understory species. Both herbaceous cover and richness of understory species were significantly higher in 'extreme redwood' compared to all other strata (*table1*). There were, however, a number of species that were limited to pygmy soils including two California Native Plant Society listed species, pygmy manzanita (*Arctostaphylos mendocinoensis*) and the coast lily (*Lilium maritimum*) (*table 2*).

Table 2—*Mean percent cover per plot of understory shrubs and herbaceous species occurring on a pygmy forest ecotone across six strata, (I) extreme pygmy, (II) short pygmy, (III) tall pygmy, (IV) transitional, (V) pygmy redwood, and (VI) extreme redwood. Association designations indicate species that were limited to specific habitat types.*

Association		I	II	III	IV	V	VI
Pygmy	*Arctostaphylos columbiana*	0	0	0.3	0	0	0
Pygmy	*Arctostaphylos mendocinoensis*[a]	0	0.2	0.2	0	0	0
Redwood*	*Blechnum spicant*	0	0	0	0	0	2.4
Redwood*	*Boschniakia strobilacea*	0	0	0	0	0	0.1
Redwood	*Calypso bulbosa*	0	0	0	0.1	0.25	0.5
Redwood*	*Cardamine californica*	0	0	0	0	0	2.1
Redwood	*Chimaphila menziesii*	0	0	0	0.1	0	0
Pygmy	*Chrysolepis chrysophylla*	0	0	1.6	0	0	0
	Claytonia sibirica	0	0.1	0	0	0.1	0.15
Redwood	*Clintonia andrewsiana*	0	0	0	0	0.5	1.5
Redwood*	*Corallorhiza mertensiana*	0	0	0	0	0	0.3
Redwood*	*Disporum hookeri*	0	0	0	0	0	0.2
Redwood*	*Epipactis gigantea*	0	0	0	0	0	0.1
Redwood*	*Equisetum telmateia* subsp. *braunii*	0	0	0	0	0	0.2
	Gaultheria shallon	8.9	8.9	9.4	7.5	9.5	23.2
Redwood	*Goodyera oblongifolia*	0	0	0	0.1	0.2	0.1
Pygmy	*Ledum glandulosum*	24.5	46	4.5	0	0	0
Pygmy	*Lilium maritimum*[b]	0	0.1	0	0	0	0
	Myrica californica	5.6	3.8	4.2	6.2	8.7	0
Redwood*	*Oxalis oregana*	0	0	0	0	0	6.2
Redwood	*Polypodium scouleri*	0	0	0	0	0.5	0
Redwood	*Polystichum munitum*	0	0	0	0	0.3	9.8
	Pteridium aquilinum var. *pubescens*	5.4	3.4	1.8	0.9	1.5	0.5
Redwood*	*Rhamnus californica*	0	0	0	0	0	0.1
	Rhododendron macrophyllum	3.5	4.2	29.5	13.1	4.6	7.3
Redwood*	*Smilacina racemosa*	0	0	0	0	0	0.1
Redwood*	*Stachys bullata*	0	0	0	0	0	0.1
Redwood*	*Tiarella trifoliata* var. *unifoliata*	0	0	0	0	0	0.5
Redwood	*Trientalis latifolia*	0	0	0	0	0.1	0
Redwood	*Trillium ovatum*	0	0	0	0.1	0.7	1.4
	Vaccinium ovatum	23.7	25.3	27	32.5	5.6	1.8
Redwood	*Vaccinium parvifolium*	0	0.1	1.7	1.1	1.2	3
	Viola sempervirens	0	0	0	0	0.1	1.25

[a]CNPS special status 1B.2

[b]CNPS special status 1B.1

*Species only observed in ‘extreme redwood’ strata

Discussion

Westman and Whittaker (1975) described the pygmy forest region of northern California as the area containing the "smallest and the largest forests in the world." This dramatic contrast between the pygmy and coast redwood communities is based on an edaphic gradient of soil pH, moisture, and nutrient availability. While massive redwoods growing on productive soils have been the subject of intense study, the tenacious specimens of *S. sempervirens* persisting on the edge of their ecological tolerance have been neglected in the literature.

While our study involved an analysis of six strata, each with unique characteristics, we observed essentially three growth-forms of *S. sempervirens*: a muti-stemmed shrub-like form; a vertical, highly stunted sub-canopy form; and a tall vertical co-dominant form with complex canopy structure. *S. Sempervirens* is able to manifest multiple growth-forms as a result of prolific basal and epicormic sprouting. Sprouting in coast redwood is a stress response, and where the stresses are unremitting the sprouting is continuous. The muti-stemmed shrub-like form of *S. sempervirens* found in the 'extreme pygmy' is essentially a mass of basal sprouts emerging from a single half buried lignotuber.

In addition to variation in the growth-form of *S. sempervirens,* stand structure and composition of the stand as a whole varied across the pygmy/redwood ecotone. As expected the dominance of pygmy cypress and bolander pine and the cover of Ericaceous shrubs (*Arctostaphylos mendocinoensis, Gaultheria shallon, Ledum glandulosum, Rhododendron macrophyllum, Vaccinium ovatum*) were at their maximum, and canopy cover and canopy height were at their minimum, within the pygmy formation. What was not predicted in the literature was the porosity of the boundary between the two communities with Douglas-fir observed within the 'tall pygmy' strata and pygmy cypress found growing in transitional stands. The influence of pygmy soil conditions on adjacent redwood stands was also surprising. In stands that would not be considered 'pygmy' by any of the traditional pygmy forest definitions, stand characteristics varied significantly from 'extreme redwood' stands and trended toward pygmy conditions. For example, lower cover of herbaceous species in the 'pygmy redwood' as compared to 'extreme redwood' suggests a sensitivity of understory species to pygmy soil conditions, possibly as a result of poor nutrient availability resulting from low soil pH (Northup et al. 1998).

There is a tendency to think of coast redwood forests as uniformly impressive in terms of stature. In fact, redwoods are highly variable, and where redwoods grow at the edge of their ecological tolerance it is their resilience, and morphological flexibility, that is truly impressive.

Acknowledgments

This project was supported by the Mendocino Institute, the California Department of Parks and Recreation, and the spirit of Lewis and Ieda "Skilly" Russell.

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Size Distribution of Unharvested Redwood Forests in Mendocino County[1]

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Abstract

Late-seral conditions in redwood forests are becoming a management goal on some timberlands. However, published information is rare regarding the structure of late-seral forests upon which silvicultural prescriptions can be guided, or upon which to measure success. Old-growth forests—those perceived to be near or at the successional climax—are the ultimate model. Yet, old-growth forests are a rare commodity in Mendocino County, limited mostly to state parks and small stands on private lands. Their rarity yields a poor sampling of the variation among stands, making historic sources of information valuable. Despite often incomplete information relative to ecologically important but non-commercial vegetation such as hardwoods or the structural complexity of their trees' canopy and boles, historic timber inventories can yield insights to the range of unharvested forests conditions. A 1929 data set derived from timber stands stratified into 20-thousand board feet per acre redwood classes on Caspar Lumber Company lands reveals an inverse J-shaped size frequency distribution based on diameter at breast height (DBH). Across the stand conditions, redwood comprised 67 to 96 percent of the tree density (31 to 51 trees per ac) and 73 to 97 percent of the basal area (116 to 537 ft^2/ac); the DBH_{50} (diameter at which half of the trees were smaller) ranged from 17 to 30 inches DBH; while the DBH_{90} ranged from 42 to 70 inches. The largest size classes measured were 126+ inches, 78 to 82 inches, and 42 to 46 inches for redwood, Douglas-fir, and grand fir, respectively.

Key words: late-seral, late-successional, old-growth, stand structure, targets, hardwoods

Introduction

To enable setting goals and evaluating management actions, descriptive criteria of late-seral conditions can be important to forest managers in the redwood (*Sequoia sempervirens*) region. Where cut-over forests have been acquired in parks, restoration of more advanced and near-climax stages is a goal (e.g., Chittick and Keyes 2007). Other land managers desire to manage forest stands for late-seral values as well as for other values including commodity production (e.g., Thornburgh 2007). Still other forest land owners who have late-seral forests in a Timber Harvesting Plan can avoid significant impacts under the California Forest Practice Rules (Anon. 2011) if their harvests retain late-seral conditions after harvests.

Whether to establish success criteria for restoration or to consider the maintenance of or the degree of departure from late seral conditions, working definitions are lacking or imperfect. The Forest Practice Rules' (Anon. 2011) definition includes trees greater than 24 in either with or lacking understory trees; an open to dense canopy; a minimum area threshold (20 ac); and the presence of an

[1] This paper was presented at the Coast redwood forest in a changing environment: a symposium for scientists and managers. June 21-23, 2011, Santa Cruz, California.

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unspecified density of large decadent trees, snags, and large down logs. This definition's lack of specifics may be due to its intended use in many forest types across the state. However, equally important, its ambiguity may be a reflection that late-seral forests are variable. Not only may they vary locally due to site condition and localized disturbance dynamics, the late-seral stands may occupy any point along a successional continuum from mid-seral to climax conditions. Criteria developed specifically for different forest types and regions could enhance the use of the late-seral concept in forest management.

Unharvested redwood forests can be the source of information upon which to develop regional management goals. To the extent they represent near-climax conditions, unharvested forest represent perhaps the ultimate example of late-seral qualities. However, in Mendocino County, current unharvested stands are rare and of generally small acreage, limiting how much variation they may encompass. Despite the focus on economic values of forests, historic information can be mined to yield stand-based information to better understand the range of conditions present in late-seral forests. This paper characterizes the "virgin" forest conditions of conifers on the Caspar Block of the Caspar Lumber Company (Anon. 1929); a management unit that closely approximates the present-day Jackson Demonstration State Forest.

Methods and materials

According to the report (Anon. 1929), the lands inventoried were the "Caspar Block", an area of 23,564 ac of which 18,629 ac were virgin timber. The Caspar Block was in the watersheds of Two Log, Chamberlain, James and Two Rock Creeks, and adjoining parts of the South Fork of Noyo and the Little North Fork of Big River.

While the methods are described in the inventory, the description is limited. In summary, Caspar Lumber Company had preexisting cruises organized by 40 ac units. However, stand conditions frequently covered such small areas and were so important to deriving an accurate inventory that evaluating the timber values based on the 40 ac units would not have shown the actual situation clearly. To solve this problem, overlapping aerial photographs, complimented by the cruise information were used to create a map of "economic types". Then "more than 200" one quarter ac sample plots separated by 5 chains were sampled. Neither the starting point(s) nor the orientation of the sampling frame is specified. A sample of trees was cored, and the thickness of the bark recorded; however, the report is not clear if the size distributions calculated included or excluded bark thickness. The plots were not laid out to achieve any specified level of sampling within or among the "economic types". Rather, a plot was assigned to its economic type based on where the plot fell on the map. The report does not provide a count of plots by economic type, except that there were "a large number of sample areas in each type".

The "economic types" to which virgin stands were assigned was based upon the quantity of redwood/acre as derived from company cruises adjusted to agree with the company's cutting experience (*table 1*).

Table 1—*Identifers, definitions of economic types, and possible percentage of sample plots.*

Identifier	Redwood volume Board ft/Acre thousands (M)	Area (Acres)	Percent of Samples
80+	more than 80	431	7
60 – 80	between 60 and 80	528	3
40 – 60	between 40 and 60	2643	10
20 – 40	between 20 and 40	7120	44
<20	less than 20	7907	36

A map in the inventory (Anon. 1929) displays the geographic distribution of assigned economic types, roads, watercourses, and ridges. Also included are 61 numbered points that are neither identified on the map index nor the body of the report. These may reflect location information (possibly start points) for the plot sampling frame. To approximate the sampling distribution if it was based on these points, I calculated the percentage of points by the "economic type" in which they were placed (*table 1*).

Results

When the plot data for all conifers is grouped within economic types, each displays an inverse J-shaped curve (*fig. 1a*). Generally the curves are ordered according stand type, with the density of trees in the 80+ type greatest and that of the <20 type least. The ordering by stand type is most notable in the larger size classes. In the 14 to 22 in size class, the <20 type had the greatest density among the types. The size class distribution of redwoods (*fig. 1b*) also was ordered similarly among types, but with density of the 80+ types greater than the other in all size classes. Notably, the curve shows the smallest size class of redwood in the 80+ type being denser relative to the larger size classes than those of the other types, despite the presumably more heavily shaded understory in the 80+ stand. The largest redwood was in the 126 in size class, and was found in the 80+ type. Among types, the Douglas-fir (*Pseudotsuga menziesii*) size class distribution (*fig. 1c*) ranged from inverse J-shaped curves for the lowest economic types to flat curves for the highest types. The density of Douglas-fir in the latter types was very low, and much lower than for the lowest economic types. The largest Douglas-fir measured was in the 78 in size class of the 20-40 type. Grand fir[3] (*Abies grandis*) also exhibited flat to declining size class distributions (*fig. 1d*), and there was no apparent relationship between their density and the economic types. Although not common in any type, grand fir was not recorded in the <-20 type and was most common in the 60-80 type. The largest grand fir (42 in size class) was measured in all but the 60-80 type.

Total tree density ranged from 30 to 51 trees/ac (calculated values are rounded to the nearest whole number), increasing from the lowest to the highest economic type. Redwood comprised 67 to 96 percent of the trees in each type, with the highest economic type substantially greater than the other types. Graphs of cumulative percent of tree density by DBH class can enable a rapid assessment of the proportion of trees in a stand either greater or less than a given DBH. For instance, among the types, the median density of trees ranged from 18 to 30 in DBH, and 90 percent of

[3] The inventory identifies the third conifer species as white fir without its specific epithet. White fir is not known from the vicinity, and the intended species was most likely grand fir.

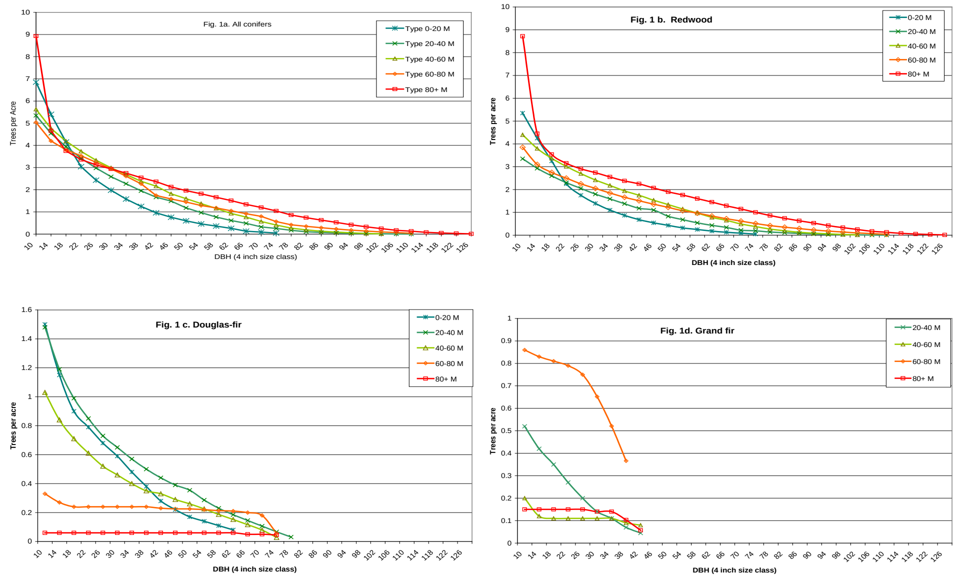

Figure 1—Conifer density (trees per acre) by size class and stand type in virgin forests of the Caspar Block, 1929; a=all conifers, b=redwoods, c=Douglas-fir, and d=grand fir.

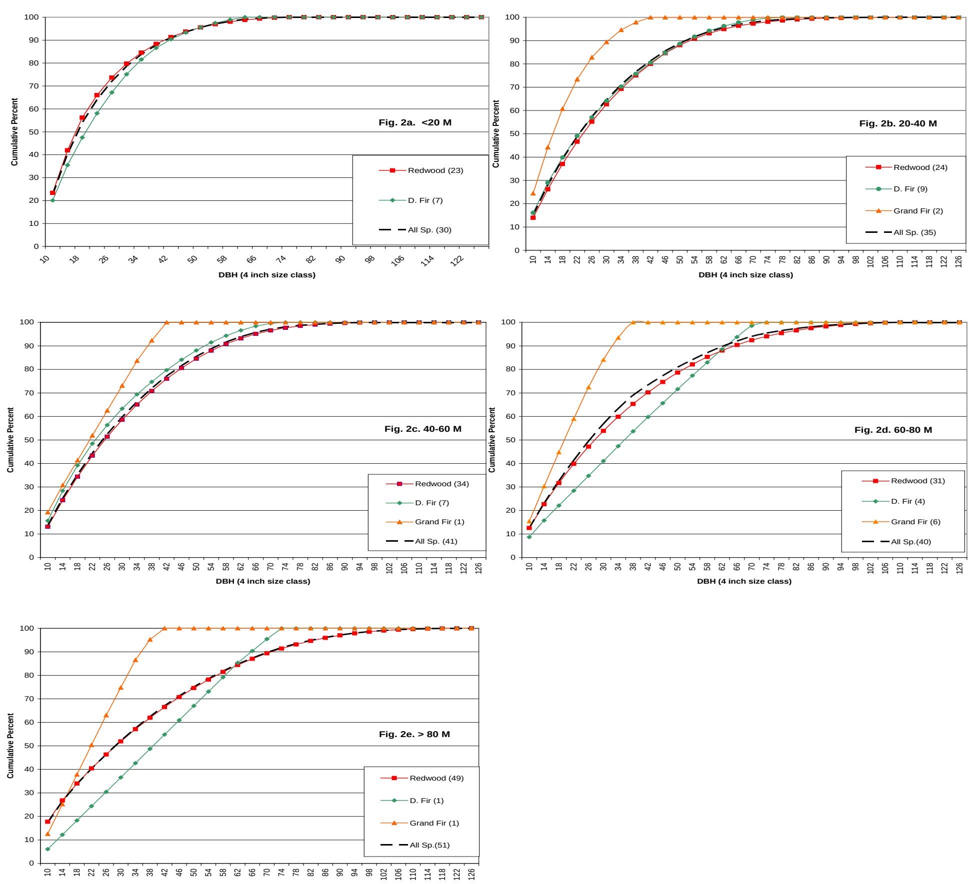

Figure 2—Cumulative percent of conifer trees by diameter class and species in virgin forests of the Caspar Block, 1929; stand types a) <20 M, b) 20-40 M, c) 40-60 M, d) 60-80 M, and e) 80+ M. Species-specific density is given parenthetically by stand type.

the trees were less than 42 to 74 in DBH (*fig. 2a* to *2e*). The values of both the median and 90 percent DBH increased with economic type. The similarity between the cumulative DBH curves for redwood and all conifers (*fig. 2a* to *2e*) displays how the former dominates this metric of the stand condition.

Basal area ranged from 116 to 536 ft^2/ac, increasing substantially with an increase in economic type. Redwood comprised about 73 percent of the basal area in the lowest two economic types, 85 percent in the 40-60 and 60-80 economic type, and 97 percent in the 80+ type. Compared to the cumulative percent of trees by DBH class, the curves for cumulative percent of basal area by DBH class (*fig. 3a* to *3e*) portrays relatively “s” shaped curves. The trace of redwood cumulative basal area by DBH class was most similar to that for all trees, showing the dominance of redwood as a determinant of basal area in these forests. The DBH of median basal area ranged from 36 to 66 in, while that at which 90 percent of the basal area was less ranged from 61 to 98 in. Both increased with economic type.

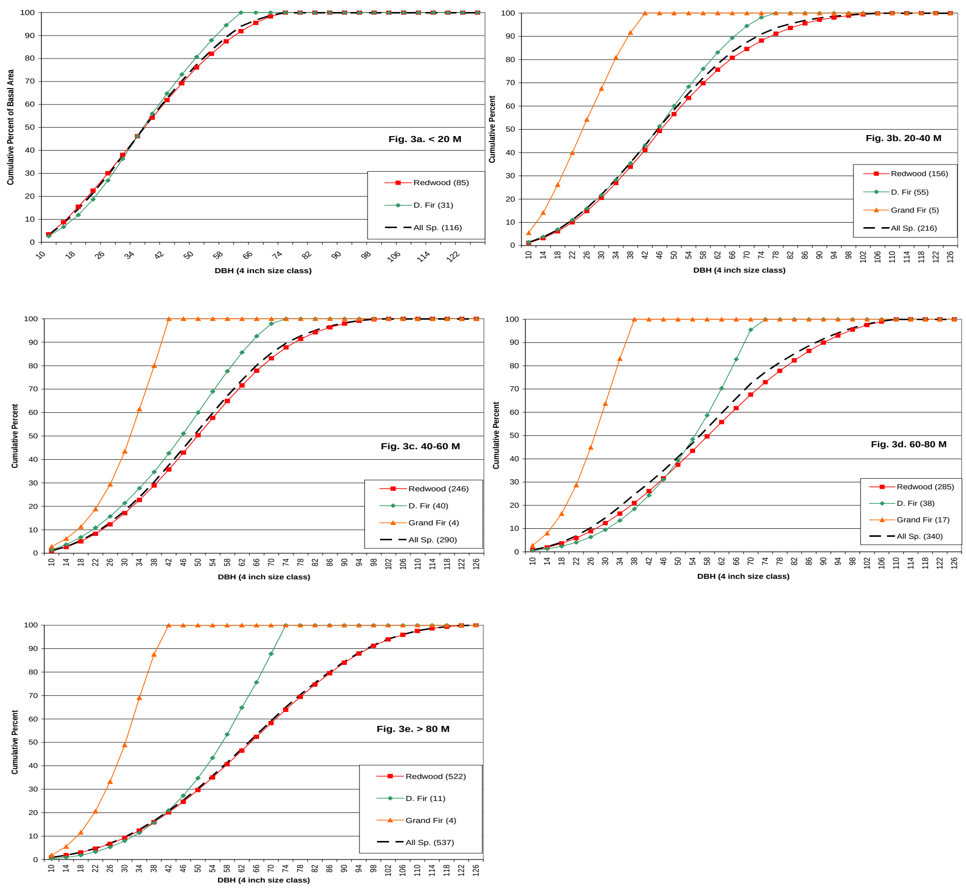

Figure 3—Cumulative percent of conifer basal area by diameter class and species in virgin forests of the Caspar Block, 1929; stand types a) <20 M, b) 20-40 M, c) 40-60 M, d) 60-80 M, and e) 80+ M. Species-specific total basal area is given parenthetically by stand type.

Discussion

Metrics associated with tree size are convenient for describing stand conditions. Because this information is important in assessing the economic value, the methods have a long history and common understanding among various users. In their literature review of coast redwood forest disturbance dynamics, Lorimer et al. (2009) characterize northern forests from several studies. Veirs (1982, in Lorimer et al. 2009) evaluated nine 2.53 ac plots in near-coastal, intermediate, and xeric slope forests near Redwood National Park. He reported the minimum size to reach canopy status to be 23.6 in, and a mean of 27.7 canopy trees/ ac (range 22.9 to 38.4 > 23.6 in). Redwood comprised 41 to 95 percent of the canopy trees. Basal area averaged 689 ft^2/ac (range 455 to 1085 ft^2/ac) with redwood accounting for 43 to 97 percent. Lorimer et al. (2009) note that of Veirs' nine sites, only two or three had size distributions that approximate a "negative exponential or rotated sigmoid curve" typical of uneven-aged, old-growth forests. This divergence from the expected size distribution is not explained for the mesic, near-coast sites; but they suggest upland sites may depart due to climatic shifts, limited sample size – and perhaps most importantly – moderate disturbance. The presence of Douglas-fir, a species requiring fire-related disturbance for recruitment was presented as evidence for the disturbance. Sample size is a likely difference between the clear inverse J-shaped curves for each economic type at Caspar and those reported by Veirs. While lots differed in size between the studies, the apparently more numerous plots in the Caspar Block data should have averaged out localized disturbance-induced variation that might overwhelm patterns at few sites.

Dagley and O'Hara (2003) summarize a number of studies on the density of trees in unharvested stands, and found the density is highly variable across the redwood range. They note that differences in methods and definitions complicate comparison. Based on upper canopy trees, the density ranged approximately from 20 to 150 trees/ac, the majority being redwood. Direct comparison of these values to the Caspar Block data is limited. The preponderance of the summarized studies sites were at Bull Creek and Redwood National Park in the northern part of the range while central Mendocino County is in the central part of the redwood forest's range (Sawyer et al. 2000). Also, while considering only canopy tree density may have ecological relevance, determining what constitutes a canopy tree needs clarification. The studies reviewed by Dagley and O'Hara (2003) used unspecified or different DBH criteria. Still, the cumulative distribution graphs (*fig. 2a* to *2e*) can facilitate enumeration by whatever DBH class is relevant.

Giusti (2007) presents historic inventory data from lands that became Redwood National Park in Del Norte County. As presented (*fig. 5* of Giusti 2007), unharvested redwood from ARCO, Simpson, and Rellim Timber Companies consistently present an inverse J-shaped curve. Although Giusti (2007) did not fully describe the methods used in these historic inventories, they likely included averaging a large number of samples including upland and riparian areas over the 9,000 ac cruised. In addition, Giusti (2007) also tabulates tree density data from Redwood creek, Little Lost Man Creek, and Bull Creek in Humboldt County. Respectively, trees greater than 40 in averaged 11.3, 12, and 35 trees/ac. The density of trees in the Caspar Block that exceeded this size class ranged from 3 to 18 trees per ac for the < 20M and the 80+ M types, respectively (*fig. 2a* to *2e*).

Hardwood associates in redwood/Douglas-fir forest likely common in the Caspar Block area include tanoak (*Lithocarpus densiflorus*), madrone (*Arbutus menziesii*), bigleaf maple (*Acer macrophyllum*), California bay (*Umbellularia californica*), and golden chinquapin (*Castanopsis chrysophylla*). These species, though of limited economic value have great ecological value. They are often overlooked in timber inventories, even though they may constitute a substantial component of forests. As for conifers, the hardwood content and stand structure of future stands is a direct outcome of present management (Harrington and Tappeiner 2009). Managers who focus on developing large conifers without simultaneously planning for a substantial hardwood component when developing restoration treatments may fail to achieve the broad ecological values of late-seral forests. Unfortunately, information is limited on the hardwood component of unharvested conifer stands upon which to evaluate either the range of natural variation or management goals.

In northwestern California, Bingham and Sawyer (1992) found the density of hardwoods and conifers to be 82 and 24 per ac, respectively, in old stands. Diameters of hardwoods averaged 15 in, while conifer DBH averaged 38 in. Using average density and DBH, the study concluded that the hardwood basal area of the old stands approximated 50 ft^2/ac, just over 20 percent of the total basal area.

The most comparable data from the central portion of the Redwood range comes from Montgomery Woods State Reserve (Giusti 2007). Data on tree species inclusive of hardwoods, size, and habitat elements were analyzed from an average of four 0.1 ac plots at 98.4 ft spacing along 14 parallel transects that crossed the alluvial plain and extended into the adjacent upland. These plots collectively characterize a single stand. Unlike the size distributions observed on the Caspar Block, the plot for Montgomery Woods displays a flat, inverse J-shaped curve with redwood and Douglas-fir about equally dominant within each of the 10 in size classes. Twenty one trees/ac exceed 40 inches, with only one tree/ac – a redwood – exceeding 80 in. The flatness of the Montgomery Woods size distribution lead Giusti to conclude that recruitment may be limited on the site. Redwood dominated with nearly 60 percent of the large conifer density, Douglas-fir had the remainder. Importantly, Giusti (2007) found that by density, tan oak made up 85 percent of the less than 10 in DBH size class, 89 percent of the trees in the 11 to 20 in DBH class, and 7 percent of the > 20 in size class. Giusti (2007) does not report basal area directly, but it can be approximated from his data. Using size class mid-points, the stocking of all tanoaks was 83 ft^2/ac, and those greater than 10 in DBH accounted for 49 ft^2/ac; approximately 15 and 10 percent of the stands total basal area. The Montgomery Woods stand data indicate that both quantitatively and proportionately, tanoaks are a heavy and dominant understory component. The degree to which effective fire suppression since the early 20th century affects the species mix, especially in the smallest size class, is unknown.

The variation among the Caspar Block's economic types may be due to many factors. The absence of plot-specific information prevents statistical analyses of differences between the types or the variance within types. Although the stands were virgin timber, the time elapsed since their last stand-replacing event may be variable; that is, their relative position along the successional continuum is unknown. The species mix may reflect topographic position, with redwood dominant on alluvial sites and a broader mix of conifer species upslope (Lorimer et al. 2009). Topography may well be the driver of processes that acts on tree physiology and disturbance dynamics to yield the differences observed in species composition and biomass across stands types. Topography constrains available moisture and soil characteristics. Both within and among economic class variation can result from disturbance dynamics that vary by topography. Localized disturbance events such as individual tree fall can lead to substantial variation if sample size is few or the area sampled is small relative to the tree fall impact area. Because succession within late seral forest when scaled to large areas likely proceeds towards an asymptote over long periods, the points in the successional continuum at which a stand is studied can lead to among stand variation. Analysis of the map associated with the inventory (Anon. 1929) may help describe topographic relationships with the economic types.

Stand structure that can be useful for describing late-seral management conditions and goals. However, the Caspar Block and Montgomery Woods studies are too few and of limited geographic range to have confidence in the amount of variation that may exist within regional unmanaged late-seral stands. Publication of historic inventories, despite their probable omission of non-conifer information could yield more useful descriptions of late seral structure. Collecting more comprehensive information on extant stands, while possibly being obscured by decades or more of fire exclusion can also broaden the picture of unharvested stand structures.

Tree size and density are in many ways a caricature of late-seral conditions. Simplistic stand tables fail to recognize or quantify the elements more important to a late-seral forest community. These elements include substantial amounts of the products that accumulate over time: tree decline, decadence, death such as snags (Giusti 2007) and logs (Bingham and Sawyer 1988, Porter and Sawyer 2007); mechanical and fire-induced disturbance such as reiterated branches (Sillett and Van Pelt 2000) and basal hollows (Finney 1996); establishment of epiphytes (Sillett and Van Pelt 2007); and late-seral associated species (Russell and Michels 2010). Forest-type and regional quantification of the elements at the scale of a stand will add substantially to tree size and density criteria to assure that late-seral management achieves ecological goals.

Acknowledgments

I would like to thank Marc Jameson, retired Forest Manager at Jackson Demonstration State Forest for surfacing and making this data set available to me. Also, Tony LaBanca encouraged me to publish this data through allotting work time.

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Structure and Dynamics of an Upland Old-Growth Forest at Redwood National Park, California

Phillip J. van Mantgem[1] and John D. Stuart[2]

Abstract
Many current redwood forest management targets are based on old-growth conditions, so it is critical that we understand the variability and range of conditions that constitute these forests. Here we present information on the structure and dynamics from six one-hectare forest monitoring plots in an upland old-growth forest at Redwood National Park, California. We surveyed all stems ≥20 cm DBH in 1995 and 2010, allowing us to estimate any systematic changes in these stands. Stem size distributions for all species and for redwood (*Sequoia sempervirens* (D. Don) Endl.) alone did not appreciably change over the 15 year observation interval. Recruitment and mortality rates were roughly balanced, as were basal area dynamics (gains from recruitment and growth versus losses from mortality). Similar patterns were found for *Sequoia* alone. The spatial structure of stems at the plots suggested a random distribution of trees, though the pattern for *Sequoia* alone was found to be significantly clumped at small scales (< 5 m) at three of the six plots. These results suggest that these forests, including populations of *Sequoia*, have been generally stable over the past 15 years at this site, though it is possible that fire exclusion may be affecting recruitment of smaller *Sequoia* (< 20 cm DBH). The non-uniform spatial arrangement of stems also suggests that restoration prescriptions for second-growth redwood forests that encourage uniform spatial arrangements do not appear to mimic current upland old-growth conditions.

Key words: forest dynamics, forest structure, old-growth, spatial pattern

Introduction

Remnant old-growth stands represent only a small fraction of current redwood forest area (Sawyer et al. 2000a), but play crucial roles in conservation and restoration of these ecosystems. As our best analogue to redwood forest conditions prior to the arrival of industrialized human settlement, remnant old-growth redwood stands are used as reference conditions for managers seeking to accelerate the development of second-growth forests (e.g., NPS 2008). In spite of their importance, relatively little is known about the basic structure and function of old-growth redwood forests (Busing and Fujimori 2002, Lorimer et al. 2009). Part of the difficulty is that old-growth conditions are relatively difficult to describe; by definition old-growth forests contain trees from a wide range of ages and sizes, with areas of closed canopy and open gaps (i.e., high spatial complexity) (Franklin et al. 2002). This structural complexity is magnified in old-growth redwood forests, which contain some of the oldest and largest trees known (Sillett et al. 2010).

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Further complications arise from the fact that old-growth redwood forest conditions vary across topographic gradients, from alluvial terraces to steep hillsides and ridgetops. Redwood forests on alluvial flats are comprised of almost pure stands of redwood *Sequoia sempervirens* (hereafter *Sequoia*) while upland stands are composed of a mix of species including *Sequoia*, Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirb.) Franco), tanoak (*Notholithocarpus densiflorus* syn. *Lithocarpus densiflorus*) and other species (Sawyer et al. 2000b). Redwood stands on alluvial terraces and upland stands experience contrasting disturbance regimes, with stands found on alluvial areas subject to periodic inundation and deposition following floods, while upland stands are more prone to fire (and to a lesser extent landslides and windthrow) (Lorimer et al. 2009). There is some concern that *Sequoia* may not be able to maintain dominance in the absence of fire, although Busing and Fujimori (2002) show that at an alluvial habitat *Sequoia* populations have been stable. It is unclear how well this finding may hold in upland habitats where historically fires were more common (Lorimer et al. 2009).

The spatial structure of stands is another important, but undocumented, potential difference between alluvial and upland habitats. The spatial arrangement of stems (the degree to which the distribution of trees in a stand can be considered clumped, random or uniform) is a defining structural element of stands, determining local competitive environments (Biging and Dobbertin 1992, Das et al. 2008). Recently, Dagley (2008) described old-growth alluvial redwood forest spatial structure as clumped to random for overstory forest trees. Establishing if non-random spatial structure is also common in upland stands is highly relevant for redwood forest restoration; many thinning prescriptions for second-growth forest restoration may create uniform conditions (NPS 2008), which may persist for decades, but may or may not mimic old-growth forest spatial patterns.

Here we present information on forest structure and dynamics from six upland forest stands in Redwood National Park. We analyze differences in forest structure among our sites in terms of species composition, stem size distributions and the spatial arrangement of stems. Our information is derived from repeated surveys of these sites, so we also are able to directly calculate key demographic rates, including recruitment, growth and mortality. Our results largely confirm earlier findings from alluvial forests (Busing and Fujimori 2002, Dagley 2008), in that redwood forest populations appear to be generally stable and forest spatial patterns are not uniform.

Methods

Study sites

The region around Redwood National Park features a Mediterranean climate, with mild, rainy winters and cool, dry summers (Sawyer et al. 2000b). Annual mean temperatures are approximately 15 °C, with annual precipitation of about 170 cm, 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, but has been largely excluded in the region of our study sites over the past 100 years (Ramage et al. 2010).

We surveyed six 1 ha plots to determine forest structure in an upland redwood forest (*fig. 1*). The sites have never been logged. Frequent fires characterized the forests prior to Euro-American settlement, but there is no recent record of fire in the areas containing the study plots. The plots were established in 1995 to compare forest structure in these old-growth areas to an adjacent second-growth stand known as the Whiskey Forty (Teraoka and Keyes 2011). Surveys mapped all stems ≥20 cm diameter at breast height (DBH, 1.37 m), measured tree diameters, and assessed tree status (live vs. dead) (*table 1*). In 2010 we surveyed these stands again for trees ≥20 cm DBH, recording current DBH, mortality (of trees extant in 1995), and recruitment (trees now ≥20 cm DBH but not recorded in 1995). Radial growth was determined from repeated stem diameter measurements. There were 13 instances of negative growth and 21 instances of unrealistically large radial growth (>15 mm year^{-1}), which probably arose from measurement error. We wished to remove outliers with exceptionally large negative and positive growth rates, but did not wish to bias our results in a positive direction by removing all negative growth rates. For this reason we removed from analysis stems that had radial growth rates < -2 mm year^{-1} or >15 mm year^{-1}, roughly 1 percent of our observations.

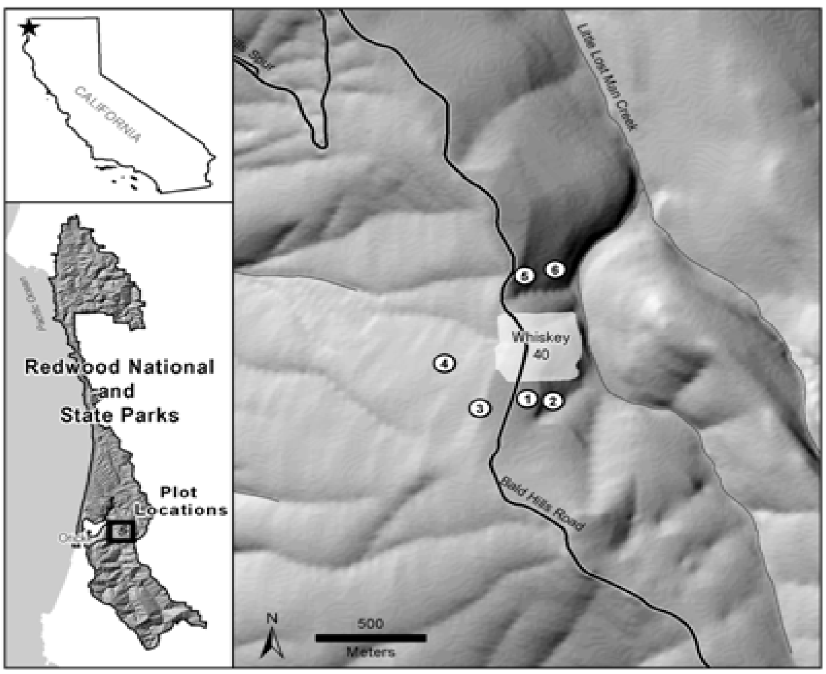

Figure 1—Location of the six upland old-growth redwood forest plots. The plots are located near an area of second-growth forest (locally known as the “Whiskey 40”), but are in a large, otherwise undisturbed old-growth forest.

Data analysis

We determined forest structure in terms of stand density (stems ha^{-1}), basal area (m^2 ha^{-1}), and size class distributions. We measured patterns of stem size distributions at each plot using a departure index, M, that is similar to the Gini coefficient, but can distinguish both the magnitude and direction of change (Menning et al. 2007). The departure index M was calculated for 1995 versus the 2010 censuses, and compared against a null model that assumed no

Table 1—*Characteristics of forest monitoring plots at the time of establishment in 1995 (stems >= 20 cm DBH).*

Plot identifier	UTM* Easting	UTM Northing	Elevation (m)	Stem count (ha) 1995	Stem count (ha) 2010	Basal area (m^2 ha^{-1}) 1995	Basal area (m^2 ha^{-1}) 2010
Plot 1	415657	4570743	472	230	244	174	165
Plot 2	415776	4570731	446	185	175	163	176
Plot 3	415440	4570690	460	134	131	205	231
Plot 4	415276	4570941	403	153	163	133	136
Plot 5	415645	4571428	461	185	193	138	146
Plot 6	415786	4571464	412	212	183	198	195

Table 1 (continued)

Plot identifier	Species comprising > 1% of stem counts**
Plot 1	TSHE 42%; SESE 27%; PSME 22%; LIDE 8%
Plot 2	TSHE 42%; SESE 29%; PSME 25%; LIDE 3%; ABGR 1%
Plot 3	SESE 46%; ABGR 21%; TSHE 16%; PSME 10%; LIDE 8%
Plot 4	SESE 38%; LIDE 27%; ABGR 17%; TSHE 13%; PSME 5%
Plot 5	LIDE 30%; SESE 29%; PSME 25%; TSHE 16%
Plot 6	SESE 37%; TSHE 37%; PSME 23%; LIDE 3%

* UTM coordinates in NAD 1983, zone 10.
** At time of plot establishment. ABGR = *Abies grandis* (Dougl. ex D. Don) Lindl.; LIDE = *Notholithocarpus densiflorus* syn. *Lithocarpus densiflorus*; PSME = *Pseudotsuga menziesii* var. *menziesii* (Mirb.) Franco; SESE = *Sequoia sempervirens* (D. Don) Endl.; TSHE = *Tsuga heterophylla* (Raf.) Sarg. Percentages may not add to 100 due to rounding.

differences between these measurement intervals. Differences in M between the observed values and null model were compared using a Wilcoxon signed rank test. We compared demographic rates (recruitment and mortality) as well as basal area dynamics (basal area gains from recruitment and growth versus basal area losses from mortality) using paired randomization tests to account for small sample sizes and non-normal distributions (Manly 2001).

We described tree spatial distributions within each plot by considering individual trees as points, and examined nearest neighbor patterns at multiple scales using the inhomogeneous K-function, a second-moment measure that accounts for variability in the average density (also called intensity) of trees across each plot (Baddeley 2008). Intensity is the expected number of points per unit area, and may be constant (homogenous) or vary across locations (inhomogeneous). Plots of kernel estimates of intensity suggested high variations in intensity, suggesting that inhomogeneous models were more appropriate for our data. For each plot we considered spatial patterns within a search radius of up to 25 percent the side of the plots, 25 m. We determined the significance of the departure from complete spatial randomness (CSR) by comparing observed values to the 95 percent simultaneous critical envelope (Baddeley 2008) [avoiding inflation of type I errors (Loosmore and Ford 2006)], calculated from 999 simulations of random point fields, with intensity patterns estimated using a kernel smoothing function. Results are presented in terms of the normalized K statistic, L(h) (where h represents the search radius), which is centered on zero. L(h) values within the 95 percent critical envelope are consistent with a random distribution of stems, while values below the critical envelope are considered significantly uniform, and values above the critical envelope are considered significantly clumped.

We used a marked point process model to directly test if *Sequoia* has a different spatial structure compared to other co-occurring species, using the inhomogeneous K-function normalized at zero [i.e., $L_{Sequoia,\,\mathrm{Other}}$ (*h*)]. We created a 95 percent simultaneous critical envelope from 999 simulations using random toroidal shifts of the pattern for each taxonomic group, where observed values within the envelope indicate similar distributions, values below the envelope indicate the species groups "repulse" each other in space, values above the envelope indicate "attraction" in the groups' spatial pattern.

Results

At the time of plot establishment in 1995 stem density for trees ≥20 cm DBH ranged from 134 to 230 trees ha^{-1} and averaged 183 trees ha^{-1} (±2 SE = 29) and basal area ranged from 133 to 205 m^2 ha^{-1} and averaged 169 m^2 ha^{-1} (±2 SE = 24) (*table 1*). Although *Sequoia* comprised on average only a small proportion of stems at each plot (mean ± 2 SE = 61 ± 8 trees ha^{-1}, 33 percent of stems), this species comprised the majority of average basal area (mean ± 2 SE = 100 ± 30 m^2 ha^{-1}, 59 percent of total average basal area). There was little change in density or basal area for trees ≥20 cm DBH from 1995 to 2010 (*table 2*).

At the time of plot establishment in 1995 stems >20 cm DBH generally followed a negative exponential ("inverse-j") size class distribution (*fig. 2A*). Stem size classes were similar in 2010, so that the negative exponential distributions were maintained (*fig. 2B*). The departure index M was relatively unchanged over this time period (average M = + 0.002, average range of M = -0.388 to 1.612), and was not significantly different compared against a null model of no change (Wilcoxon signed rank test, P = 1). For *Sequoia*, there was a suggestion that stems size distributions were becoming more skewed in favor of large stems from 1995 to 2010 (average M = + 0.044, average range of M = -0.611 to 1.389, Wilcoxon signed rank test, P = 0.049), but changes were small (*fig. 2*).

The relative lack of change in stand structure from 1995 to 2010 was also supported by demographic measurements. Plot-level average recruitment and mortality rates were balanced (annualized recruitment rate = 0.700 percent, annualized mortality rate = 0.698 percent, paired randomization test, 9999 iterations, P = 0.46) (*table 2*). The demographic rates for *Sequoia* also were essentially equivalent (annualized recruitment rate = 0.229 percent, annualized mortality rate = 0.325 percent, paired randomization test, 9999 iterations, P = 0.77). The dynamics of basal area from 1995 to 2010 did not suggest substantial changes; annualized basal area gain from recruitment and growth (average = 0.98 m^2 ha^{-1} $year^{-1}$) was roughly balanced by losses from mortality (average = 0.80m^2 ha^{-1} $year^{-1}$) (paired randomization test, 9999 iterations, P = 0.26) (*table 2*). Similar patterns in basal area dynamics also were found for *Sequoia* (average basal area gain from recruitment and growth = 0.50 m^2 ha^{-1} $year^{-1}$; average basal area loss from mortality = 0.38 m^2 ha^{-1} $year^{-1}$; paired randomization test, 9999 iterations, P = 0.35) (*table 2*).

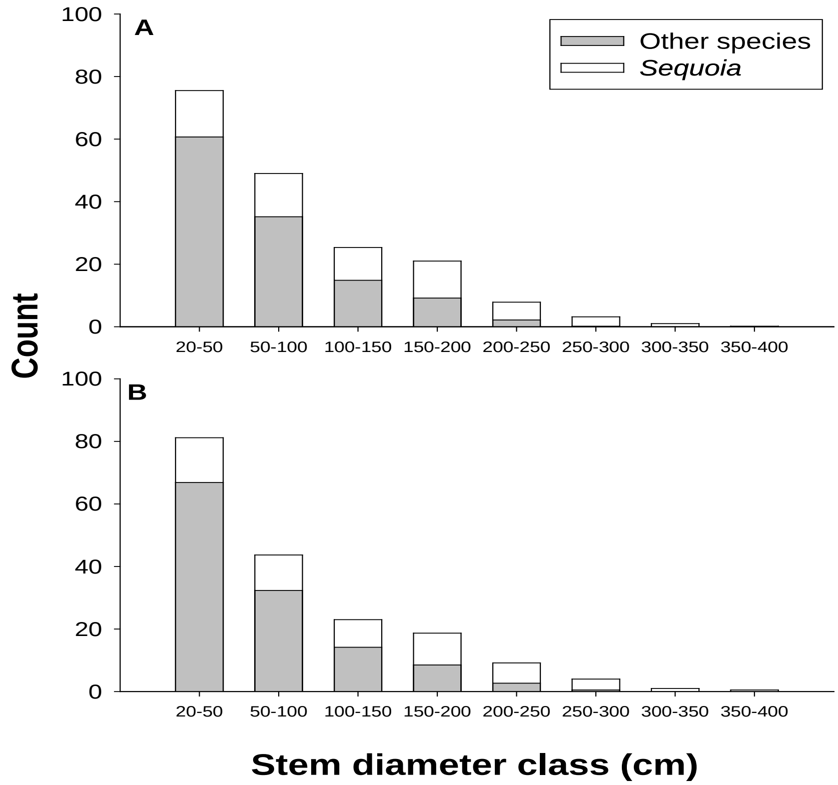

Figure 2—Stem size class distributions averaged across six 1-ha plots for trees ≥20 cm DBH in 1995 (A), and in 2010 (B).

Contour plots of the spatial intensity showed that the density of stems was highly variable across each plot (not shown), indicating the need to use the inhomogeneous measures of spatial pattern. The inhomogeneous L(*h*) statistic showed a random arrangement of stems at most spatial scales across all plots (although a single plot showed a minor degree of clumping at small scales) (*fig. 3A*). Estimates of the spatial pattern of *Sequoia* alone suggest significant clumping at small scales for three of the six plots (*fig. 3B*). Direct comparison of *Sequoia* versus other co-occurring species suggests that these groups are strongly segregated at spatial scales greater than approximately 10 m (*fig. 3C*).

Table 2—*Annualized demographic rates and basal area dynamics (gains from recruitment and growth and losses from mortality) from 1955 to 2010 for trees >20 cm DBH. Demographic rates were roughly balanced (P = 0.46), as were losses and gains in basal area (P = 0.26) (paired randomization tests). Similar patterns were found for Sequoia. Confidence intervals were determined from 1000 bias corrected bootstrapped samples.*

	Plot identifier	**Recruitment rate (%)**	**Mortality rate (%)**	**Basal area gain (m^2 ha^{-1} $year^{-1}$)**	**Basal area loss (m^2 ha^{-1} $year^{-1}$)**
All species	Plot 1	1.02	0.73	1.02	1.57
	Plot 2	0.35	0.64	1.04	0.36
	Plot 3	0.53	0.46	1.23	0.17
	Plot 4	1.01	0.59	1.02	0.82
	Plot 5	0.98	0.60	0.75	0.48
	Plot 6	0.31	1.16	0.78	1.41
	average	***0.70***	***0.70***	***0.98***	***0.80***
	95% CI	*0.46 to 0.94*	*0.58 to 0.99*	*0.85 to 1.1*	*0.41 to 1.28*
Sequoia only	Plot 1	0.11	0.67	0.42	0.40
	Plot 2	0.00	0.26	0.44	0.20
	Plot 3	0.11	0.11	0.92	0.02
	Plot 4	0.66	0.35	0.64	0.66
	Plot 5	0.25	0.13	0.21	0.26
	Plot 6	0.25	0.44	0.36	0.77
	average	***0.23***	***0.33***	***0.50***	***0.38***
	95% CI	*0.1 to 0.43*	*0.19 to 0.5*	*0.35 to 0.71*	*0.19 to 0.61*

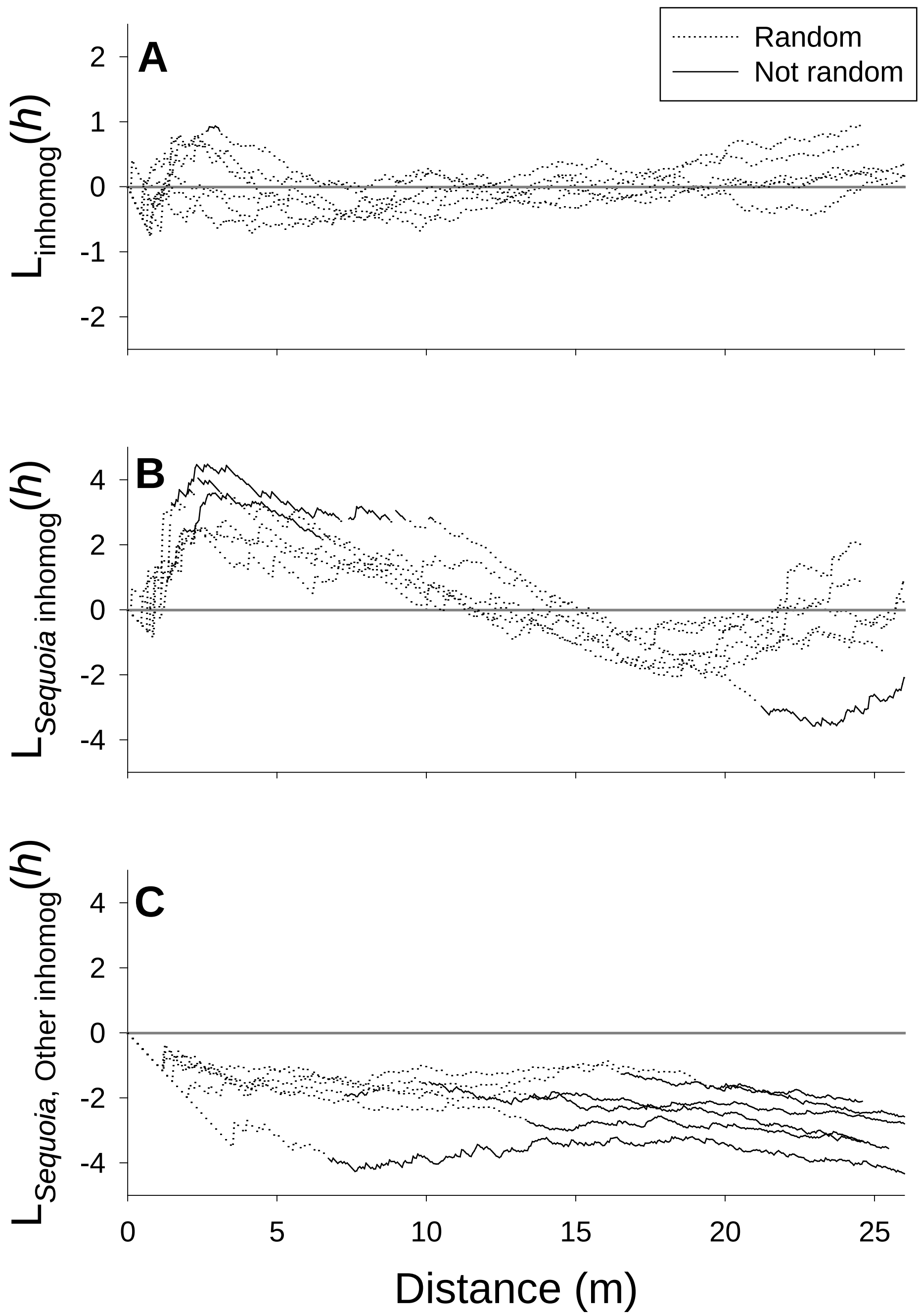

Figure 3—Tree spatial pattern for all species at six upland old-growth redwood stands (A), the pattern for *Sequoia* alone (B), and *Sequoia* versus all other species (C). Lines are the observed normalized K-function [L(*h*)] for individual plots. Dashed intervals represent random distributions and solid intervals above and below the zero control line indicate statistically significant (α = 0.05) clumping or repulsion, respectively. Significance was determined by departures from 95 percent simultaneous critical envelopes (Baddeley 2008).

Discussion

Our old-growth redwood stands appeared to be generally stable from 1995 to 2010, a finding in agreement with earlier observations from an alluvial terrace (Busing and Fujimori 2002). Stem size class distributions generally followed an "inverse-j" distribution, which suggests stable populations in old-growth stands (Avery and Burkhart 2002). Stable stem size distributions and direct measurements of forest dynamics suggest that these stands did not experience large changes over the 15 year observation interval.

There has been speculation that in the absence of moderate disturbance from fire *Sequoia* might fail to maintain its dominant status in these communities (Lorimer et al. 2009), but our observations failed to provide clear support for this idea. We found some evidence that *Sequoia* stem size distributions may have become less dominated by small trees over time, but recruitment and mortality rates for *Sequoia* were roughly balanced, as were gains and losses in basal area. However, our measurements of recruitment include only stems that were ≥ 20 cm DBH, so it is possible that we are missing early signals of reproductive failure in the absence of fire (Ramage et al. 2010). More detailed observations of small tree dynamics are needed to fully document the reproductive output of *Sequoia* (whether by seed or sprouting) in upland old-growth forests under current conditions. Moreover, the 15 year length of observations presented here is short in comparison to ages of dominant trees in our study site. It may be that large changes in stand and population dynamics occur infrequently or will only be detectable over longer time scales.

In terms of spatial structure, with very little exception we found a random arrangement of stems across all spatial scales at all six upland stands. This finding was promoted by the use of the inhomogeneous formulation of the K-statistic and a more statistically defensible method of determining statistical significance, resulting in tests that were more conservative and less likely to show significant clumping or uniformity. In spite of this, we still identified strong evidence of clumping at small spatial scales (from approximately 1 to 5 m) for *Sequoia* at half of the plots, an unsurprising finding given the sprouting habit of this species. Moreover, there appeared to be segregation between *Sequoia* and other species at large (approximately > 10 m) spatial scales. Thus, the general conclusions of Dagley (2008) concerning the restoration targets for alluvial stands appear to hold for upland stands: thinning prescriptions that encourage uniform spatial arrangements of stems do not appear to mimic current old-growth structure.

How well current old-growth conditions reflect stand structure prior to European settlement is difficult to determine precisely (Stephenson 1999). The disruption of historic disturbance regimes, exotic species (including pathogens), and other changes have likely altered these forests. But until a better model for restoration targets is identified, these stands provide our best understanding of reference conditions. Continued measurement and monitoring of old-growth upland stands is needed to insure we are able detect any subtle, pervasive trends that may be occurring (van Mantgem et al. 2009).

Acknowledgments

We thank Jon Hollis, Laura Lalemand and Jon Chin for their assistance in the field and Janelle Deshais for creating the plot map. Funding for this project was provided by the National Park Service. Any use of trade names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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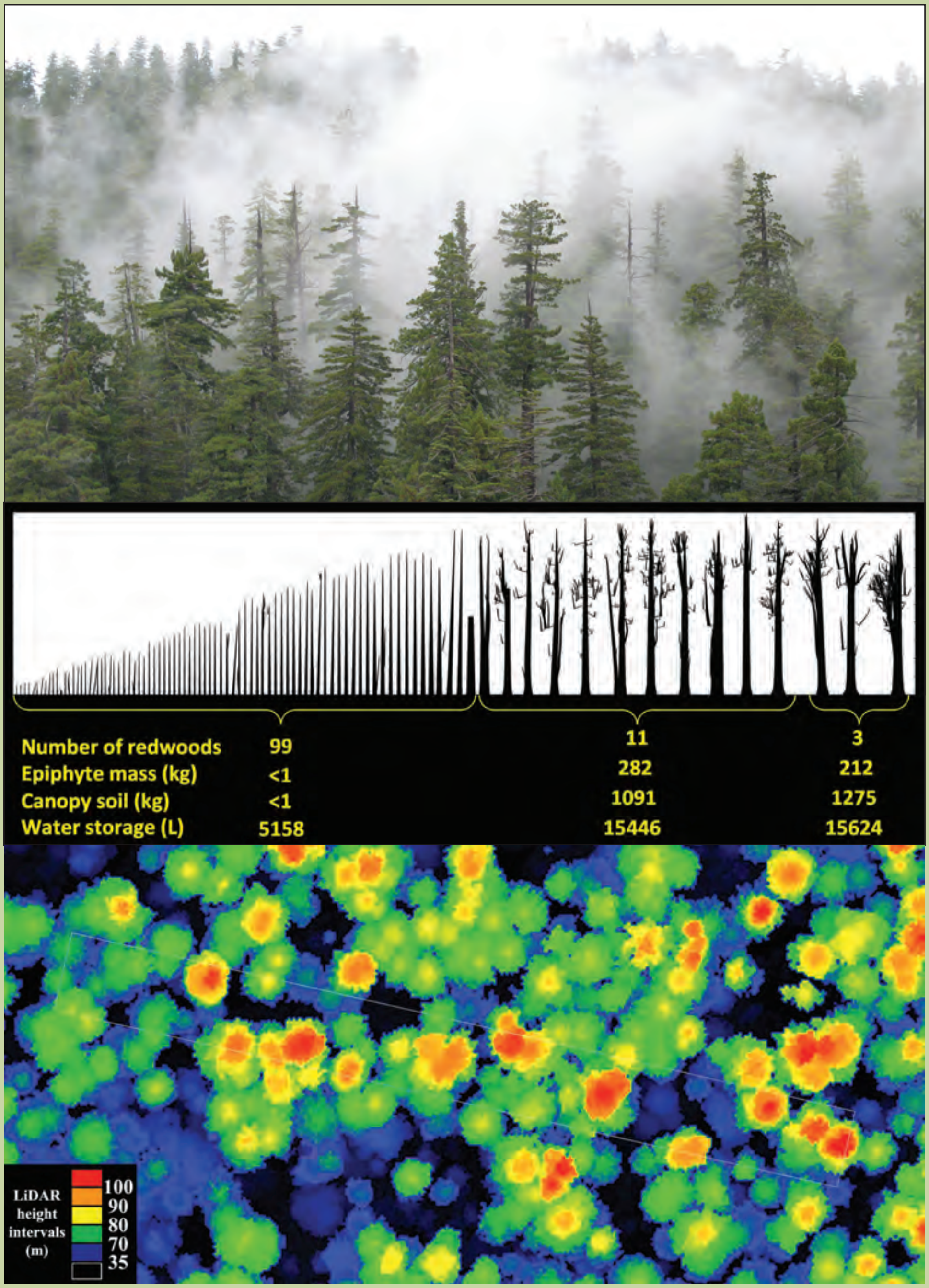
Forest Health



Damage and Mortality Assessment of Redwood and Mixed Conifer Forest Types in Santa Cruz County Following Wildfire

Steve R. Auten[1] and Nadia Hamey[1]

Abstract
On August 12, 2009, the Lockheed Fire ignited the west slope of the Santa Cruz Mountains burning approximately 7,819 acres. A mixture of vegetation types were in the path of the fire, including approximately 2,420 acres of redwood forest and 1,951 acres of mixed conifer forest types representative of the Santa Cruz Mountains. Foresters and land managers were left with tough decisions on how to treat tree damage and mortality compounded by the Pine Mountain Fire which occurred in the same area in 1948. Big Creek Lumber Company (BCL), Cal Poly's Swanton Pacific Ranch (SPR) and other professionals familiar with this region of redwood teamed up to develop a method for evaluating damage and mortality. Qualitative criteria for evaluating stand damage focused on historic defect, cambial death, root damage, and associated fire intensity. Quantitative damage criteria was used to contrive three mortality assessment tables, broken up by diameter class (1 through 8, 9 through 16, 17+), for all tree species and tested against 83, 1/5th acre fixed plots from SPR's Continuous Forest Inventory. Since the initial mortality evaluation using the new tables in fall of 2009, each of the 2877 trees have been re-evaluated in spring 2010 and spring 2011. Accuracy against the initial evaluation is 89.3 percent.

Key words: damage, hardwood, mortality, redwood

Introduction

What should be harvested to encourage regeneration of selectively-managed forestland in the Southern Subdistrict of the Coast Forest District following wildfire? What determines tree mortality for the purpose of amending the sustainability analysis (SA) of a Non-industrial Timber Management Plan (NTMP) following wildfire? Charged with managing and maintaining the health and vigor of the forest ecosystem, foresters and land managers need an accurate way of field-evaluating damage and mortality in conifers and hardwoods immediately following wildfire. These questions and many others not related to forest health, loomed for the local forestry community following the Lockheed Fire. This paper is a case study to provide other foresters and land managers with information and knowledge gained through our experiences following the Lockheed Fire to help make decisions about damage and mortality in conifers and hardwoods as a result of wildfire.

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Seasonal context

The time of year and prevailing wind direction during the Lockheed Fire was very similar to conditions during the Pine Mountain Fire, which is the last recorded and mapped fire perimeter within the Lockheed Fire area. The Pine Mountain Fire burned in September 1948, 61 years ago, and consumed 15,899 acres. The Lockheed Fire also took place in late summer, a season when historically lightning strikes have caused fires in the Santa Cruz Mountains; although the Lockheed Fire was human-caused. Other large historic fires are known to have occurred in the vicinity of the Lockheed Fire near the turn of the 19th century associated with logging and power generation projects in the Scotts Creek watershed.

Damage assessment

The land management goals of the large Timber Production zoned parcels are for timber production and protection of associated resources such as wildlife, fisheries, and watershed. The mission of the landowners affected by this fire was to determine the level of defect and remove some of the most damaged wood to bring defect down to an acceptable level for future management.

Consider the accounting for a timber harvest conducted a few years ago in an area heavily hit by the Pine Mountain Fire, where the defect was 23 percent, about 11 percent higher than "average" for the Santa Cruz Mountains. The loss of net return to the landowner was about $71,500 for approximately 1 million board feet harvested.

Fire intensity

In order to understand burn intensity patterns, fire behavior was estimated for the day the fire started under the prevailing weather and fuel conditions. Five different fuel models, representing five vegetation types, were used in the BehavePlus computer simulation to estimate fire behavior. The BehavePlus fire modeling system is a collection of models that describe fire behavior, fire effects and the fire environment.

Burn severity analysis was conducted initially by a Risk Assessment Team put together by Cal-Fire. The team used a Burned Area Reflectance Classification (BARC) satellite-derived map of post-vegetation condition made by comparing satellite near and mid infrared reflectance values. This data was evaluated in several ways. Some team members flew over the fire area on the first clear day and photographed post-fire conditions. Other team members made ground-based observations over the fire area while assisting with suppression and suppression repair efforts. Burn severity across the entire burn area was estimated to be 14 percent very high, 37 percent high, 43 percent moderate, and 6 percent low.

Research

Foresters working in the burned area began researching what to expect from the various levels of burn damage. Little research or literature is available relative to wildfire impacts on redwood defect and mortality, especially in this region. Several papers about fire damage and anticipated mortality from prescribed burns in mixed-conifer forests were helpful. One paper emerged with practical insights on tree mortality and wildfire from Willis W. Wagener in 1961 (Wagener 1961), called "Guidelines for Estimating the Survival of Fire Damaged Trees in California."

Following a literature review, many conversations occurred with other foresters and land managers throughout the state to learn from their experiences. Contacts included:

- Mike Jani and John Anderson from Mendocino Redwood Company, with experience the year prior conducting salvage operations on the Lightning Complex Fire.
- Lathrop Leonard from California State Parks with experience in prescribed burning.
- Rich Casale, District Conservationist, USDA Natural Resources Conservation Service.
- David Van Lennep and Mike Duffy from Redwood Empire Sawmills. We toured the Summit Fire area that has been salvage logged by them the year prior.
- Dale Holderman, Chief Forester emeritus for Big Creek Lumber Company, with significant experience in defect levels in the Santa Cruz Mountains.

Repeatedly other land managers stated that the salvage operations undertaken in burned areas did not remove enough of the damaged timber. Significant loss of future commercial value is expected as a result of the persistent fire-scar defect.

Cambium death, root damage and crown scorch

Determining the level of internal defect from the external visual indicators on the tree immediately post-fire was very difficult. The methodology evolved over the course of months investigating the extent of the burn and initiating salvage operations in the most severely burned areas, and is still ongoing. Foresters started out chipping into the bark of trees with a hatchet to see what types of bark characteristics are indicative of damage to the cambium layer. Immediately after the fire as the cambium was dying, that layer appeared very dry compared to live cambium which oozed resin. Dying cambium also sometimes appeared grey versus red or pink for live cambium. A particularly useful tool turned out to be delivering a sound kick to the tree. A hollow sound would indicate that the cambium had died and the wood separated from the bark.

On some trees bark burned completely through exposing the wood underneath. Other trees had cracks in the bark showing the wood indicating that the cambium had been burned. Other trees had hollowed-out bases where large quantities of roots had been burned. Reduced root mass affects the tree's growth rate and if the damage is substantial, it can affect the trees stability and slow the rate of recovery. Many scorched trees that experienced crown fire lost all of their needles during the first winter, which provided a good carpet of mulch on the ground.

Trees with cambium damage in multiple quadrants were considered to be substantially damaged, especially when occurring in conjunction with previous scars. Although redwood compartmentalizes rot as it grows over it, the dead cambium will rot wood interior to it. The effects of dead cambium introducing defect are immediate. As the water transport cells break down, the sapwood dries out and is therefore susceptible to dry rot and termite attacks.

As the trees began to grow, the cracks in the bark where cambium had burned began to separate. Now, 2 years after the fire, the bark plates have separated where cambium died and you can see the cambium growing over the dead wood layer and re-sealing the perimeter of the tree. Badly burned trees developed a variety of fungus and lichen on the scorched bark, giving clues about the extent of internal damage.

There was increased burn damage in certain environmental settings such as: where ladder fuels were present, next to a more flammable vegetation type, on ridges, in topographic chimneys, in dense stands of un-thinned trees and where there was a heavy duff layer, legacy stumps, or lots of downed wood. Canopy damage ranged from an un-phased crown to all but the biggest limbs completely consumed in a crown fire. Severely burned trees did not re-sprout from the limbs, but only from the bole, like a bottle-brush. Where new limbs or tops are formed, the dead wood will be subsumed in the new growth, creating a weak spot in the wood.

Most second growth redwood trees that have experienced an intense wildfire in the past have an interior cylinder of rot. Using the U.S. Department of Agriculture, Forest Service Log Scaling and Grading Rules, that defect is “squared out” in order to account for waste when manufacturing dimensional lumber. The quality of the wood going into the mill is very important for the end result of producing sound building materials. Compounded defect from multiple fires would severely impact the manufacturing of high grade lumber. Experience dictates that defect levels elevated above 20 percent have dramatic consequences for the net return of a timber harvest for the landowner.

Adaptive management

During the first fall after the fire, a small area (4 ac) was salvage logged with ground based equipment to see the extent of the burn damage. Initiating the salvage effort offered an opportunity for foresters to verify the external indicators used to determine defect. Once the trees were cut, the dead cambium sections were discernible in the cross-section because the usually fuzzy, fibrous inner bark becomes smooth and sometimes separated from the wood when it is dead.

The rot from the fresh fire scars had already started. Landowners wishing to grow sound trees for sustainable harvest had to react. More damage was sustained to the timber resource than was recouped in the salvage. Trees left with fire scaring and dead wood are more vulnerable to fire and insect attack going into the future. Due to the practice of leaving the healthiest residual trees to retain structure for wildlife and maintain microsite characteristics, many trees were left. It was not inexpensive to log, nor did it produce windfall profits. Compounding the loss, the low profitability in Douglas-fir markets necessitated that many thousands of dead Douglas-fir trees be left standing. Redwood seedlings were under-planted in some of these areas.

Helicopter salvage harvest

Multiple land managers elected to proceed with a salvage harvest by helicopter. Helicopter harvesting is extremely expensive, but avoids the ground disturbance of conventional harvest methods such as tractor logging. The salvage operation was a long-term land management decision to "cut the losses" and establish better wood quality in the future. Helicopter harvesting affects the bottom line, but weighing all decision-making factors, it made sense for the initial salvage effort due to the resource protection it affords as a result of the minimal ground impact. Other factors affecting the decision to fly included the relative inaccessibility of the ground, seasonal restrictions for sensitive species, the helicopters availability, and the desire to recover the damaged wood quickly.

Mortality assessment

The SPR NTMP was approved in June of 2008. Encompassed in this document is the sustainability analysis (SA) required by the Forest Practice Rules to demonstrate movement toward a fully regulated state of harvest over time. In essence, this means a "cut what you grow" type system focused on the long term sustained yield of forest products. Once the Lockheed Fire burned over the majority of the SPR NTMP area, the SA had to be re-evaluated. What determines tree mortality for the purpose of amending the NTMP's sustainability analysis following wildfire?

Mortality assessment method

SPR's Continuous Forest Inventory (CFI) samples approximately 2 to 3 percent of the forested area, probably considered by most to be a lower sample for most forest inventories. Where the SPR CFI system excels is in sampling intensity within the plot, measuring all trees in the plot down to one inch Diameter at Breast Height (DBH). Each tree is numbered and has a distance and bearing recorded to plot center so it can be tracked over time. Last re-measurement of the 83 CFI plots in the burn area was 2008 and 2003. This means an excellent record of tree condition prior to the fire now exists.

The purpose of the mortality assessment was to test a set of guidelines based on professional opinion and available scientific literature for the forester or land manager to utilize to evaluate conifer and hardwood mortality levels immediately following wildfire.

The mortality assessment began with an initial evaluation in fall of 2009. The data listed below was gathered on each tree in the plot and applied to *tables 1*, *2* and *3* (below) to evaluate tree mortality:

1. Percent of crown remaining on each tree to 1 inch DBH
2. Percent of crown sprouting on each tree to 1 inch DBH
3. Basal sprouting? Y/N
4. Percent of root system and cambium quadrants (four possible) destroyed by fire (not recorded, but evaluated)
5. Will the tree live or die? Y/N
6. 10^{th} acre plot regeneration less than 1 inch DBH
7. Plot photos in cardinal directions

The tables below state the characteristics that were assessed on each tree to determine whether the tree would live or die. For example, look at *table 2*, Tanoak:

- zero percent canopy remaining
- less than 5 percent canopy sprouting

- missing more than 66 percent of its root system
- Two quadrants of cambium destroyed by fire
- The tree was considered dead.

Table 1—*Thresholds for determining tree mortality*: *1 – 8 inch DBH classes* (*1.0" – 8.9"*).

Tree Species	Percent canopy remaining	Percent canopy sprouting	Percent of root system missing	Cambium quadrants destroyed by fire
Redwood (RW)	0%	10%	33%	1
California nutmeg (CN)	0%	10%	33%	1
Live oak (LO)	0%	10%	33%	1
Tanoak (TO)	0%	10%	33%	1
Red alder (RA)	0%	10%	33%	1
California bay laurel (CL)	0%	10%	33%	1
Pacific madrone (PM)	0%	10%	33%	1
Big leaf maple (BM)	0%	10%	33%	1
Douglas-fir (DF)	70%	N/A	33%	1
Monterey pine (MP)	70%	N/A	33%	1
Knobcone pine (KP)	70%	N/A	33%	1

Table 2—*Thresholds for determining tree mortality: 9 – 16 inch DBH* (*9.0" – 16.9"*).

Tree Species	Percent canopy remaining	Percent canopy sprouting	Percent of root system missing	Cambium quadrants destroyed by fire
Redwood (RW)	0%	5%	66%	2
California nutmeg (CN)	0%	5%	50%	2
Live oak (LO)	0%	5%	66%	2
Tanoak (TO)	0%	5%	66%	2
Red alder (RA)	0%	5%	50%	2
California bay laurel (CL)	0%	5%	50%	2
Pacific madrone (PM)	0%	5%	50%	2
Big leaf maple (BM)	0%	5%	50%	2
Douglas-fir (DF)	60%	N/A	33%	1
Monterey pine (MP)	60%	N/A	33%	1
Knobcone pine (KP)	60%	N/A	33%	1

Table 3—*Thresholds for determining tree mortality: 17.0+ inch DBH classes* (*17.0"+*).

Tree Species	Percent canopy remaining	Percent canopy sprouting	Percent of root system missing	Cambium quadrants destroyed by fire
Redwood (RW)	0%	0%	66%	3
California nutmeg (CN)	0%	0%	50%	2
Live oak (LO)	0%	0%	66%	3
Tanoak (TO)	0%	0%	66%	3
Red alder (RA)	0%	0%	50%	2
California bay laurel (CL)	0%	0%	50%	2
Pacific madrone (PM)	0%	0%	50%	2
Big leaf maple (BM)	0%	0%	50%	2
Douglas-fir (DF)	60%	N/A	33%	2
Monterey pine (MP)	60%	N/A	33%	2
Knobcone pine (KP)	60%	N/A	33%	2

Each tree was evaluated, qualitatively, around these guidelines, but if the tree met all of the thresholds by DBH category it was considered dead.

Mortality assessment results

The mortality assessment began with an initial evaluation in fall of 2009 These same 83 plots (2,877 trees) were re-evaluated in spring of 2010 and spring of 2011 to determine whether the initial mortality assessment in fall of 2009 was correct. Overall weighted average for percent accuracy to date is 89.3 percent. See *tables 4*, *5*, and *6* below for more detailed information on species, diameter class, percent accuracy and sample size.

Table 4—*Accuracy of tree mortality assessment: 1 – 8 inch DBH.*

Tree Species	Percent accuracy	Sample size
Redwood (RW)	90.4%	470
California nutmeg (CN)	58.3%	12
Live oak (LO)	85.9%	207
Tanoak (TO)	88.0%	302
Red alder (RA)	62.5%	8
California bay laurel (CL)	84.0%	119
Pacific madrone (PM)	77.0%	61
Big leaf maple (BM)	0.0%	2
Douglas-fir (DF)	96.2%	290
Monterey pine (MP)	82.3%	17
Knobcone pine (KP)	0.0%	0
Weighted average	**88.7%**	1488

Table 5—*Accuracy of tree mortality assessment: 9 – 16 DBH.*

Tree Species	Percent accuracy	Sample size
Redwood (RW)	93.2%	223
California nutmeg (CN)	80.0%	5
Live oak (LO)	88.0%	150
Tanoak (TO)	83.9%	181
Red alder (RA)	0.0%	0
California bay laurel (CL)	66.0%	50
Pacific madrone (PM)	73.0%	15
Big leaf maple (BM)	0.0%	0
Douglas-fir (DF)	97.5%	122
Monterey pine (MP)	78.9%	19
Knobcone pine (KP)	100.0%	3
Weighted average	**88.1%**	768

Table 6—*Accuracy of tree mortality assessment: 17.0+ inch DBH.*

Tree Species	Percent accuracy	Sample size
Redwood (RW)	97.7%	271
California nutmeg (CN)	66.0%	3
Live oak (LO)	89.1%	74
Tanoak (TO)	95.4%	66
Red alder (RA)	28.6%	7
California bay laurel (CL)	55.0%	9
Pacific madrone (PM)	50.0%	6
Big leaf maple (BM)	0.0%	0
Douglas-fir (DF)	90.5%	180
Monterey pine (MP)	100.0%	2
Knobcone pine (KP)	100.0%	3
Weighted average	**92.4%**	621

Mortality assessment discussion

The mortality assessment was led in the field by the same SPR forestry technicians for all three evaluation periods maintaining continuity in an assessment that had a manageable level of subjectivity.

Following the initial evaluation of the 83 plots in fall 2009 portions of the burn area were salvaged harvested in 2010 and harvested under the SPR NTMP in 2011. Forestry technicians were very systematic at determining if trees evaluated in 2009 were affected by harvesting activities. If so, those trees were removed from the sample for the purposes of this test.

Accuracy for the mortality assessment tables in the first two evaluation periods (spring 2010 and spring 2011) following the initial evaluation in fall 2009 at 89.3 percent was strong, but the mortality assessment table faltered a little in the thinner bark species; California nutmeg, red alder, California bay laurel, Pacific madrone, and big leaf maple in the mid (9" to 16" DBH) and upper (17.0+" DBH) diameter classes. Based on the data set, reducing the "Cambium quadrants destroyed by fire" to "one" in both the mid and upper diameter classes would increase accuracy. These trees did not withstand as much cambial damage as expected.

The mortality assessment showed very strong results in redwood, tanoak, live oak, Douglas-fir, Monterey pine, and knobcone pine to date. Adjustments to increase accuracy for these species would likely require additional data on other burn characteristics such as scorch height and bark thickness.

Conclusion

A combination of factors affected the decision making process in regards to what trees were harvested to encourage regeneration of managed forestland following wildfire. One or more of the following criteria affecting an individual tree were reason to suspect substantial damage and sustained negative impacts from the fire: cambium damage on multiple sides of the tree, extensive root damage with voids under the tree, extensive crown consumption or scorching that kills limbs, and significant prior defect combined with damage. This is an ongoing learning experience that will be refined as time passes and managed forestland continues to burn and be harvested.

The process of damage evaluation provided significant direction to determine an applied set of quantitative criteria to develop the mortality assessment. It is clear that more years need to pass to understand the full effects that the Lockheed Fire, compounded by the Pine Mountain Fire, had on the forested area in the Scotts Creek watershed. Follow-up surveys will re-visit the 2,877 trees each spring to see if the mortality assessment table continues to hold its accuracy. There is also movement from Cal Poly San Luis Obispo to begin sampling scorch height and bark thickness in addition to the mortality assessment in an attempt to create a local wildfire mortality equation.

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Decomposition and N Cycling Changes in Redwood Forests Caused by Sudden Oak Death

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Key words: decomposition, litterfall, nitrogen mineralization, *Phytophthora ramorum*, redwood, selective species removal, sudden oak death

Introduction

Phytophthora ramorum is an emergent pathogen in redwood forests which causes the disease sudden oak death. Although the disease does not kill coast redwood (*Sequoia sempervirens*), extensive and rapid mortality of tanoak (*Notholithocarpus densiflorus*) has removed this important tree in much of the central and southern distribution of the redwood forest type. *Phytophthora ramorum* was first described in the 1990s and while the native range of the pathogen remains unknown, it has a narrow range of genetic variation characteristic of an exotic species (Rizzo et al. 2005). *P. ramorum* has a broad host range in California but infection leads to very different impacts to host health. Infection in tanoak leads to rapid host mortality which can occur in as little as 2 years in large diameter trees (Cobb 2010a). Sporulation rates are also highly variable among redwood forest hosts. Tanoak and California bay laurel (*Umbellularia californica*) support sporulation sufficient for emergence of disease, but sporulation can be as much as 10 times greater in bay laurel during warm spring rain events (Davidson et al. 2008, 2011). Infection in bay laurel has no known negative impacts on bay laurel health at the individual or population level (Cobb et al. 2010, Davidson et al. 2008). These differences in sporulation are important drivers of pathogen spread across the range of redwood (Meentemeyer et al. 2011). Within stands, increased prevalence of infected bay laurel and tanoak increase tanoak mortality rates but infection in bay laurel has the greatest effect on tanoak mortality commensurate with the high rates of sporulation in this species (Cobb et al. 2010). Redwood stands with high densities of bay laurel are most likely to be invaded by *P. ramorum* (Meentemeyer et al. 2008) while stands with high densities of tanoak are likely to experience the greatest ecosystem impacts (Lamsal et al. 2011, Ramage et al. 2011).

Foresters have recognized the importance of native insect pests and pathogens since the beginning of modern forestry but over the last 100 years, exotic pests and pathogens increasingly challenge the goals of ecosystem management including the maintenance of biodiversity and functional processes such as nutrient cycling (Eviner and Likens 2009, Lovett et al. 2006). Insects and pathogens can alter the chemical composition of plant material by physically damaging foliage or eliciting plant chemical defenses. These changes can result in altered rates of litter decomposition

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and nutrient release (Cobb et al. 2006). Mortality of individual trees arrests nutrient uptake and alters forest microclimate (Orwig et al. 2008) which can in turn cause increased nutrient cycling and decomposition rates (Cobb and Orwig 2008). When outbreak results in the removal of one species, a shift in forest community composition can lead to long-term changes to decomposition and nutrient cycling. Species removal can cause disproportionately large changes to an ecosystem if the removed tree has ecologically unique functions such as casting deep shade, producing especially nutrient rich, or nutrient poor litter (Cobb 2010b, Lovett et al. 2006). While it is clear that tanoak is declining in much of the range of redwood forests, the implications of tanoak removal for redwood ecosystem function are unknown (Rizzo et al. 2005).

Eradication of many exotic forest pests and pathogens is rarely an attainable goal. For *P. ramorum,* the broad host range, extensive geographic area which has been invaded, and the lack of impacts to bay laurel health suggest this pathogen is a naturalized part of many redwood forest communities. Furthermore, *P. ramorum* is expected to establish throughout the entire redwood range within the next 20 to 30 years (Meentemeyer et al. 2011). Understanding how the pathogen has altered redwood ecosystem processes is essential to assessing the costs and benefits of disease management. However, almost no published studies have reported baseline rates of ecosystem processes for redwood forests. How substantial are *P. ramorum* impacts on redwood ecosystem processes? What mechanisms are responsible for ecosystem change in these forests? We conducted a series of field studies quantifying soil N cycling, litterfall, and litter decomposition to begin addressing these questions. The objective of this paper is to report baseline rates of nutrient cycling for redwood forests and summarize the overall affects of sudden oak death on these processes.

Study design

We quantified litterfall dynamics in two second-growth redwood forests in Sonoma and Marin Counties, Jack London State Park and the Marin Municipal Water District respectively. At each site, 30 circular 500 m^2 plots were established in 2002 to study the influence of community composition on pathogen establishment and disease progression (Maloney et al. 2005). We selected a subset of 15 plots at each site which spanned a range of tanoak biomass and pathogen prevalence. We examined plots across a range of disease impacts with the expectation that increasing mortality will lead to greater impacts to ecosystem processes. Litter traps were collected every 6 to 12 weeks from 2007 to 2009. Soil N cycling was quantified *in situ* using open-top, intact soil cores with incubation periods of 10 to 18 weeks. We made an additional measurement of species effects on soil N cycling at the Jack London site by collecting soils directly beneath individual redwood, tanoak, bay laurel, and *P. ramorum* killed tanoak. These soils were collected outside of our permanent study plots from the top 20 cm from eight locations at cardinal directions around each tree. All soils were returned to the laboratory on ice, sieved to pass a 2 mm mesh screen and extracted with 1N KCL within 48 hours of collection. Litter decomposition of redwood, bay laurel, and tanoak litter was measured over 2 years with 1 mm mesh bags.

Results

Litterfall amounts were variable across seasons and among species. Monthly amounts of tanoak and bay laurel litterfall tended to peak in midsummer while redwood litterfall was greatest in late fall and early winter. Tanoak litterfall was the most consistent across seasons but reached the maximum (~100 kg ha^{-1} $month^{-1}$) in May and June. Bay laurel litterfall peaked at ~50 kg ha^{-1} in August. Redwood contributes the greatest mass to litterfall in these forests with maximum amounts of ~800 kg ha^{-1} in December. Cumulative tanoak mortality was associated with lower tanoak litterfall mass.

Soil N cycling rates were within the range of other coniferous forests (*fig. 1*) and N pools and N mineralization were frequently dominated by NO_3-N. Mineralization rates in the field were an order of magnitude higher than rates measured in the laboratory (*fig. 1*) which may be a function of moisture availability. The open-top cores used in field incubations allow moisture to reach microbes and maintain mineralization processes while in the laboratory, we used field moist soils and did not add any additional water.

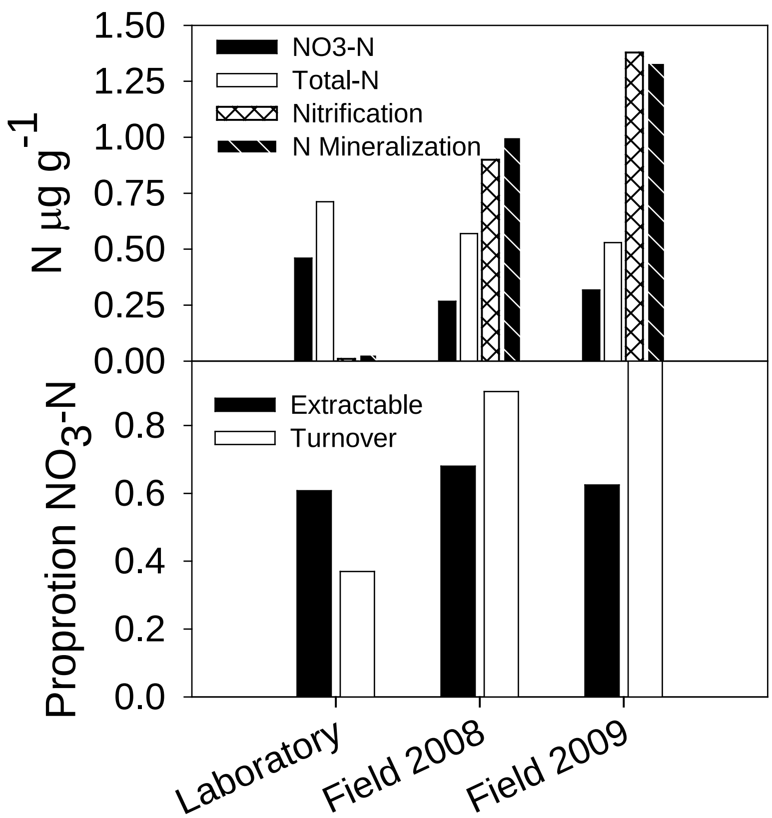

Figure 1—Extractable N and N turnover rates (top) and the proportion of NO_3-N in extractable N and N turnover (Bottom). Data are from a 5-week measurement of field-moist soils conducted in the laboratory at 22 °C (laboratory) and average rates (µg N g^{-1} soil d^{-1}) or amounts (µg N g^{-1} soil) from 2 years of measurements in the field. All data are means.

Litter decomposition was most rapid in bay laurel, slowest in redwood, and intermediate for tanoak. Litter decomposition rates were very low during summer months and increased during the wet season between October and May. Bay laurel consistently had the lowest mass remaining and was about 15 percent lower than tanoak and 30 lower than redwood.

Discussion

Phytophthora ramorum and the subsequent disease sudden oak death have impacted redwood forests throughout the central and southern range of this forest type (Meentemeyer et al. 2008, Ramage et al. 2011, Rizzo et al. 2005). In our study sites, tanoak mortality decreased tanoak litterfall mass and increased the availability of NO_3-N. Changes to NO_3-N availability were modest, but tanoak litterfall is rapidly reduced by disease. Tanoak rapidly develops basal sprouts following mortality of above ground biomass (Cobb et al. 2010, Ramage et al. 2011) which maintains tanoak as part of redwood forest litterfall even after substantial mortality has occurred. Although tanoak will remain as a component of redwood litterfall for several years following disease outbreak, the amounts of litterfall from tanoak will be much lower relative to redwood forests not invaded by *P. ramorum*.

Changes in species composition are likely to have the most long-lasting and the greatest magnitude of effect on ecosystem processes following the emergence of sudden oak death within stands. Although bay laurel accounts for a relatively small proportion of litterfall in redwood ecosystems, it has the highest N concentration of all species in these forests. Redwood has the lowest N concentration and tanoak was intermediate between the two species. These patterns suggest that redwood forest litter decomposition rates will either increase or decrease depending on whether bay laurel, redwood, or other species are able to exploit resources made available by tanoak mortality. Species shifts have been responsible for pronounced and long-lasting ecosystem changes following other major outbreaks (Cobb 2010b, Paillet 2002) which highlights the importance of understanding changes in species composition following outbreak of sudden oak death (e.g., Cobb et al. 2010, Maloney et al. 2006, Ramage et al. 2011). It is unclear how species composition will change over longer time scales in *P. ramorum* invaded redwood forests. However, shifts in species composition will likely have the greatest magnitude and longest-lasting impacts on the ecosystem processes focused on by this study.

Identifying management goals is the most important step to assessing the need for disease management in redwood forests. While redwood is not directly threatened by *P. ramorum* (infection does not cause mortality in redwood), increased fire impacts and fire severity in some *P. ramorum* invaded forests may justify fuel reduction treatments (see Metz et al. 2011). Preemptive management is also likely to be most effective in retaining overstory tanoak and the unique flora and fauna associated with this tree (Cobb 2010a). For disease treatments to be effective in retaining overstory tanoak it will often be necessary to remove bay laurel and reduce the overall density of tanoak such as through the removal of small tanoak stems. Treatments will be most effective when they are designed or associated with management to increase the growth of redwood which may in turn lead to suppression of regeneration from sporulation supporting species like bay laurel and tanoak. Preventative treatments would be similar to many current silvicultural prescriptions to reduce tanoak competition with redwood but would require modifications that intentionally retain and protect individual tanoak with high-biodiversity value. By shifting species composition to a greater dominance of redwood, ecosystem processes will also reflect characteristics typical of redwood ecosystems; specifically, slower litter decomposition rates, increased above ground biomass, and possibly increased soil C storage. However, the metric of success for these treatments should be based on their effectiveness in retaining unique biodiversity associated with tanoak. Lack of action is likely to increase the importance of bay laurel in many stands and this will shift ecosystem processes towards overall rates of N cycling, litter decomposition, and above ground biomass dynamics that reflect the autecology of bay laurel. For the individual manager facing the emergence of sudden oak death, the costs of these disease impacts must be weighted in the light of their overall goals to determine appropriate management actions.

Acknowledgments

We thank H. Mehl, S. Lynch, A. Westbrook, F. Fogel and C. Shoemaker for their field and laboratory support of this research. We thank M. Metz, E. Fichtner, and V. Eviner for helpful comments on the design of the study and analysis of the data. Additionally, we are grateful to the California State Parks and the Marin Municipal Water District for facilitating this research on their lands. This research was funded by the NSF-NIH Ecology of Infectious Diseases program EF-0622770, the Gordon and Betty Moore Foundation, the USDA Forest Service Pacific Southwest Research Station, and an Ernest E. Hill Climate Change Fellowship to RCC.

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Post-fire Response of Coast Redwood One Year After the Mendocino Lightning Complex Fires

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Abstract

Coast redwood (*Sequoia sempervirens*) forests have undergone significant changes over the past century and are now in state more conducive for wildfires. Because fires have been uncommon in redwood forests over the past 80 years, managers have limited data to make decisions about the post-fire environment. In June 2008, a series of lightning storms moved through northern California igniting numerous fires throughout the redwood region of Mendocino County. Here, we collected fire-injury data on 1024 redwood trees on commercial timberlands three months after the fire and quantified mortality and biological responses one year later. Although over half the sampled trees had at least 90 percent of their crowns scorched, only 18.2 percent were completely top-killed after one year; and 87.7 percent of this mortality was confined to trees less than 20.3 cm in diameter. Over 80 percent of the trees regenerated leaves and shoots from axillary buds, and a similar percentage resprouted basally. Logistic regression modeling indicated that diameter at breast height (DBH), cambium kill, and percent crown scorch were significant predictors of tree mortality. These results indicate that although small redwoods are predicted to have the highest mortality, their ability to resprout may obviate the need for replanting in areas where redwood is dominant. The management implications for the larger surviving trees are less clear and require further long-term study to examine delayed mortality, growth rates, and wood quality.

Key words: basal sprouting, fire injury, mortality, redwood, Sequoia, wildfire

Introduction

The fire ecology of coast redwood (*Sequoia sempervirens*) continues to be an area of active inquiry. Over the past century, timber harvesting, fire suppression, and human encroachment have substantially altered the environmental context of coast redwood (Sawyer et al. 2000, Scanlon 2007). A major concern in many western forests is that increased woody debris, along with younger and denser forests, and a drier regional climate will lead to higher fire frequency and more catastrophic wildfires (Stephens et al. 2009). Conditions within many redwood forests throughout the region have changed substantially in recent decades and are now reaching states more conducive for wildfires.

Historically, large wildfires have been uncommon in coast redwood forests because of reduced ignitions and suppression policies of the past 80 years (Scanlon 2007). The lack of fire in redwood forests has resulted in a significant information gap regarding the effects fire on this species. Currently, managers have limited data to make decisions about harvest in the post-fire environment (Finney and Martin 1993, Ramage et al. 2010). In fact, there are no quantitative data on fire injury

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characteristics and the post-fire response of coast redwood on second- and third-growth growth commercial forestlands.

On 20 and 21 June 2008, a series of lightning storms moved through Mendocino County during a second-year of drought igniting 129 fires and burning 22,193 ha (54,817 ac) over a four-week period in what became known as the Mendocino Lightning Complex (MLC). Approximately, 42.3 percent of the burned area encompassed a single forest landowner. This unprecedented event provided an opportunity to examine the relationship between fire-injury characteristics and the response of redwood on forestlands over a 5-year period.

We present preliminary results on the response of coast redwood 1 year after the MLC fires. Specifically, our objectives were to: 1) quantify the biological response; 2) model fire-injury characteristics and tree mortality; and 3) use these data to develop future management guidelines.

Methods

Study area

Sampling sites were located in the North Coast physiographic province of Mendocino County, California and consisted of second- and third-growth coast redwood forests owned and managed by Mendocino Redwood Company for commercial timber production. These forestlands are primarily dominated by coast redwood, Douglas-fir (*Pseudotsuga menziesii*), and tanoak (*Notholithocarpus densiflorus*), and a small percentage of other hardwood and conifer species.

Sampling

During September 2008 we established permanent plots across four separate fire areas of the MLC. Within each fire we selected plot locations from fire severity maps based on landscape-level crown scorch indices obtained from satellite imagery. Plots (17.8 m (0.1 ha) radius) were installed until ~250 redwood trees ≥ 9.9 cm (3.9 in) diameter at breast height (DBH) were measured for each fire. Adjustments to the plot centers were made to maximize the number of redwood trees in each plot and cover a range of tree sizes and crown scorch percentages. In each plot trees were painted and tagged to facilitate monitoring in subsequent years.

Tree and fire-injury measurements

We measured tree DBH with a diameter tape (at 1.37 m above ground); total tree height and pre-fire height-to-crown base (HTCB) with a hypsometer. Fire-injury metrics such a percent volume crown scorched (PVCS) were visually estimated using a double-observer method until calibration was within 5 to 10 percent of each other. Crown scorch height (CSHT), maximum char height (CHHTMX), and minimum char height (CHTHMN) were measured with a hypsometer. Bark char was assessed by dividing tree circumference into four quadrants based on cardinal directions and visually classifying char severity within the first 30 cm above ground. Each quadrant was given a score (0=none; 1=low; 2=moderate; 3=severe) and all quadrant scores were summed to provide an overall score ranging from 0 to 12. Cambium kill was assessed by dividing the base of the tree into quadrants (similar to the bark char assessment). In the center of each quadrant, a cordless drill (with a 1.9 cm wide bit) was used to access the cambium below the bark to score (0=live; 1=dead). A cambium kill rating was calculated by summing all of the quadrant scores for an overall score ranging from 0 to 4.

Biological response measurements

During the fall of 2009 all fire plots were revisited to evaluate tree status (live or dead) and measure specific biological responses. Percentages of live and dead crown were measured visually, and the percentage of bole sprouting was measured as a proportion of total tree height. Bole sprouting was considered a distinct form of regeneration so it was separated from the live crown measurement, which was further divided into residual canopy (unburned) and regenerating canopy (limb sprouting). Basal sprouts were assigned to parent trees based on proximity, and the total number of sprouts was tallied and the tallest sprout in each group was measured with a height pole.

Statistical analysis

We calculated means and standard errors to illustrate measures of central tendency and variation, respectively. Most of the tree and fire-injury data deviated from the assumptions of parametric statistics, so we used the non-parametric Wilcoxon rank-sum test to compare means between groups.

We used logistic regression to develop a statistical model of tree mortality based on several combinations of tree and fire-injury variables (bole injury and crown scorch). Variables were evaluated for collinearity prior to inclusion in logistic regression models. Model fit was assessed using the χ^2 goodness-of-fit test and the discriminatory ability of each model was evaluated using the Receiver Operating Characteristic (ROC). All statistics were analyzed with the program Stata version 11.1 and statistical significance was set at α=0.05.

Results

A total of 1024 trees were measured across a total of 29 plots (6 to 9 plots sampled per fire). Tree size distribution followed a reverse J-shape distribution, with over 72.8 percent of the trees occurring in the two smallest size classes, and the remaining trees distributed among the merchantable size classes (> 40.6 cm DBH) in descending order of increasing size. One year after the MLC fires, 187 redwood trees (18.2 percent of all trees measured) were observed to be dead (defined as a completely top-killed primary stem). All top-killed trees were in the two smallest size classes, 164 (87.7 percent) of which were in the smallest size class (9.9 to 20.3 cm DBH).

Comparisons of tree characteristics showed that live trees were, on average, nearly twice as large (in diameter and total height) as dead trees. Dead trees, on average, exhibited significantly higher levels of crown scorch, bole char proportion, bark charring, and cambium kill compared to live trees (*table 1*).

Crown regeneration for 837 live trees exhibited a pattern of increasing mean percentage of live crown and a decreasing mean percentage of bole sprouting in progressively larger size classes of trees (data not shown). Only 318 (37.9 percent) live trees exhibited sprouting on greater than 5 percent of the bole, usually confined

Table 1—*Mean tree and fire injury characteristics for 1024 redwood trees sampled with standard errors and ranges shown.*

Variables[a]	Total Mean (n=1024)	Live Mean (n=837)	Dead Mean (n=187)
DBH (cm)	31.9 (0.68)	35.6 (0.78)	15.3 (0.35)[b]
	9.9-151.9	9.9-151.9	9.9-33.8
TLHT (m)	19.5 (0.33)	21.4 (0.37)	11.1 (0.29)[b]
	3.7-55.5	4.6-55.5	3.7-26.8
HTCB (m)	8.7 (0.22)	9.5 (0.23)	5.2 (0.19)[b]
	1.2-36.0	1.5-36.0	1.2-18.6
CSHT (m)	14.1 (0.27)	14.9 (0.32)	10.8 (0.28)[b]
	0-48.2	0-48.2	3.7-26.8
PRCS (%)	69.7 (1.15)	63.7 (1.31)	96.3 (0.85)[b]
	0-100	0-100	10-100
CHHTMX (m)	5.5 (0.14)	5.7 (0.16)	4.8 (0.20)
	0-28.0	0-28.0	0-19.8
CHHTMN (m)	2.8 (0.10)	2.7 (0.11)	3.1 (0.20)[b]
	0-27.1	0-27.1	0-19.8
PRCHMX (%)	30.4 (0.64)	27.1 (0.61)	45.6 (1.83)[b]
	0-100	0-100	0-100
BCR	8.5 (0.06)	8.3 (0.06)	9.4 (0.15)[b]
	0-12	0-12	0-12
CKR	0.9 (0.04)	0.6 (0.04)	2.2 (0.11)[b]
	0-4	0-4	0-4

[a] DBH = diameter at breast height; TLHT = tree height; HTCB = Pre-fire height-to-crown base; CSHT = crown scorch height; PRCS = crown scorch; CHHTMX = maximum bole char height; CHHTMN = minimum bole char height; PRCHMX = bole char proportion; BCR = bark char rating; CKR = cambium kill rating

[b] Significantly different from live trees ($P<0.05$)

to trees with a greater percentage of fire-injury, particularly those with dead tops and higher percentages of dead crowns. Mean percentage of limb sprouting increased with larger size classes indicating lower levels of canopy injury with increasing tree height.

Basal sprouting was associated with a total of 822 out of 1024 trees (80.2 percent). Sprouts were associated with 654 of 837 live trees (78.1 percent) and 168 of 187 dead trees (89.8 percent). Basal sprouting patterns showed significant differences in both sprout number and maximum sprout height between live and dead trees (*table 2*).

Logistic regression modeling yielded five significant models consisting of two- and three-variable combinations, all of which had excellent model fit and discriminatory ability in predicting live and dead trees based on fire-injury (*table 3*; ROC values ≥0.8). The best model contained DBH, PRCS, and CKR as significant predictors of mortality. Alternative models with different measures of bole damage (BCR and PRCHMX) were also significant, but ranked lower than the three-variable model with CKR. DBH was the single-most important variable in determining probability of mortality, followed by measures of bole damage, and PRCS.

Table 2—*Mean maximum height and mean total number of basal sprouts by size class, live and dead trees. Standard errors () and ranges shown.*

	Live			Dead		
DBH Class (cm)	Mean Max Height (m)	Mean No. of Sprouts	N	Mean Max Height (m)	Mean No. of Sprouts	N
9.9-20.3	0.96 (0.04)	5.3 (0.30)	228	1.31 (0.05)[b]	7.7 (0.47)[b]	151
	0.10-3.00	1-28		0.15-3.00	1-40	
20.3-40.6	0.97 (0.04)	5.8 (0.34)	234	1.39 (0.19)	6.1 (1.26)	17
	0.05-3.30	1-30		0.30-3.00	1-20	
40.6-60.9	0.94(0.05)	7.1 (0.63)	128	NA	NA	NA
	0.15-2.50	1-50				
60.9-81.2	0.99 (0.10)	5.2 (0.60)	43	NA	NA	NA
	0.15-2.50	1-16				
> 81.2	0.99 (0.13)	7.2 (1.00)	21	NA	NA	NA
	0.15-2.10	1-17				
All Size Classes	0.96 (0.02)	5.9 (0.21)	654	1.32 (0.05) [b]	7.6 (0.44) [b]	168
	0.05-3.30	1-50		0.15-3.00	1-40	

[b] Significantly different from live trees (P<0.05)

Table 3—*Top five mortality models for coast redwood one year post-fire. Model 1 is the best overall model.*

Model Rank	Intercept	DBH	PRCS	CKR	BCR	PRCHMX	ROC
1	-1.5039	-0.1698	0.0305	0.5425	-	-	0.912
2	-4.4398	-0.1989	0.0427	-	0.3541	-	0.909
3	-1.9276	-0.1743	0.0349	-	-	0.0253	0.904
4	1.2200	-0.1786	-	0.6690	-	-	0.902
5	-1.8892	-0.1797	0.0454	-	-	-	0.893

Probability of mortality for the most significantly burned trees (PRCS ≥80, CKR =4) was high (>0.7) for trees 13.0 cm (5.11 in) DBH or less *(fig. 1*). As DBH increases, probability of mortality declines, nearly reaching zero (survival probability of 1) for trees with a DBH of 46.0 cm (18 in). We did not observe any mortality in trees greater than 33.8 cm (13.3 in) DBH.

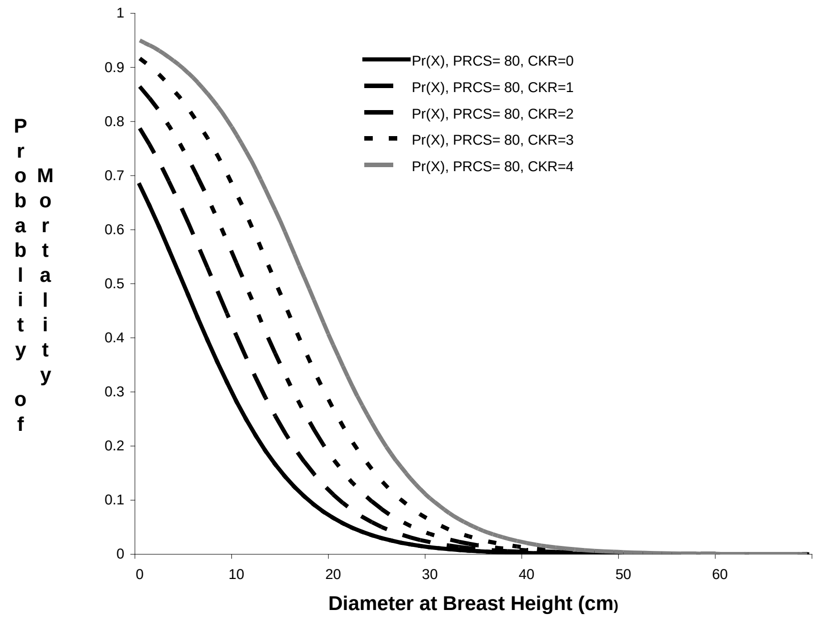

Figure 1—Probability of mortality based on DBH for trees with 80 percent crown scorch (PRCS) at all levels of cambium kill (CKR).

Discussion

The ability of coast redwood to resist fire and other disturbances has been documented in several observational accounts (Abbot 1987, Fritz 1932). Our study is the first to provide quantitative measures of fire-injury and biological response of second- and third-growth redwoods, and possibly, for one of the largest wildfire complexes (by area burned) in the redwood region in over 70 years (Scanlon 2007). Our initial concern regarding the MLC fires was that a landscape comprised of young coast redwood combined with moderate to severe fires might result in significant mortality of merchantable size classes (> 40.6 cm), with even greater concerns about decreases in growth and development of defects.

Mortality of merchantable sized trees was not observed in any of the plots one year following the MLC fires. Mortality was confined to the two smallest size classes, which is not surprising given that fire typically moves from the ground up and tends to impact smaller trees. Finney and Martin (1993) also observed mortality in the same size classes but for prescribed fires of differing intensities. Interestingly, they did not observe any mortality in trees larger than 32.4 cm (12.7 in) DBH, which was close to our maximum mortality size of 33.8 cm (13.3 in) DBH. However, a substantial portion of their sampling distribution included trees < 9.9 cm DBH whereas ours did not (Finney, pers. comm.). This sampling difference is an important distinction because mortality of trees < 9.9 cm DBH was disproportionate compared to larger DBH trees in their study. Although we did not directly measure trees this small, > 90 percent of these trees were observed dead in all plot-level tallies across our study sites.

Coast redwood exhibited both resistance and system resilience to fire across all plots. Despite having sustained significant crown and bole injuries, nearly 75 percent of the trees ≤ 40.6 cm (16 in) DBH were still alive and were regenerating new crowns from epicormic sprouts. The incidence and extent of bole sprouting was greater in small trees because the original crown and dormant buds along their limbs were severely damaged or killed. Higher levels of limb sprouting were also seen in larger trees whose buds were located sufficiently above convective heat currents to avoid lethal temperatures. Basal sprouting occurred around all size classes of trees and was most vigorous around dead trees. Thus, already established redwood stands that experience significant fire, including those comprised of juvenile-sized trees, are likely to maintain site dominance into the future without the need for replanting. Furthermore, coppicing severely damaged young trees may further enhance sprout development and regeneration time.

Our logistic regression modeling supports previous observations that tree size (DBH) and different measures of fire injury are important factors in predicting mortality of coast redwood (Finney and Martin 1993, Ramage et al. 2010). Based on our best model, trees exceeding 46 cm DBH are predicted to have nearly 100 percent survival probability regardless of fire injury. However, as tree DBH becomes progressively smaller, fire injury becomes a more important determinant of survival probability. This concurs with plot-level bole survival data recently published by Ramage et al. (2010) for second- and old-growth redwood stands. Mortality models for prescribed fire also illustrate a similar effect of DBH on survival probability (Finney and Martin 1993).

The utility of the cambium kill metric in mortality models following wildfire has been reported by others but not for coast redwood (Hood et al. 2007). Measures of cambium kill significantly improved mortality predictions for trees ≤ 20.3 cm DBH with high levels of crown scorch. However, assessing cambium is labor intensive so alternative measures of bole injury (BCSUM and PRCHMX) may be preferable for field application. Previous research has shown that cambial damage is associated with fuel loading and fire behavior immediately surrounding individual trees (Ryan and Frandsen 1991). Although we did not have any pre-fire fuels data for our plots, we hypothesize that cambial damage was not only influenced by accumulated litter and woody debris, but also by the clumped distribution of regenerating trees surrounding remnant parent stumps that continued to burn and emit radiant heat long after the fire passed through. This latter hypothesis is supported by our observation, as well as others in the redwood region (Finney and Martin 1993), that many of the stump-facing sides of coppice-generated trees proximal to burned stumps had severely burned bark and dead cambium.

Although PVCS was significant in the model, its ability to improve prediction was not as significant as CKR and other measures of bole damage. The ability of coast redwood to readily resprout from aerial buds may explain the lesser importance of this variable when compared to mortality models developed for other non-sprouting tree species. Nonetheless, PVCS along with measures of bole injury are important metrics of fire severity as they may capture physical and behavioral aspects of the fire that are spatially distinct and driven by multiple factors such as fuel loading, topography, adjacent vegetation, and weather conditions.

The high regenerative capacity of coast redwood to wildfire is problematic for modeling mortality. First, categorizing responses into discrete groups (dead or alive) potentially results in significant loss of information that might be better represented along a continuum. For example, 40.7 percent of live trees (341 of 837) had dead crowns of 50 percent or greater and varying amounts of cambium mortality, yet were grouped with live trees having significantly less or no damage. Such trees, if retained to grow for several decades, may exhibit reduced growth rates, have poor wood quality, and many defects. Uncertainty regarding the long-term effects of wildfire on redwood highlights a need to better define what characteristics constitute damage because using the likelihood of mortality as a standard for harvest may be inappropriate for this species.

Conclusions

Redwood exhibited considerable resistance and resilience to wildfire even on a landscape that has been significantly altered by over a century of commercial timber harvesting and several decades of fire exclusion. Overall, the fires of 2008 were an admonition that conditions conducive for wildfire exist on many landscapes throughout the redwood region. While many of the fires on our forestlands caused significant damage to young stands of redwood and other more fire-sensitive conifer species, at least 50 percent of the fires had minimal impact on the forests as they behaved like a prescribed fire. Given the state of our forests, this observation underscores the importance of reintroducing fire as a tool to reduce wildfire risk in many areas on large forested landscapes within the redwood region. Although our understanding of the short-term responses of coast redwood to wildfire fire is greatly enhanced by this research, further study is necessary to test many of our assumptions how wildfire affects coast redwood over a longer period of time. Annual monitoring of our permanent plots in these fire areas will continue into the future and is necessary to answer questions regarding delayed mortality, reduced growth rates, increased pathogen susceptibility, wood quality, and generation time for wildlife habitat.

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The Effects of Sudden Oak Death and Wildfire on Forest Composition and Dynamics in the Big Sur Ecoregion of Coastal California

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Introduction

Sudden oak death (SOD), caused by *Phytophthora ramorum*, is an emerging forest disease associated with extensive tree mortality in coastal California forests (Rizzo et al. 2005). *P. ramorum* is a generalist pathogen that infects many hosts, but hosts differ in their ability to transmit the disease and in the impacts caused by the disease. In coast redwood forests, tanoak (*Notholithocarpus densiflorus*) is the main host dying from SOD and the main source of pathogen inoculum. SOD leads to compositional changes in these habitats through selective mortality of tanoak.

The effect of SOD on forest dynamics does not happen in isolation from other disturbances. Rather, fire is an important part of the natural disturbance regime in California's coastal forests. Elevated mortality by the disease may lead to increased fuel loads in these forests, leading to the potential for SOD to interact with wildfire severity (Valachovic et al. 2011). In 2008, wildfires in SOD-impacted forests presented a natural experiment to examine the interaction between these two disturbances (Metz et al. 2011).

We used a long-term network of forest monitoring plots in the Big Sur region of central California to examine the separate and joint impacts of an emergent forest disease and wildfire on forest dynamics and diversity. The network contains burned and unburned areas, and areas with and without *P. ramorum*. Specifically, we asked: 1) Did redwood stands that became infested by *P. ramorum* differ in species composition from uninfested areas? 2) How is *P. ramorum* changing species composition in redwood forests? 3) How did mortality from SOD impact fire severity in burned areas? 4) Was fire mortality selective by species?

Methods

The Big Sur region represents the southern limit of the ranges of both SOD and the coast redwood. Big Sur forests were a relatively early site of invasion by *P. ramorum* and are among the most impacted by the pathogen (Meentemeyer et al. 2008). In Big Sur, the historical and current distribution patterns of redwood consists of scattered stands, primarily in wetter canyon areas near the coasts. The redwood habitat is usually found in close proximity to a variety of other habitat types, including oak woodlands, chaparral, or grasslands, leading to a very heterogeneous matrix of habitat types. Forest and disease dynamics in this landscape can therefore differ dramatically from what is observed further north in the ranges of redwoods and

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P. ramorum, where the climate is cooler and wetter and redwood stands exist in much larger swaths.

We established a network of 280 monitoring plots throughout the Big Sur ecoregion in coastal California in 2006 and 2007 to examine the impacts of SOD on forest dynamics. These plots were located in two habitat types, mixed-evergreen forests (n=164) and redwood-tanoak forests (n=116). Within each plot, we marked, measured, and assessed for pathogen symptoms and health every live or dead stem ≥1 cm DBH. We also quantified the volume of large coarse woody debris, and the vegetation cover of all herbs, shrubs and trees. Here, we present results based on the 116 redwood-tanoak plots. We used live basal area to represent current plot composition or species abundance during a given census. Total basal area, calculated by summing the basal area of live and dead standing stems, was used to re-construct pre-epidemic stand conditions, without considering disease-related or other mortality.

Two wildfires in 2008 (the Basin Complex and the Chalk fire) burned over a third of this network, providing the opportunity to compare forest dynamics in uninfested and infested areas, both within and outside burned areas, using pre-fire data on fuel availability and disease impacts and post-fire data on burn severity and tree mortality. Standing dead basal area and the volume of downed logs were used as measures of large woody fuel abundance prior to the fire. We conducted a survey of burn severity in 2008 (Metz et al. 2011), using the Composite Burn Index, and surveyed for tree mortality in 2009.

Results

Of the 116 redwood plots, 70 were infested with *P. ramorum* in 2006 and 2007. There was a gradient among plots in dominance of tanoak or redwood, and some plots also contained another sporulating host, bay laurel. The species composition of infested plots differed significantly from that of uninfested stands. Stands that became infested had significantly greater total basal area (live and dead stems) of the sporulating hosts tanoak and bay laurel, and significantly lower abundance of redwood. At the same time, tanoak experienced far greater mortality in infested plots than in uninfested plots, and there was a trend towards lower mortality in bay and redwood in infested plots.

Infested plots had significantly greater abundance of standing dead snags and large surface fuels. However, burn severity did not vary according to pathogen presence alone, but rather with the abundance and type of fuels that occur at different stages of the disease progression (Metz et al. 2011). In areas with new SOD-caused mortality, where dead trees still retain a crown of dry, dead material, burn severity increased with increasing SOD impacts. Similarly, in areas where the disease had progressed further and surface fuels had accumulated, soil and substrate burn severity increased significantly with greater volume of host logs. However, the abundance of dead trees and logs were not significant predictors of burn severity across the entire network, when infested and uninfested areas were examined together.

For trees that had been alive upon plot establishment in 2006 and 2007, the amount of basal area that was dead in 2009 was significantly higher in burned areas than unburned areas, regardless of the presence/absence of SOD. Outside of the burned areas, mortality was negligible except for canker hosts in infested areas, where mortality is increased by SOD. Within the burned area, fire-caused mortality varied greatly among canker host species and non-host or foliar host species (which do not have lethal infections). Canker host mortality, primarily driven by tanoak mortality, was as high as 34 percent in burned, uninfested areas and 57 percent in burned, infested areas, on average, with great variation (difference between groups was nearly significant; $t=-1.9669$, $df=36.98$, $p=0.057$). Mortality of non-canker hosts (primarily redwoods, but also bay laurel) was much lower than for canker hosts. However, even non-canker host mortality was elevated in infested stands (16 percent) relative to uninfested stands (6 percent), indicating the potential for SOD mortality to worsen the effects of wildfire, although the difference was not significant ($t=-1.6008$, $df=27.67$, $p=0.12$).

Discussion

Both sudden oak death and the 2008 wildfires have been altering the species composition in Big Sur redwood forests through selective mortality (Maloney et al. 2005, Metz et al. 2011). Tanoak is dying in large numbers in areas infested with *P. ramorum*, and the 2008 wildfires caused greater mortality in canker hosts such as tanoaks relative to other dominant species in these stands (primarily redwoods). These changes in composition may lead to long-lasting impacts to forest diversity, dynamics, and structure in areas impacted by SOD in these fire-prone systems.

When trying to understand the impact of SOD on California’s coastal forests, it is very important to understand that *P. ramorum* is an important biotic disturbance that has been introduced into a system that already had a significant disturbance regime consisting primarily of periodic wildfires. Thus, the full impacts of either disturbance cannot be examined in isolation. The relationship between disease impacts and burn severity is not straightforward. Our surveys of tree mortality 1 year following the fire suggest that mortality of all species is higher in infested areas.

We hypothesize that long-term forest composition will depend on the joint and interacting effects of disease and fire. Changes to tanoak abundance, whether caused by SOD or fire, will have important implications for establishment of *P. ramorum* and disease dynamics, especially in stands where tanoak is the sole sporulating host; stands with increasing redwood abundance are less likely to be infested. Within burned areas, SOD impacts may be elevating redwood mortality, perhaps through longer fire residence times or continued smoldering of surface fuels at the base of large trees that may otherwise be somewhat resistant to a fire that passes through quickly. This leads to the potential for SOD and fire to interact to create longer-lasting impacts to the composition of these forests. Further investigation into these interactions is ongoing.

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Regeneration and Tanoak Mortality in Coast Redwood Stands Affected by Sudden Oak Death[1]

Benjamin S. Ramage[2], Kevin L. O'Hara[2], and Alison B. Forrestel[2]

Abstract

Sudden oak death, an emerging disease caused by the exotic pathogen *Phytophthora ramorum*, is impacting coast redwood (*Sequoia sempervirens*) forests throughout coastal California. The most severely affected species, tanoak (*Notholithocarpus densiflorus*), is currently widespread and abundant in the redwood ecosystem, but diseased areas have begun to experience considerable mortality. Tanoak, which is extremely valuable as food source to numerous wildlife species, is unlikely to successfully regenerate in these areas, and thus affected redwood forests are transitioning to a novel state. In this study, to predict which species might replace tanoak, we investigated regeneration patterns in heavily impacted stands in Marin County, California. Our main findings were: (1) despite reductions in canopy cover, there is no evidence that any species other than tanoak has exhibited a regenerative response to tanoak mortality, (2) the regeneration stratum was dominated by redwood and tanoak (other tree species were patchy and/or scarce), and (3) some severely affected areas lacked sufficient regeneration to fully re-occupy available growing space. Our results indicate that redwood is likely to initially re-occupy the majority of the ground relinquished by tanoak, but also provide evidence that longer-term trajectories have yet to be determined and may be highly responsive to management interventions.

Key words: *Lithocarpus densiflorus*, *Notholithocarpus densiflorus*, *Phytophthora ramorum*, regeneration, redwood, *Sequoia sempervirens*, sudden oak death, tanoak

Introduction

Tanoak (*Notholithocarpus densiflorus* syn. *Lithocarpus densiflorus*), a broadleaf evergreen in the Fagaceae family, is currently widespread and abundant in coast redwood (*Sequoia sempervirens*) forests and is believed to be an integral component of the structure and function of these unique ecosystems (Burns and Honkala 1990, Hunter et al. 1999, Noss 2000). However, the close association between redwood and tanoak may be relegated to history if sudden oak death (SOD), an emerging disease caused by the exotic pathogen *Phytophthora ramorum* continues to spread throughout coastal California. Current research demonstrates drastic declines in tanoak populations and mounting evidence, from numerous field studies (for example, Maloney et al. 2005, McPherson et al. 2010, Ramage et al. 2010) and

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[1] A longer version of this paper has been published. If you wish to cite this paper, please reference the original publication (Ramage, B.S.; O□Hara, K.L.; Forrestel, A.B. 2011. **Forest transformation resulting from an exotic pathogen: regeneration and tanoak mortality in coast redwood stands affected by sudden oak death**. Canadian Journal of Forest Research 41: 763-772.).

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several disease progression models (for example, Meentemeyer et al. 2004), suggests that SOD could eventually drive tanoak to extinction in redwood forests.

Several tree species succumb to SOD, but tanoak is the most severely affected tree and the most abundant SOD-susceptible species in redwood forests (Burns and Honkala 1990, Rizzo et al. 2005). The extreme susceptibility of tanoak results from a combination of factors: (a) little or no genetic resistance, (b) susceptibility of all ages and size classes, and (c) ability of tanoak foliage and twigs to support pathogen sporulation, facilitating eventual lethal infection of the bole (all other species that are killed by SOD appear to require the presence of a secondary foliar host for bole infection; Maloney et al. 2005, Ramage et al. 2010, Rizzo et al. 2005). Furthermore, because many native species support sub-lethal foliar infections, including redwood (Davidson et al. 2008), *P. ramorum* has almost certainly become a permanent resident of infested areas, and thus it is unlikely that tanoak will ever return to pre-SOD abundances in diseased redwood forests. Tanoak stumps often sprout prolifically following death of the main bole (Cobb et al. 2010, Ramage et al. 2010), and it is feasible that root systems could be maintained indefinitely if adequate amounts of photosynthate are consistently produced prior to episodic SOD-induced sprout dieback; a similar situation has been occurring for approximately a century with the American chestnut, *Castanea dentata*, and the exotic disease chestnut blight, caused by *Cryphonectria parasitica* (Ellison et al. 2005). However, even if this scenario was to manifest, such an outcome would still represent a form of *functional extinction* (sensu Ellison et al. 2005).

It is often assumed that redwood forests are relatively poor in tree species diversity because of the strong competitive effects of redwood, but there is surprisingly little evidence to support this conjecture. Given that tanoak is a nearly ubiquitous associate of redwood (Burns and Honkala 1990, Noss 2000), we cannot discount the possibility that tanoak is competitively excluding one or more species otherwise capable of persisting in redwood forests. Of particular concern is the question of whether functionally similar native tree species will be able to colonize and persist in areas previously occupied by tanoak. Tanoak regularly produces large nutritious acorns that are utilized by many wildlife species (for example, bear, deer, and several rodent and bird species), while redwood produces unpredictable crops of small and light seeds with limited wildlife value (Burns and Honkala 1990). If tanoak is not replaced by one or more functionally similar tree species (for example, a true oak species), its loss could result in serious cascading impacts. For instance, acorns are a primary food source for the dusky footed woodrat (*Neotoma fuscipes*), which is in turn a primary food source for the northern spotted owl (*Strix occidentalis caurina*; Courtney et al. 2004).

Regeneration in SOD-induced mortality gaps is likely to differ from regeneration in areas not experiencing mortality, and – because SOD gaps represent a novel occurrence – unexpected patterns may emerge. Features such as standing dead trees and debris piles may attract birds and thereby increase the input of bird-dispersed seeds into disturbed areas (Rost et al. 2009). Seedling and sprout survival and growth rates may increase following the death of mature trees via: (a) improved photosynthetic capacity, which can occur with even small reductions in canopy cover, and/or (b) a reduction in the intensity of competition for water and soil nutrients (Oliver and Larson 1996, Smith et al. 1997); increased basal sprouting incidence could also occur due to bole damage from falling trees. These mechanisms vary among species (Grubb et al. 1977) and thus relative abundances of regeneration in mortality gaps are likely to differ from the surrounding matrix. In the case of redwood, studies have shown that basal sprout survival and growth rates were greater in higher light environments (O'Hara and Berrill 2010, O'Hara et al. 2007), and that a greater proportion of established redwood stems had basal sprouts in an area experiencing SOD-induced tanoak mortality than in an unaffected area (Waring and O'Hara 2008).

Our specific objectives were to: (1) test the hypothesis that regeneration is positively associated with tanoak mortality, (2) identify the species that are regenerating in the greatest numbers in areas severely affected by sudden oak death, and (3) consider the potential implications of the observed patterns.

Methods

Fieldwork was conducted at the Marin Municipal Water District (MMWD), which occupies approximately 8500 ha of protected land on the northern slope of Mount Tamalpais in Marin County, California. Redwood forest, primarily second-growth stands that originated at the end of the 19th century or beginning of the 20th century, is scattered throughout the watershed, covering a wide range of slopes, slope positions, and aspects. Unusual levels of tanoak mortality were first observed in Marin County in 1994, and SOD has been causing extensive tanoak mortality at MMWD since at least 2000 (McPherson et al. 2010, Rizzo et al. 2002). Most study plots (1/20 ha; 12.62 m radius) were randomly located (in redwood forest), but we also used a stratification protocol to ensure adequate representation of: (a) areas with high tanoak abundance but little or no tanoak mortality, and (b) areas with very high levels of tanoak mortality. Additional details are provided in the full version of this paper (Ramage et al. 2011).

Data for mature trees ($\geq$ 10 cm diameter at breast height; 1.37 m height; DBH) were collected during the summer of 2008. Within each plot, we recorded species and DBH for all tree species, as well as health and deterioration status for all tanoaks; details are provided in the full version of this paper (Ramage et al. 2011). In 2008, we also collected one sample per plot of symptomatic tanoak (5 to 10 leaves and 2 to 3 twigs) and/or California bay (*Umbellularia californica*; 5 to 10 leaves), to test for the presence of *P. ramorum* via polymerase chain reaction (PCR); methods are described in Hayden et al. (2006) and results are presented and discussed in the full version of this paper (Ramage et al. 2011). Regeneration and canopy cover data were collected during the summer of 2010. Counts of all seedlings (< 1.37 m height), basal sprouts (< 1.37 m height), saplings (seed or sprout origin stems $\geq$ 1.37 m height and < 3 cm DBH), and juvenile trees (3to 10 cm DBH) were conducted for all tree species in two randomly selected quadrants per plot (for example, NW and SE). Canopy cover was measured with a spherical densiometer at five points per plot (plot center and 3 m in each cardinal direction) and values were averaged.

To limit any potentially confounding effects of tanoak abundance, all plots with less than the median basal area (BA) of total tanoak (living and dead trees combined, calculated with the randomly located plots only; 14.4 m^2 per ha) were excluded from all analyses. Using this dataset (n = 16 plots), we tested the effects of tanoak mortality on canopy cover and tree regeneration with generalized linear models. For each response variable, we fit models in which either dead tanoak BA (in 2008) or the number of dead tanoak stems (in 2008) was specified as the sole predictor variable (with and without squared terms). Response variables, data for all of which were collected in 2010, consisted of canopy cover and all measures of regeneration (seedlings, basal sprouts, saplings, and juvenile trees) for several species groups (all species combined, all non-tanoak species, non-tanoak hardwoods, and non-redwood conifers), as well as each tree species individually: tanoak, redwood, pacific madrone (*Arbutus menziesii*), California bay, bigleaf maple (*Acer macrophyllum*), Douglas-fir (*Pseudotsuga menziesii*), and California nutmeg (*Torreya californica)*.

To determine which species are beginning to replace tanoak, we examined tanoak mortality and species-specific regeneration patterns in areas heavily impacted by SOD (plots with $\geq$ 300 dead stems per ha and/or $\geq$ 15 m^2 dead BA per ha), termed "severe" plots (n=8), and we present relevant data for these plots. We also re-executed all of the models described above using only the "severe" plots. Finally, we identified three plots in which the total density of non-tanoak seedlings (in 2010) was less than the density of dead tanoak stems (in 2008), termed "regen-deficient" plots. Additional analytical details are provided in the full version of this paper (Ramage et al. 2011).

Results

Effects of tanoak mortality on canopy cover and regeneration

Canopy cover in 2010 was significantly affected by tanoak mortality in 2008 (BA: $p < 0.0001$; stems: $p = 0.0014$). Squared terms were not significant in either model; predicted values are curved because of the logit transformation prior to model fitting and subsequent back-transformation prior to plotting (*fig. 1*). In the plots with the greatest mortality, canopy cover was below 60 percent, while canopy cover was generally above 90 percent in plots with little or no mortality.

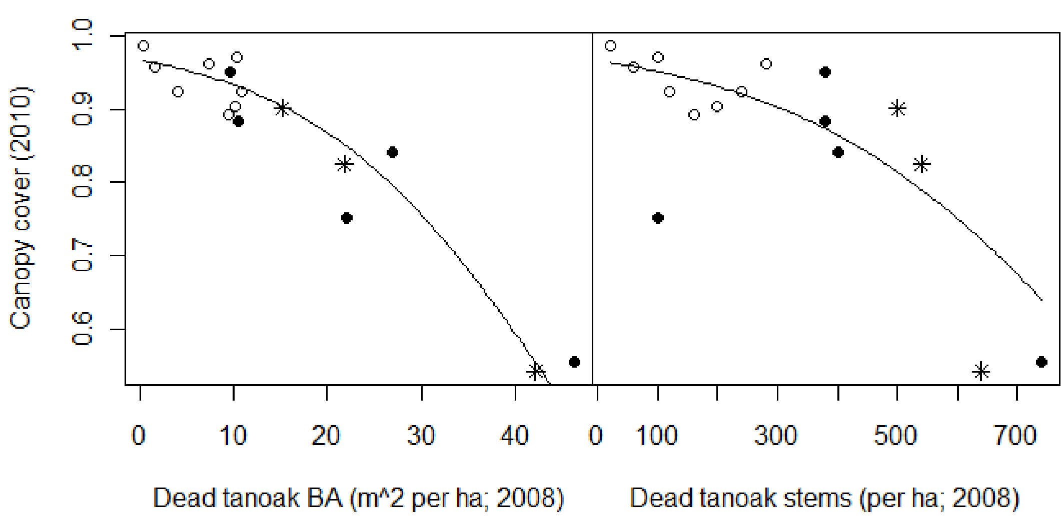

Figure 1—Canopy cover (2010) as a function of tanoak mortality (2008). Solid circles and asterisks were deemed severely impacted ("severe" plots); asterisks indicate "regen-deficient" plots; open circles represent all other plots (i.e., not "severe" or "regen-deficient").

Dead tanoak BA (in 2008) did not affect the density of tanoak seedlings, basal sprouts, or juvenile trees (in 2010), but the density of tanoak saplings was positively related to dead tanoak BA ($p = 0.0105$). The density of dead tanoak stems (in 2008) did not affect the density of tanoak seedlings, basal sprouts, or saplings (in 2010), but the density of tanoak juvenile trees was negatively related to dead tanoak stems ($p = 0.0007$). Squared terms were not significant in either model; predicted values are curved because of the log transformation prior to model fitting and subsequent back-transformation prior to plotting.

Tanoak mortality (in 2008) was entirely unrelated to regeneration (in 2010) of all species other than tanoak, including redwood. This was true across both tanoak mortality metrics (BA and stem counts) and all regeneration categories (seedlings, basal sprouts, saplings, and juvenile trees), and regardless of whether each species was analyzed individually or pooled into functional groups (non-tanoak hardwoods, non-redwood conifers). Similarly, tanoak mortality was unrelated to total regeneration (all species including tanoak and redwood), as well as all non-tanoak species. We also re-executed all these analyses using only the eight "severe" plots; all results were qualitatively identical, except for the relationship between tanoak juvenile trees and dead tanoak stems, which was insignificant in the "severe" analysis.

Severely impacted areas: regeneration and mortality

In the eight "severe" plots (those with $\geq$ 300 dead stems and/or $\geq$ 15 m^2 dead BA, per ha), the regeneration stratum was generally dominated by tanoak and redwood, but seedlings of other species were present at higher levels in certain areas. The median density of tanoak seedlings, basal sprouts, saplings, and juvenile trees was 1960, 4600, 180, and 0 per ha, respectively. Corresponding values for redwood were 1380, 1500, 100, and 40. Redwood accounted for 100 percent of non-tanoak basal sprouts, saplings, and juvenile trees, and the majority of non-tanoak seedlings in most plots (median non-tanoak seedlings = 1740; median redwood seedlings = 1380). Douglas-fir seedlings occurred in three of the eight severely impacted plots and exceeded densities of 3500 per ha in two of these plots. Pacific madrone and California bay seedlings were each present in four severely impacted plots, but their densities never exceeded 600 and 160 per ha, respectively. Seedlings of California nutmeg and bigleaf maple each occurred in only one plot, at densities of 240 and 40 per ha, respectively. Complete regeneration data for all "severe" plots are provided in *Table 1* of the full version of this paper (Ramage et al. 2011).

In these plots, the median amount of dead tanoak (absolute value and percent of total), in terms of BA and stem counts, was 21.9 (m^2 per ha; 66.4 percent) and 450 (per ha; 68.0 percent), respectively. As quantified by percent dead, the most severely impacted plot exhibited mortality exceeding 90 percent, in terms of both stem counts and BA. When quantified with absolute mortality, the most severely impacted plot contained 46.4 m^2 dead BA per ha and 740 dead stems per ha. The median amount of dead tanoak that was *broken/fallen* (bole broken at a diameter of $\geq$ 5 cm), in terms of BA and stem counts, was 15.8 (75.2 percent of dead) and 270 (79.1 percent of dead), respectively.

Three plots (which we refer to as "regen-deficient") exhibited a total density of non-tanoak seedlings that was less than the density of dead tanoak stems. Regeneration in these plots was consistently dominated by tanoak and redwood, but numbers were highly variable between plots and regeneration categories. With regard to other tree species, no basal sprouts, saplings, or juvenile trees were present in any plot, and seedlings were very uncommon. Seedlings of California bay (160 per ha) and bigleaf maple (40 per ha), the only other species present, occurred in only one plot each, both at densities insufficient to replace the number of tanoak trees that had already died by 2008. Densities of redwood basal sprouts, saplings, and juvenile trees, as well as tanoak regeneration (all categories combined), exceeded densities of dead tanoak stems in most (in the case of redwood) or all (in the case of tanoak) plots, but it is unlikely that these sources of regeneration will be able to fully re-occupy mortality gaps; this statement is justified in the discussion.

In the three "regen-deficient" plots, canopy cover was highly variable (54.2, 82.5, and 90.1), as was dead tanoak BA (42.2, 21.8, and 15.2 m^2 per ha). The density of dead tanoak stems was more consistent (640, 540, and 500 per ha), as was the percent of total tanoak that was dead, whether quantified by BA (93.0, 95.6, and 82.6 percent) or stem counts (86.5, 93.1, and 80.6 percent). The percent of dead tanoak that had broken/fallen was also consistently high, whether quantified by BA (81.5, 91.7, and 76.3 percent) or stem counts (78.1, 88.9, and 80.0 percent), suggesting that much of this mortality occurred well before our 2008 measurements.

Discussion

Broad regeneration patterns

Despite a significant reduction in canopy cover, our data suggest that SOD-impacted redwood forests in our study area are not exhibiting a regenerative response to tanoak mortality; tree regeneration was abundant in some mortality gaps, but regeneration levels were generally equivalent in severely impacted areas and relatively unaffected areas. These results were consistent across all mortality metrics, regeneration categories, and species (with the exception of tanoak saplings, which increased with dead tanoak basal area, and tanoak juvenile trees, which exhibited a negative relationship with dead tanoak stem density). Redwood and tanoak dominated the regeneration stratum in heavily diseased areas, and throughout the entire study area, while regeneration of other tree species was present only in isolated patches, and typically in very low densities.

Cobb et al. (2010) suggested that California bay may benefit more than any other tree species from SOD-induced tanoak mortality in redwood forests, because of similarities in growth form and size between tanoak and California bay, as well as positive feedbacks between inoculum loads and the abundance of California bay (which supports the most prolific sporulation of any host, but is not killed by *P. ramorum*; Davidson et al. 2008). However, at our study site, current regeneration patterns do not support this hypothesis. Although California bay seedlings occurred in 50 percent of "severe" plots, their densities were uniformly low (never exceeding 160 per ha, with a median of 20), and California bay basal sprouts, saplings, and young trees were entirely absent from all "severe" plots. In contrast, redwood regeneration occurred in all of these plots, and redwood seedlings alone had a median density of 1380 per ha (69 times that of California bay), suggesting that redwood is currently much better poised to claim the space previously held by tanoak. Regeneration patterns may differ in other affected areas (for example, stands with a greater abundance of mature California bay), but given that no previous studies have comprehensively examined tree regeneration in SOD-impacted redwood forests, researchers and land managers should consider the possibility that our findings will be applicable beyond our study area.

Deficiencies in regeneration

Although broad patterns indicate that regeneration is sufficient to replace dead tanoaks, we have identified some patches (1/20 ha in size) in which the density of dead tanoak stems exceeded the density of non-tanoak seedlings. Tanoak regeneration and other forms of redwood regeneration (basal sprouts, saplings, and juvenile trees) were abundant in some of these plots, but we have deemed these plots deficient in regeneration for the following reasons: (a) *P. ramorum* is established in all SOD-impacted areas and tanoak regeneration is thus unlikely to survive to maturity (Cobb et al. 2010), and (b) redwood basal sprouts will not be able to fully re-occupy large mortality gaps because these sprouts necessarily emerge at the base of existing redwood trees, which tend to exist in dense discrete clumps in second-growth redwood-tanoak forest (Douhovnikoff et al. 2004, Ramage and O'Hara 2010), and generally exhibit a strong vertical growth habit. The same rationale applies in large part to redwood saplings and young trees, many of which were associated with established trees and most likely of sprout origin. While additional regeneration will almost certainly appear in the future, an insufficient passage of time does not appear to fully explain the paucity of regeneration in some mortality gaps, or the corresponding absence of a regenerative response to tanoak mortality throughout the study area; in the full version of this paper (Ramage et al. 2011), we provide a justification for this assertion, as well as a thorough examination of mechanisms that may be inhibiting recruitment in SOD-impacted areas.

Tanoak regeneration patterns

The positive relationship between tanoak saplings and tanoak mortality (dead basal area) probably reflects: (a) increased growth rates of advance regeneration in mortality gaps, and/or (b) the initial tendency of tanoaks that are top-killed by SOD, as well as tanoaks that are infected but still living, to sprout vigorously (Cobb et al. 2010, Ramage et al. 2010). The absence of a relationship between tanoak mortality and tanoak seedlings or basal sprouts may: (a) indicate a balance between disease-induced recruitment and mortality within these regeneration classes, and/or (b) reflect the fact that tanoak seedlings and sprouts (which are extremely shade tolerant; Burns and Honkala 1990) are often abundant in healthy stands. The strong negative relationship between tanoak juvenile stems and tanoak mortality (dead stems) suggests that individuals in this size class (3 to 10 cm DBH) are suffering high rates of SOD-induced mortality and/or not recruiting in diseased areas.

Conclusions and management considerations

SOD may ultimately create a *niche opportunity* (an opportunity for an absent or uncommon species to invade or increase in abundance; sensu Shea and Chesson 2002), but we have not discovered evidence that this phenomenon is occurring in the redwood forests of our study area. Rather, tree species other than redwood and tanoak have made only small and highly variable incursions into mortality gaps, and some areas appear to lack sufficient regeneration for full re-occupancy of growing space, demonstrating that the future composition of SOD-impacted redwood forests is still far from certain. The ultimate ability of potential tanoak replacement species to co-exist with redwood in areas previously dominated by tanoak may only be apparent if and when such species are able to recruit in high numbers; at present, dispersal and recruitment limitation (both of which may be highly stochastic) are likely the dominant community assembly processes, but as these species begin to actively compete in areas previously occupied by tanoak, deterministic niche-related processes may become more important. For instance, tanoak develops a deep taproot (Burns and Honkala 1990), a characteristic that likely helps it to co-exist with redwood (which does not develop a taproot; Burns and Honkala 1990); this divergence in root morphology suggests that other deeply rooted tree species may be best equipped to compete with redwood in the absence of tanoak.

Numerous long-term impacts may result from SOD-induced tanoak decline (for example, trophic cascades resulting from the loss of tanoak acorns, reduced *resistance* and/or *resilience* in the face of future threats; sensu Suding et al. 2004). Land managers who wish to minimize the threat of such impacts should consider the intentional establishment of other native tree species in heavily impacted areas. Such efforts could optionally focus upon species at or near the northern extent of their range, in anticipation of generally warming climatic conditions. By choosing to direct ecological trajectories, managers may successfully alter long-term characteristics such as species composition and stand structure, but such actions will be most efficient in the early stages of community assembly (Thompson et al. 2001). Furthermore, because SOD-induced tanoak mortality gaps are a novel occurrence, and novel ecosystems are likely to present unfamiliar and unforeseen challenges (Hobbs et al. 2006), it is prudent to assume that successful plantings (or other mitigation actions) may require considerable experimentation. As an alternative approach, managers may opt to actively maintain the open nature of these sites, so that if and when SOD-resistant tanoak genotypes are discovered, these genotypes can be readily reintroduced into areas where tanoak previously dominated.

Acknowledgments

We thank Save the Redwoods League for providing funding for this research and the University of California-Berkeley Forest Pathology Laboratory for molecular testing for *Phytophthora ramorum* in our study plots. In addition, we thank the Marin Municipal Water District for allowing this study to be conducted on their land, and two anonymous reviewers for valuable feedback. Finally, we are very grateful for the excellent work of our dedicated field assistants (Ben Ewing, Karla Martinez, and the many others who donated their time).

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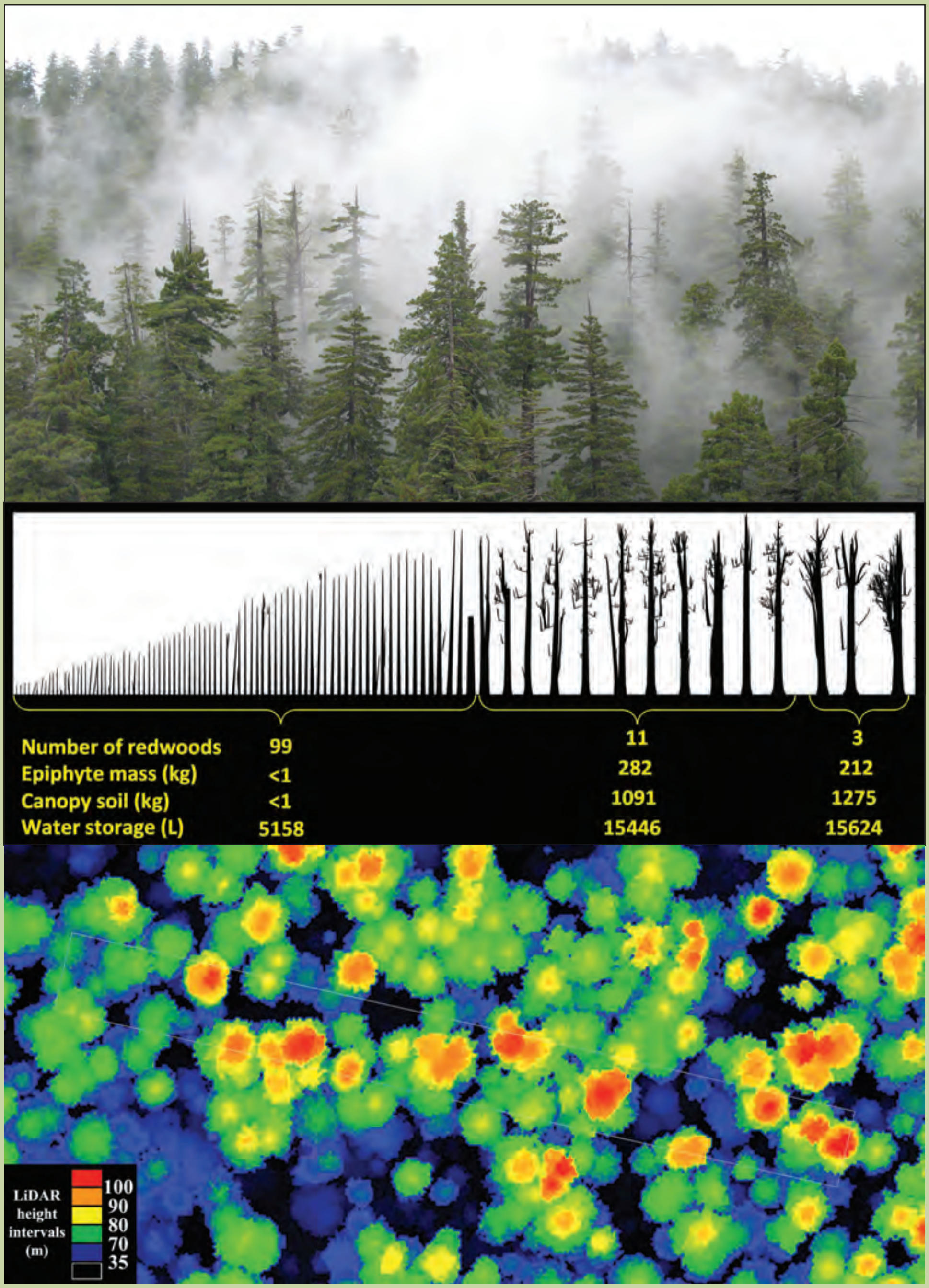

Sonoma Tree Vole Habitat on Managed Redwood and Douglas-fir Forestlands in North Coastal California[1]

Sal J. Chinnici,[2] David Bigger,[3] and Eric Johnson[4]

Abstract

The Sonoma Tree Vole (*Arborimus pomo*) – a small arboreal mammal associated with mature forests – is a California Species of Special Concern due to concerns regarding loss of habitat from harvest, fire, and conversion. By counting their nests, we examined *A. pomo* use of pole to mature forest seral stages from 2001 to 2005 using line transects at 64 study sites distributed across redwood (*Sequoia sempervirens*), mixed conifer, Douglas-fir (*Pseudotsuga menziesii*), and hardwood dominated stands. A total of 441 nests were found including 215 active and 226 inactive nests. The highest percentage (33 percent) of nests was in unharvested and partially harvested old growth Douglas-fir stands. All pole and young stands, and stands that were predominantly redwood, had 77 percent fewer nests. This study suggests that *A. pomo* could benefit from forest management strategies aimed at retaining a mature Douglas-fir stand component.

Key words: conservation, Douglas-fir, forest inventory, habitat, modeling, redwood, Sonoma tree vole

Introduction

Tree voles (*Arborimus* sp.) are small nocturnal mammals that primarily inhabit coniferous forests dominated by Douglas-fir, but they also live where Douglas-fir co-occurs with other species, including redwood, Sitka spruce (*Picea sitchensis*), western hemlock (*Tsuga heterophylla*) or grand fir (*Abies grandis)* (Jones 2003). Even in such forests, populations appear to have a patchy distribution (Carey 1991).

With their arboreal nature and diet almost entirely of Douglas-fir needles, tree voles are among the most unique and highly specialized rodents in the world. Further, their habitat use patterns and behavior make studies problematic. As a result, tree vole presence and relative abundance is most commonly inferred by their nests, distinguishable from other nests by discarded resin ducts from the needles they have consumed (Carey 1991, Thompson and Diller 2002).

The abundance and distribution of suitable habitat may be a limiting factor for tree voles (Carey 1989). Although some studies (Swingle 2005, Thompson and Diller

[1] This paper was presented at the redwood science symposium: coast redwood forests in a changing California. June 21-23, 2011, Santa Cruz, California.

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2002, Wooster and Town 1998[5]) have shown that tree voles nest in younger forests, several studies (Aubry and others 1991, Corn and Bury 1986, Gillesberg and Carey 1991, Gomez and Anthony 1998, Huff et al. 1992, Martin 1998) suggest that the species is most abundant in older forests. Thus, timber harvest may impact the species.

In northwestern California, the Sonoma tree vole is restricted to coastal forests of Humboldt, Mendocino, and Sonoma Counties (Jones 2003). The Sonoma tree vole is recognized in California as a Species of Special Concern (California Department of Fish and Game 2011). In addition to its status, tree voles are important prey species of the northern spotted owl (Forsman and others 1984), the Humboldt marten (*Martes americana humboldtensis*), and the Pacific fisher (*Martes pennanti pacifica*)[6].

The Sonoma tree vole is a covered species of the Humboldt Redwood Company (HRC) Habitat Conservation Plan (HCP) (PALCO 1999). A HCP management objective for the Sonoma tree vole is to sustain viable tree vole populations through retention of suitable habitat. Given the difficulties surrounding the study of this species and the questions concerning its habitat, we developed this study to test the assumptions regarding the availability and distribution of habitat that then could inform models relating timber management to vole abundance, and map current distribution on HCP lands.

For this case study we: (1) examined Sonoma tree vole use of pole to mature forest seral stages from 2001 to 2005 at 64 study sites distributed across redwood, mixed conifer, Douglas-fir and Douglas-fir/hardwood stands, (2) conducted a statistical analysis to quantify the relationship between the number of nests and forest stand features, and (3) applied them to the current (2011) forest inventory to assess the availability and distribution of Sonoma tree vole habitat.

Study area

HRC lands encompass approximately 84,000 ha, and are located in coastal Humboldt County, California. These lands are characterized by mountainous terrain, a maritime climate, and dense coniferous forests, primarily dominated by coastal redwood and Douglas-fir, with an understory typically composed of tanoak (*Lithocarpus densiflorus*), Pacific madrone (*Arbutus menziesii*), salal (*Gaultheria shallon*), and sword fern (*Polystichum munitum*). Elevations range from 45 m on river or creek benches to over 800 m along ridges.

Methods

Study sites (n = 64) were randomly selected based on the following stand types: (1) Douglas-fir, (2) redwood, (3) Douglas-fir/hardwood, and (4) Douglas-fir/redwood. All sites contained canopy closure >25 percent and were a minimum of 10 ha in area. Within each type, 16 sites were selected among three seral stages: pole (15 through 28 cm Diameter at Breast Height [DBH]), young-growth (29 through 61

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[5] Wooster, T.W.; Town, P. 1998. **California red tree voles within a coastal second-growth forest.** State of California, Department of Fish and Game, Yountville, CA. Unpublished report.
[6] Slauson, K. 2011. Unpublished data on file, USDA Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, Arcata, CA.

cm DBH), and mature (>61 cm DBH).

From 2001 to 2005, we surveyed according to the draft study plan developed by the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station[7]. Following Anderson and others (1979), surveyors recorded tree vole nests along five 100 m transects in each stand. We attempted to distribute transects 60 m apart and parallel to elevation contour lines. The UTM locations of each transect's mid-point was located with a GPS.

We attempted to locate Sonoma tree vole nests with a visual search of trees along both sides of the transect line. When a suspected nest structure was detected, the ground below the nest was searched for resin ducts, evidence indicative of tree vole activity (Jones 2003). If a nest could not be confirmed as belonging to a tree vole from the ground, the tree was climbed and the nest was inspected.

A general linear mixed-model analysis of variance (ANOVA) was used to quantify the relationship between the number of nests and habitat type as derived from forest inventory information. The number of nests per transect was the dependent variable. Tree size class (20 to 40 cm, 40 to 60 cm, 60 to 80 cm, and 80 to 100 cm) and density of Douglas-fir by canopy cover (trace [<25 percent], sub-dominant [25 to 50 percent], dominant [50 to 75 percent], and pure [>75 percent]) were linear continuous effects. Site was treated as a random effect. The variance component was selected for the covariance structure. The full maximum likelihood estimation was used to model the fixed effects.

To address the habitat mapping objective using 2011 forest inventory information, the Sonoma tree vole nest model was applied to HRC lands in the Mattole River watershed near Petrolia, CA using current vegetation type mapping and forest inventory data.

Results

Field study

A total of 441 Sonoma tree vole nests in the 64 sites were found, with 215 active and 226 not active. Occupancy was detected in 57 of the 64 surveyed sites (89 percent). The sites varied widely in terms of stand area and vegetative characteristics. Stand size ranged from relatively small stands of 10 ha (minimum stand size) to relatively large contiguous stands of 898 ha. Stand characteristics ranged from un-harvested old growth stands with dense canopies, to thinned pole sites with thick brushy understory vegetation.

The greatest numbers of nests were in Douglas-fir trees (*table 1*). Twenty-nine trees were found to contain multiple vole nests. The diameter breast height (DBH) of trees with active nests ranged from 10.5 to 226.3 cm (mean 100.6 cm).

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[7] Biswell, B.; Blow, M.; Finley, L.; Madsen, S.; Schmidt, K. 1999. **Survey protocol for the red tree vole *Arborimus longicaudus.*** (Interim Version 2.0). U.S. Forest Service, Pacific Northwest Research Station. Portland, OR. 31 p. Unpublished.

Table 1—*Percentage of Arborimus pomo nests by tree species on HRC land in northwestern California, 2001 to 2005 (n = 395).*

Common name	Scientific name	# *A. pomo* nests	Percent of total nest trees
Douglas-fir	*Pseudotsuga menziesii*	337	85.3
Grand fir	*Abies grandis*	29	7.3
Tan oak	*Lithocarpus densiflorus*	17	4.3
Redwood	*Sequoia sempervirens*	8	2.0
Pacific madrone	*Arbutus menziesii*	2	0.5
Interior live oak	*Quercus wislizenii*	1	0.3
Pepperwood	*Umbellularia californica*	1	0.3

Sonoma tree vole nests were in all seral stages except the young growth Douglas-fir/hardwood type (*fig. 1*). By stand type and seral stage, the largest number (33 percent) of vole nests was in mature Douglas-fir stands, followed by mature redwood/Douglas-fir stands (21 percent). Within stand types, mature stages contained the most nests and pole had the least. Approximately 11 percent of Sonoma tree vole nests were in pole Douglas-fir/hardwood stands. Contrary to the general positive relationship of nest density with seral stage, this finding may be due to the presence of residual old growth Douglas-fir trees at a density that did not affect stand classification but still provided the structure for nesting and an adequate food source. Sonoma tree vole nests were not found in three of the hardwood pole sites, two of the redwood pole sites, and two of the redwood young growth sites.

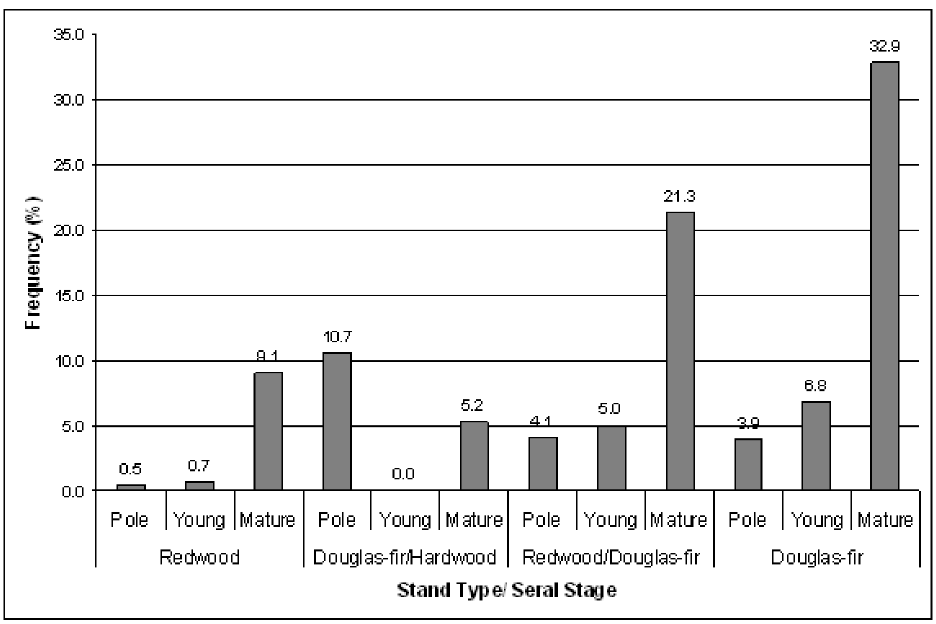

Figure 1—Percent of *Arborimus pomo* nests in pole, young, and mature stands relative to species composition for stands surveyed on HRC land in northwestern California, 2001 to 2005 (n = 441).

Habitat analysis

When the stand size class increased, the number of nests per transect also increased (*fig. 2*). The estimated slope describing the relationship between tree size class and number of nests (b = 0.659 ± 0.174 (±1SEM)) was significantly greater than zero (t62 = 3.79, $p < 0.001$,). When the relative density of Douglas-fir increased from stand to stand, the number of nests per transect also increased (*fig. 3*). The estimated slope describing the relationship between relative density of Douglas-fir and nests (b = 0.519 ± 0.177 (±1SEM)) was significantly greater than zero (t62 = 2.931, p = 0.005).

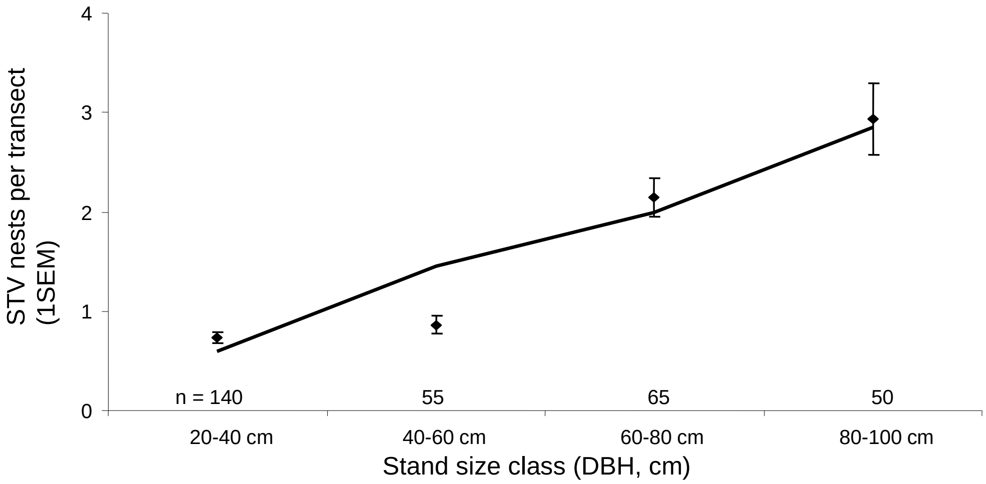

Figure 2—Number of Sonoma tree vole (STV) nests in relation to a stand's tree size class. Each point is the mean number of nests across transects, error bars are ±1SEM, n is the number of transects that were surveyed.

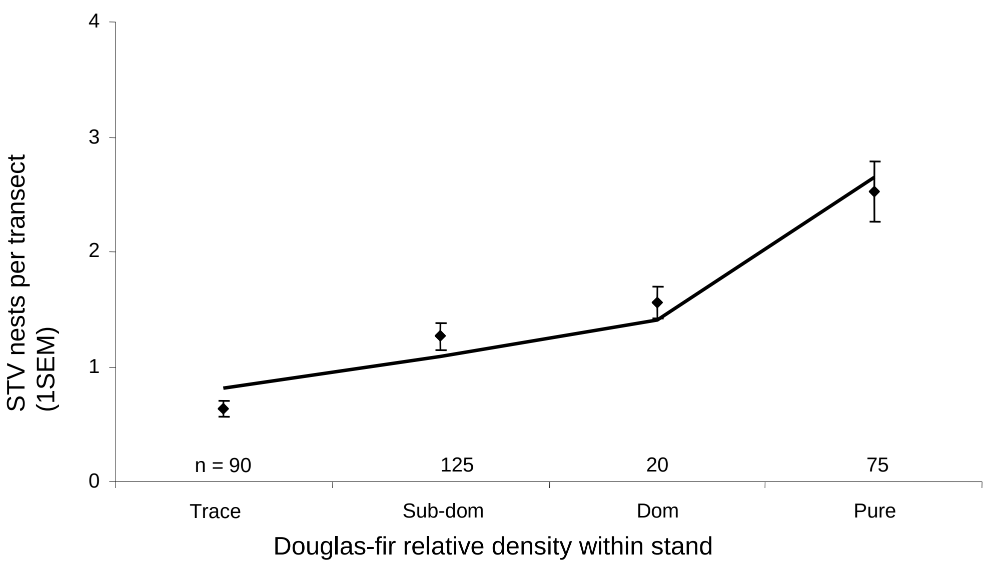

Figure 3—Number of Sonoma tree vole (STV) nests in relation to Douglas-fir relative density within stand. Each point is the mean number of nests across transects, error bars are ±1SEM, n is the number of transects that were surveyed.

The following equation describes the model:

Nests = -1.934 + 0.659(SIZECLASS) + 0.519(DFRANK)

Where *Nests* is the number vole nests along a 100 m transect. SIZECLASS is the tree size class where 3 = 20-40 cm; 4 = 40-60 cm; 5 = 60-80 cm, and 6 = 80-100 cm. DFRANK is the relative density of Douglas-fir where 0 = trace; 1 = sub-dominant; 2 = dominant; and 3 = pure.

Application of habitat model to current (2011) timber inventory

The Sonoma tree vole nest model was applied to HRC lands using current (2011) vegetation type mapping. As an example, the estimated number of Sonoma tree vole nests per transect and the area of relative tree vole habitat are shown for the Mattole River watershed *(table 2*).

Table 2—*Relative density of Sonoma tree vole nests predicted per 100 m transect and habitat area relative to Douglas-fir size and rank from the 2011 HRC forest inventory in the Mattole River watershed on HRC land in Humboldt County, CA.*

Size Class	DF Rank	Nests/transect	Acres	Hectares
3	0	0.0	4847.3	1938.9
3	1	0.6	644.4	257.8
4	0	0.7	255.7	102.3
3	2	1.1	576.3	230.5
3	3	1.6	726.3	290.5
4	3	2.3	1702.5	681.0
5	3	2.9	2047.7	819.1
6	3	3.6	529.4	211.8

The habitat information resulting from the vegetation typing was used to map the estimated density per transect of vole nests on HRC lands in the Mattole River watershed (*fig. 4*). On *figure 4* black and dark grey areas represent locations where nests should be dense or moderately dense respectively. Stippled areas represent places the model predicts nests to be absent or few, or are non-forested.

In the Douglas-fir dominated Mattole River watershed, approximately 38 percent of habitat (1711.2 ha) is in the high density category and is scattered throughout the forestlands, with some relatively significant concentrations to the northwest on Long Ridge and near Taylor Peak (*table 2* and *fig. 4*).

With the exception of Long Ridge and Taylor Peak, the best habitat appears to be relatively scattered but connected by stands with lower suitability and riparian zones. Thus, in this watershed retention of clumps of habitat and the connectivity between them would likely benefit Sonoma tree voles.

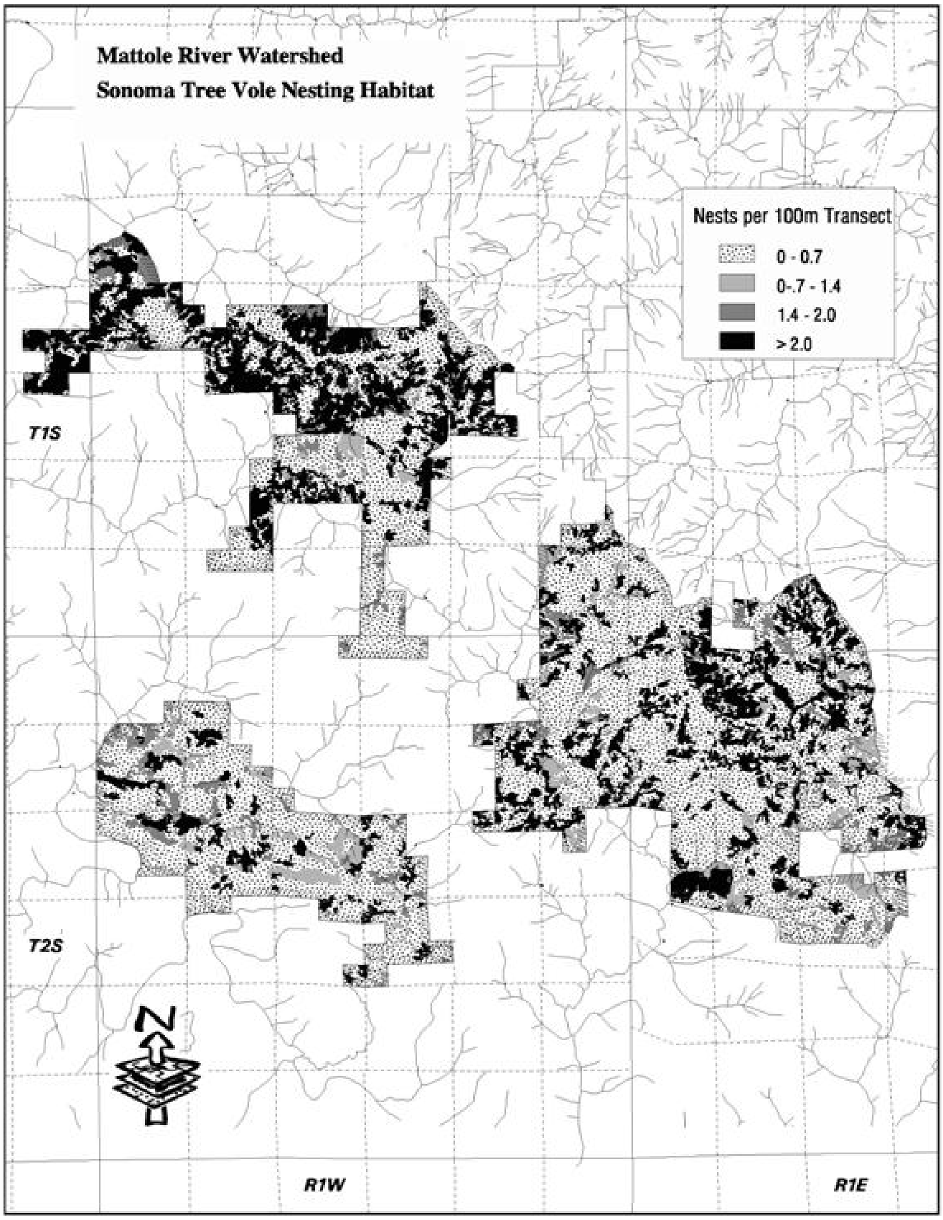

Figure 4—Relative density of Sonoma tree vole nests on HRC lands in the Mattole River Watershed, Humboldt County, CA.

Management implications

Swingle (2005) found through comparisons of nests located by visual searches from the ground versus nests located by following radio collared voles that many active nests could not be seen from the ground, and that nests located by visual searches were biased towards larger nests. Thus, our surveys should best be considered indices to relative abundance because we do not know the true number of nests in any stand type. Findings of this study relative to Sonoma tree vole habitat are consistent with others (e.g., Jones 2003, Thompson and Diller 2002). Mature stands with larger Douglas-fir trees and those with higher densities of Douglas-fir tend to have the most nests while pole and young stands with relativity few or no Douglas-fir trees tend to have fewer nests (*fig.2* and *fig.3*). The retention of large Douglas-fir trees combined with other conservation strategies such as riparian protection, other species protection measures (e.g. northern spotted owl nest zones), or as part of a structural element retention strategy wherein large trees with complex structure are retained may help maintain Sonoma tree vole populations.

Maps of Sonoma tree vole habitat derived by applying the habitat model to timber inventory (e.g., *fig.4*) as the inventory is renewed over time can enable changes in Sonoma tree vole habitat to be visualized and monitored. Validating the model by predicting Sonoma tree vole nest density in stands outside the stands and watersheds where the model was developed would add confidence to its use for monitoring purposes.

Looking forward, HRC plans to use the ForSee growth and yield model (California Growth and Yield Modeling Cooperative 2009) to project future stand conditions under planned management scenarios. The ForSee model produces present and future stand characteristics that can be used as input to the vole nest model enabling forecasts of tree vole habitat value. Next steps for HRC will be to explore opportunities for refinement of the model using new or different inputs. Other forestland ownerships that have forest inventory data and Geographic Information System technology may be able to use these techniques as a tool for evaluating species conservation programs.

Acknowledgments

We wish to thank Christiana and Dana Laughlin for their tremendous effort in conducting the field work and gathering the data necessary to prepare this paper. Without their help it would not have been possible to complete this project.

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Two Decades of Research and Monitoring of the Northern Spotted Owl on Private Timberlands in the Redwood Region: What do We Know and What Challenges Remain?

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Abstract

Surveys for northern spotted owls on Green Diamond Resource Company's (formerly Simpson Timber Company) ownership in coastal northern California were initiated in 1989. The following year, a long-term demography study was initiated that has continued to the present time. A Habitat Conservation Plan was developed for the species in 1992 and numerous habitat studies followed. The extensive dataset generated was used to estimate the trend in owl numbers, develop resource selection functions for nighttime activity and nesting habitat, and analyze the factors influencing spotted owl survival, fecundity and habitat fitness (i.e., ability of the habitat to support a stable population of owls). Important conclusions generated to date include that habitat heterogeneity (i.e., juxtaposition of young and older stands) is critical to both survival and fecundity, as is precipitation during the early nesting season. A landscape projection of current and future habitat indicated an abundance of high quality habitat that has the potential to support an increasing population of owls. Demographic analysis found the owl population was stable from 1990 to 2001, but has declined in recent years. The decline coincided with an apparent increase in barred owls. Growing evidence including barred owl removal experiments indicate that the invasion of barred owls into northern California is responsible for the decline. Preliminary results of the recently initiated removal experiment suggest that control of barred owls is feasible, and that spotted owls respond rapidly and favorably where barred owls are removed.

Key words: northern spotted owl, habitat fitness, habitat conservation plan, demography, barred owl removal experiment

Introduction

There has been extensive research on northern spotted owl (*Strix occidentalis caurina*) habitat requirements over the years. The majority of these studies have been conducted in landscapes with significant amounts of mature or old forests, which are the principal habitats for this species in most areas where it has been studied (Courtney et al. 2004). However, almost nothing was known about the northern spotted owl on private managed timberlands until the Northern Spotted Owl Status Review Supplement was published on April 21, 1989 by the US Fish and Wildlife Service (FWS). This review concluded that listing the northern spotted owl as a threatened species throughout its entire range was warranted, which we observed to have unleashed a flurry of spotted owl surveys on private lands in northern California. Included in this survey effort was the initiation of spotted owl surveys

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across Green Diamond Resource Company's (formerly Simpson Timber Company) in north coastal California.

Contrary to expectations, we found large numbers of spotted owl on Green Diamond's (GDRC) ownership, which put the company on a collision course with the federal Endangered Species Act (ESA) when the owl was listed in 1990. In response, GDRC launched a long-term research and monitoring program on spotted owls that still continues today, they developed and got approved the first Habitat Conservation Plan (HCP) for spotted owls in 1992 and are in the process of developing a new HCP for spotted owls along with coverage for fishers (*Martes pennanti*) and tree voles (*Arborimus longicaudus* and *A. pomo*). Here we report the major findings of 2 decades of research and monitoring on spotted owls and describe the new challenges that threaten the long-term viability of this iconic species of the Pacific Northwest.

Methods

Each year, we attempted to locate all individual territorial spotted owls within the entire study area. Surveys were initiated each year beginning 1 March using protocols initially adapted from Forsman (1983) and further modified to support GDRC's approved spotted owl HCP. Owls were initially located primarily at night using vocal imitations or recordings of their calls. Daytime surveys were used primarily to locate roosts and determine the status of owls at sites where they had been previously located or where nighttime responses had been heard. Once an owl was located, it was typically offered live mice to determine its identity (if previously captured and marked), paired and reproductive status (Forsman 1983). Most nests were initially located by following the male back to the nest where the male would commonly attempt to deliver the mouse to the female. Once a nest was located, it was revisited one or more times after the typical period for fledging (late May through early June) to determine the status of the nesting attempt.

Most owls were captured with a snare pole and marked with a unique numbered FWS band attached to one tarsus and a plastic color band attached to the other tarsus. For the radio telemetry study, we fitted owls with tail-mounted radio transmitters (Holohil, model RI2C 6 g) equipped with either a 1- or 2-year battery (total mass = 5.5 or 7.5 g, respectively). We attached radio transmitters to the proximal portion of the rachis of the two central rectrices using small zip-ties and epoxy (#332 Titan Corp., Lynnwood, WA). The time that transmitters remained attached to individual birds ranged from 1 to 15 months (mean ≥ 7 months) until the rectrices were molted. Owls fitted with radio transmitters were tracked with a two-element, hand-held directional H-antenna (Telonics model RA-14) and portable receiver (Telonics model TR-4). We radio tracked owls during all seasons, from April 1998 through September 2000. Triangulations were attempted on two to four birds per night from roadside locations throughout the study area, with subsequent visual sightings if possible. When located, a bird was followed for 4 to 8 hours or until lost. Locations were recorded when the bird changed roosting sites, or every 30 minutes, whichever came first.

Results

Distribution

The first surveys of northern spotted owls on GDRC's ownership in 1989 did not cover the entire ownership, but it did demonstrate that spotted owls occurred throughout the majority of the ownership and their population density was unusually high in some regions. Surveys of all the main contiguous blocks of GDRC's ownership occurred every year since 1989 and the pattern that emerged indicated that spotted owls were located throughout the ownership, but there were substantial differences in the density of owl sites. In general, owl densities were highest in regions with a mixture of mature second growth and young regenerating stands, and few owls were found in homogeneous stands of second growth. There was also a pattern of high density of owl sites distributed lower on the slopes along rivers and major creeks. A study based on surveys from 1990 to 1997 indicated that two regions (Korbel and Mad River) produced the highest estimates of crude densities for northern spotted owls, but there have been few statistically rigorous attempts to estimate population density throughout its range (Diller and Thome 1999).

Habitat

GDRC's original 1992 HCP was developed when there had been little spotted owl research in the coastal redwood region and only 3 years of site-specific owl surveys and research in the study area. As a result, only simplistic definitions of habitat existed and suitable habitat for spotted owls was defined as forest stands >30 years old, because at least some stands in this age class were known to be used by spotted owls for foraging, roosting, and nesting (Folliard 1993). At the time, it was assumed that recently regenerated stands (0 to 5 years) had no direct value to owls. Stands 6 to 30 years were known to be woodrat habitat (Hamm 1995) and therefore potential spotted owl foraging habitat. Foraging, roosting, and occasional nesting occurred in stands 31 to 45 years old, and forest stands >45 years old were considered to be prime nesting and roosting habitat as well as foraging habitat.

To describe and quantify the unique site-specific habitat used by spotted owls on a managed forest in the redwood region, GDRC used the extensive research and monitoring to develop a more sophisticated and spatially explicit definition of spotted owl habitat. Determining what habitat spotted owls used during their period of nocturnal activity was the first step. To gather data on what habitat spotted owls used at night, we conducted a radio telemetry study from 1998 to 2000 that included 28 total owls that were followed all or a portion of that time. The resulting data was used to construct 95 percent kernel distributions based on what owls used versus a random selection of available points within the same area. These data were used to develop a resource selection function (RSF) for spotted owl nighttime activity. The top model indicated that owls tended to be found low on the slope in areas composed of approximately 70 percent age class 41+ years with a high percentage of hardwood. Furthermore, selection was highest if the nearest stand to the owl's location was either 6 to 20 or 21 to 40 years, and lower if the nearest stand was either 0 to 5 or 41+ years. In other words, when active at night, spotted owls on GDRC's ownership were most likely to be found in older more complex forest stands that were in close proximity to younger stands that have high densities of woodrats (Diller et al. 2010).

To further refine spotted owl habitat, we conducted an analysis of the habitat selected for nesting by spotted owls on GDRC's timberlands. Based on the locations of 182 successful nests (fledged at least one young) from 1990 to 2003, we estimated a RSF to characterize the habitat of an 'average' successfully nesting spotted owl. The top model for managed timberlands indicated that the relative probability of locating a successful nest increased with age of the stand and 'edge density' within a 600 m radius of the nest. 'Edge density' was calculated as m of edge (line between any two age classes) per ha, or in other words, the amount of stand age heterogeneity. In addition, selection was greatest in stands with approximately 55 percent basal area of residual older trees, 30 percent hardwood basal area and a large amount of good nighttime activity habitat within a 400 m radius of the nest. This indicated that for nesting, spotted owls were selecting older more complex stands that were in fairly close proximity to areas that had a high potential as foraging habitat (Diller et al. 2010).

Habitat fitness

'Fitness' (i.e., ability to survive and reproduce) has traditionally been considered an individual attribute, but the quality of the habitat occupied by a particular individual also influences its fitness. Therefore, the quality of the habitat relative to its impact on the fitness of individuals occupying that habitat can be defined as 'habitat fitness' (Franklin et al. 2000). Combining the influence of habitat on both survival and reproduction provides the ultimate measure of habitat quality such that areas with high habitat fitness are capable of supporting a stable or increasing 'source' populations while areas of low habitat fitness are associated with habitat 'sinks.' GDRC's long term demographic study along with geographically referenced and relatively detailed forest stand information on all parts of its ownership made it possible for us to directly relate habitat characteristics with survival and fecundity to estimate site-specific habitat quality (i.e., habitat fitness) for spotted owls.

To estimate the impact of habitat on survival, we utilized capture-resight data from 1990 to 2003 and we used nesting information from the same time period to analyze the impact of habitat on fecundity. We included both habitat and non-habitat covariates such as regional precipitation and mean air temperature in the analyses along with some covariates related to the 1992 spotted owl HCP conservation program. The HCP allowed harvesting of a limited amount of occupied owl habitat (i.e., 'take' of the owl site), which provided a unique opportunity to assess the direct impacts of timber harvesting on spotted owls. We also analyzed the effect of 39 no-harvest set-asides totaling 10,331 ac that were designated as part of the spotted owl conservation strategy in the HCP.

Increased days of precipitation during the early nesting entered with a negative coefficient the top survival and fecundity models. In contrast, increased temperatures during early nesting entered the top survival model with a positive coefficient. Positive effects on survival were also associated with increased nest site selection values (i.e., areas good for nesting conferred higher survival probabilities). Four habitat covariates were associated with higher fecundity, but collectively they were representative of areas having higher habitat heterogeneity. Harvesting or take of an owl site did not enter the survival model, but it did have a negative effect on fecundity. The effect of set-asides was complex with the highest survival and fecundity associated with areas near (< ½ mile) but not inside set-asides (Diller et al. 2010).

From the average survival and fecundity at a specific location, the largest Eigenvalue of the Leslie projection matrix was computed and defined to be the habitat fitness of the site. Relative to other categorical variables, habitat fitness was most sensitive to the location of the nest site/activity center relative to a set-aside.

Habitat fitness values were highest in the ½ mile buffer surrounding a set-aside with all other covariates being realistically equal. While considerably lower relative to the magnitude of the effect, take (i.e., harvesting an owl site) was the second most important categorical variable relative to habitat fitness. Relative to continuous variables, habitat fitness was most sensitive to changes in precipitation during the early nesting period such that increases in the total number of days of measurable precipitation within the early nesting period caused habitat fitness to decline. The second most important continuous variable was edge density, where increases in this variable resulted in higher values of habitat fitness. Relative to latent variables (i.e., variables generated from other models), habitat fitness was most sensitive to changes in survival followed by changes in fecundity and nesting habitat (Diller et al. 2010).

Using anticipated harvest plans in the next 10 years and projected harvests derived through a newly developed harvest schedule model, we were able to project spatially explicit stand conditions into the future. Assuming important non-habitat variables (e.g., weather and barred owls) remain at the median values from the past, we were able to project spatially explicit estimates of habitat fitness on GDRC's study area at 10 year intervals from 2010 to 2060. The changes in habitat fitness across GDRC's ownership were dynamic at the sub-basin level with specific areas waxing and waning in their relative habitat value for owls. However, the overall proportion of the ownership in the highest category of habitat fitness (i.e., habitat capable of supporting an increasing population of spotted owls) increased from 35 percent in 2010 to 64 percent of the ownership in 2060. In 2060, a total of 87 percent of GDRC's ownership is projected to be in the two highest categories of habitat fitness, which is projected to be capable of supporting a stable or increasing populations of spotted owls if other non-habitat variables (e.g., weather and barred owls) remain within acceptable limits (Diller et al. 2010).

Based on a sensitivity analysis of habitat fitness, the habitat variables that most likely contributed to this trend were edge density and the proportion of older stands (41 to 60 years) adjacent to younger stands (6 to 20 and 21 to 40 years). Implementation of GDRC's aquatic HCP that was approved in 2007 will create a future landscape in which an estimated 20 to 25 percent of the landscape will be in older stands associated with protected riparian or geologic reserves. Along with smaller clearcut openings mandated by the California Forest Practice Rules, the net affect will be much greater overall open edge density and a higher overall level of habitat heterogeneity, which appears to be highly beneficial to spotted owls in GDRC's ownership. In summary, current models based on site-specific habitat data indicate a strong positive trend in habitat quality for spotted owls on GDRC's ownership.

Population trend

Mark-recapture studies were initiated throughout GDRC's ownership in 1990 to estimate key demographic parameters and trends in the population. Along with other range-wide demographic studies of northern spotted owls, GDRC participated in three previous meta-analyses in 1998, 2004 and 2009. Results from the most recent meta-analysis that analyzed GDRC data from 1990 to 2008 indicated that mean apparent annual survival probabilities of adult spotted owls on GDRC land were 0.851 and 0.853 for males and females, respectively, while estimated mean annual fecundity for adult spotted owls on GDRC land was 0.305. The estimated rate of spotted owl population change ("lambda RJS") on GDRC land was 0.972, which is similar to but slightly lower than the two nearby Willow Creek and Hoopa study areas. The trend in estimates of the realized population change indicated that the population of spotted owls on the GDRC study area was apparently stable or increasing until 2001 after which the population began an apparent downward trend. The barred owl covariate entered the top model for both survival and fecundity, which suggested that barred owls were the most likely cause for the recent decline of spotted owls on GDRC's study area (Forsman et al. in press).

Potentially, this downward trend was reversed with a 26.6 percent increase in the number of occupied owl sites on GDRC's density study area in 2009 followed by a 3.0 percent increase in 2010. Factors that may have contributed to this increase included modifications (i.e., using high quality digital callers and including a greater variety of recorded calls) of the survey protocol in 2009 to increase spotted owl detection rates, which resulted in locating banded resident spotted owls in historical sites that had appeared to have been abandoned in recent years. In addition, the lower Mad River Tract has large areas of third growth that apparently were just now reaching suitable habitat attributes for colonization by spotted owls. GDRC also initiated a barred owl removal experiment in 2009, which involved removing all barred owls from treatment areas (i.e., approximately half of the total study area). Barred owls were removed from historical spotted owl sites, which allowed these sites to be re-colonized by spotted owls and these treatment areas could be colonized by spotted owls free from interference from this expanding competing species (see preliminary barred owl results below). Therefore, the recent improved trend probably was a combination of improving survey techniques, increasing amounts of suitable habitat, and freeing approximately half of the study area from barred owls.

Response of spotted owls to experimental removal of barred owls: preliminary results

In 2006, Green Diamond assisted the California Academy of Sciences (CAS) in obtaining a small collection of barred owls in California. These initial collections provided an opportunity to do preliminary "removal case studies" that would document the response of individual spotted owls to the removal of barred owls. Seven barred owls were collected from four different historical spotted owl sites during May and June 2006 on GDRC's ownership in Humboldt County. Although based on just four "case studies", these results suggested that spotted owls were quick to re-colonize their former territories following removal of barred owls.

As part of the implementation of the recovery plan, a Barred Owl Work Group (BOWG) was formed to consider implementation of a suite of barred owl removal studies (USFWS 2008). The BOWG provided full support for a study on GDRC's ownership that was designed to be complementary to other removal experiments that were being proposed for public lands in Washington, Oregon and California. Still working under CAS scientific collecting permits that allowed removal of 20 additional barred owls, a preliminary barred owl removal experiment was initiated on a portion of GDRC's spotted owl demographic study area in 2009. The study was designed to determine the impact of barred owls on site occupancy, survival and fecundity of spotted owls using paired treatment and control areas where barred owls were removed in treatment areas, while they were allowed to increase 'naturally' in control areas. The additional collections in 2009 also allowed for added removal case studies of the response of individual spotted owls to the removal of barred owls from sites that were previously known to be occupied by spotted owls.

The results of our early barred owl removal work were encouraging, but the sample sizes were still too small and the time interval too short making conclusions preliminary. In response, we were granted the authorization from the FWS and California Department of Fish and Game to fully implement a barred owl removal study within the GDRC spotted owl demography study area in 2010. The authorization allowed for a total of 70 barred owls to be taken over 3 years with a maximum of 30 individuals in any year. To account for geographic variation in habitat and both spotted and bared owl population densities, GDRC's demographic study area was subdivided into three treatment (Salmon Creek, Korbel/Mad River/Little River and Wilson/Hunter/Terwer Tracks) and three control areas (Ryan Creek, Redwood Creek and Bald Hill/County Line Tracts).

One of the valuable findings from these initial removal efforts was that using a lethal method (i.e., adult territorial individuals were attracted to within 20 to 30 m using recorded calls and shot with a 20 gauge shotgun), barred owls could be collected humanely, efficiently and with relatively little effort and cost. For example, in late winter early spring of 2009, one field person made a total of 16 visits to six territories with a mean of 1 hour 23 minutes per visit (not including vehicle travel time) to collect 100 percent of the known (11) territorial barred owls in the Korbel/Mad River treatment area. This equated to approximately 2 field hours per barred owl removed, which has been repeated in 2010 and 2011. By far, the biggest cost associated with the barred owl removal study is the initial surveys of the study area, but this is also needed for the spotted owl monitoring program so the barred owl removal study adds little additional cost above GDRC's existing commitments.

To date, nine historical spotted owl sites meet the criteria for a removal case study (i.e., barred owls removed from a historical spotted owl nest site or activity center). All of the sites have been re-occupied by spotted owls with the time for re-occupation ranging from a minimum of 13 days to a maximum of 1 year. Four of the sites were re-occupied by the original resident spotted owls, including one female that had not been seen for 7 years, and the remaining sites were re-occupied by new or unknown individuals. The spotted owls were again displaced at three sites by re-occupation of barred owls: one the following year, one after 3 years and one after 4years.

The barred owl removal experiment has been too limited in time and scope to assess the impact on spotted owl fecundity and survival rates, but we have been able to assess preliminary changes in spotted owl occupancy rates for a portion of the study area. All the known barred owls (16) were removed from the Korbel/Mad River area in 2009, and in 2010, all of the barred owls (eight individuals) that recolonized this area were removed. The control area (i.e., no barred owl removal) immediately adjacent to the Korbel/Mad River treatment area occurred in the Redwood Creek watershed. Although there has not been enough time to fully investigate differences in occupancy rates between these two regions, we did a preliminary cursory analysis of the trend in the number of occupied spotted owl sites for the Korbel/Mad River treatment versus immediately adjacent Redwood Creek control area. The year 1998 was selected as a reference point, which was the year prior to when barred owl numbers began to rapidly expand in our study area. *Figure 1* below shows the recent trend in occupied spotted owl sites between the control and treatment area.

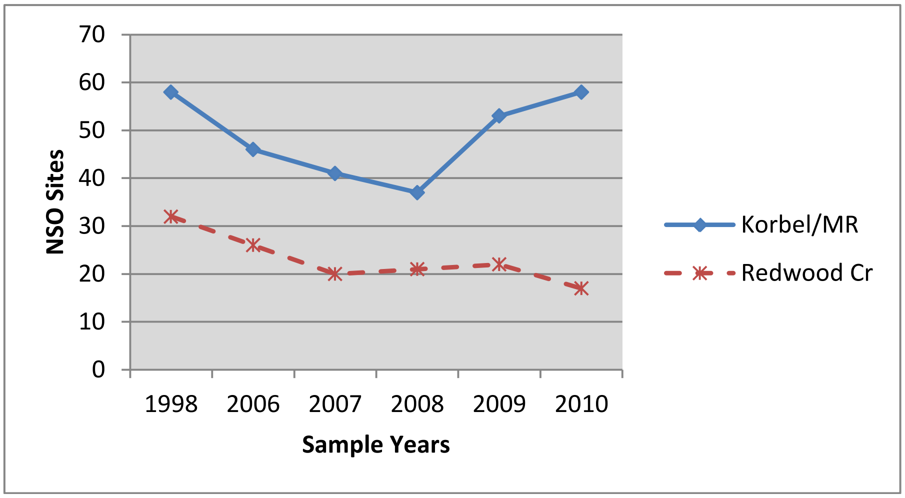

Figure 1—Trend in occupied Northern Spotted Owl (NSO) sites on two adjacent tracts of land on Green Diamond Resource Company's ownership in coastal northern California. Barred owls were removed from "Korbel/MR" in 2009 and 2010, but no barred owls were removed from "Redwood Creek."

The apparent dramatic 43 percent increase in the number of occupied spotted owl sites in the Korbel/Mad River area from 2008 to 2009 was not due exclusively to historical spotted owl sites being "freed" of barred owls, because the number of newly occupied sites (16) greatly exceeded the number of sites (6) from which barred owls were removed. It probably was a combination of removing barred owls from historical spotted owl sites, improving survey techniques and increasing amounts of suitable habitat that led to the increase in occupied spotted owl sites in the treatment area. However, the lack of a similar increase in the Redwood Creek tract suggested that removal of barred owls in the Korbel/Mad River area may have had a synergistic effect on all of these factors. Specifically, creating an area free of barred owls may have increased the probability that resident spotted owls that previously were silent would now be more inclined to respond to surveys and it would make colonization of the area more likely when floater spotted owls were not being rebuffed by resident barred owls. The negative impact of barred owls in some stretches along Redwood Creek that historically had high densities of nesting spotted owls is quite dramatic. The spotted owls appear to have been almost totally replaced by barred owls, although some spotted owls because of their high adult survival are likely to still persist in these areas. However, every attempt to survey for them results in an aggressive response from a barred owl, which precludes any further attempts to locate spotted owls.

When this study is complete, we will also be able to estimate impacts of barred owls on apparent survival and fecundity of spotted owls. Currently, we lack even preliminary estimates on these demographic parameters, but there is little doubt that if the trend in occupancy between treatment and control areas continues, there also will be dramatic differences in apparent survival and fecundity.

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How Do We Know How Many Salmon Returned to Spawn? Implementing the California Coastal Salmonid Monitoring Plan in Mendocino County, California

Sean P. Gallagher[1] and David W. Wright[2]

Abstract

California's coastal salmon and steelhead populations are listed under California and Federal Endangered Species Acts; both require monitoring to provide measures of recovery. Since 2004 the California Department of Fish and Game and NOAA Fisheries have been developing a monitoring plan for California's coastal salmonids (the California Coastal Salmonid Monitoring Plan- CMP). The CMP will monitor the status and trends of salmonids at evolutionarily significant regional scales and provide population level estimates. For the CMP, data to evaluate adult populations are collected using a spatially balanced probabilistic design (e.g., Generalized Random Tesselation Stratified- GRTS). Under this scheme a two-stage approach is used to estimate status. Regional redd surveys (stage 1) are conducted in stream reaches in a GRTS sampling design at a survey level of 15 percent or ≥ 41 reaches, which ever results in fewer reaches, of available habitat each year. Spawner: redd ratios are derived from smaller scale census watersheds (stage 2) where "true" escapement is estimated using capture-recapture methods. These are used to estimate regional escapement from expanded redd counts. In 2008 and 2009 we applied the results of our previous studies to estimate salmonid escapement for the Mendocino coast region, the first implementation of the CMP in the state. Here we present the results of the first 3 years of this monitoring effort and discuss our findings in context of expanding the CMP to all of coastal California. We discuss sample frame development, sample size, and present escapement data for six independent and eight potentially independent populations and two Diversity Strata within the Central California Coho Salmon Evolutionarily Significant Unit.

Key words: coho salmon, population monitoring, spawning surveys, status, trends

Introduction

Recovery of salmon and steelhead listed under the Federal and California Endangered Species Acts primarily depends on increasing the abundance of adults returning to spawn (Good et al. 2005), and monitoring the trend in spawner escapement is the primary measure of recovery. In California watersheds north of Monterey Bay, Chinook (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), and steelhead (*O. mykiss*) are listed species. Delisting will depend on whether important populations have reached abundance thresholds (Spence et al. 2008).

In 2005, the California Department of Fish and Game (CDFG) and NOAA

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Fisheries published an action plan for monitoring California's coastal salmonids (Boydstun and McDonald 2005). This plan outlines a strategy to monitor salmonid populations' status and trends at evolutionarily significant regional spatial scales and provide population level estimates. The monitoring is similar to the adult component of the Oregon Plan, where data to evaluate regional populations' are collected in a spatially explicit rotating panel design. Crawford and Rumsey (2009) and the Salmon Monitoring Advisor (https://salmonmonitoring advisor.org/) recommend a spawner abundance sampling design using a spatially balanced probabilistic approach (e.g., Generalized Random Tessellation Stratified -GRTS, Larsen et al. 2008). Similarly, Adams et al. (2010) propose a two-stage approach to estimate regional escapement. Under this scheme, first stage sampling is comprised of extensive regional spawning surveys to estimate escapement based on redd counts, which are collected in stream reaches selected under a GRTS rotating panel design at a survey level of 10 percent of available habitat each year. Second stage sampling consists of escapement estimates from intensively monitored census streams through either total counts of returning adults or capture-recapture studies. The second stage estimates are considered to represent true adult escapement and are used to calibrate first stage estimates of regional adult abundance by associating precise redd counts with true fish abundance (Adams et. al. 2010).

The Action Plan was tested and further developed in a 3 year pilot study (Gallagher et al. 2010a, 2010b). This study compared abundance estimates derived from a regional GRTS survey design to abundance measured using a more intensive stratified random monitoring approach, evaluated sample size and statistical power for trend detection, and evaluated the quality of the stage two data for calibrating regional surveys. Gallagher et al. (2010a) recommended that annual spawner:redd ratios from intensively monitored watersheds be used to calibrate redd counts for regional monitoring of California's coastal salmonid populations because they were reliable, economical, and less intrusive than tagging, trapping, underwater observation, weirs, and genetics. Converted redd counts were statistically and operationally similar to live fish capture-recapture estimates, but required fewer resources than the other methods they evaluated. Gallagher et al. (2010b) found that redd counts and escapement estimates using annual spawner:redd ratios were reliable for regional monitoring using a 10 percent GRTS sample, and that increasing sample size above 15 percent did not significantly improve the estimates. Their evaluation of sample size suggested that a sample size of ≥ 41 reaches or 15 percent, whichever resulted in fewer reaches, would have adequate precision and sufficient statistical power to detect regional trends in salmon populations.

The 10 percent sample size recommended by Boydstun and McDonald (2005) was provided with little justification. Their Mendocino Coast example 10 percent GRTS sample resulted in an annual sample of 203 reaches. This size sample draw would likely result in costly over sampling of more reaches than necessary to encompass intra-reach variance. NOAA (2007) wrote that the issue of sampling intensity for a Coastal Monitoring Plan (CMP) has not yet been resolved.

Beginning in 2008-09 we applied the results of our previous studies to estimate salmonid escapement for the Mendocino coast region. The study's purpose was to 1) provide spawner: redd ratios for calibrating regional redd surveys and, 2) conduct regional spawning surveys in the Mendocino coast region (*fig. 1*) to estimate escapement and assess sample size at this scale. We present the coho salmon results from the first 3 years of study and discuss our findings in context of the CMP. We discuss sample frame development, sample size, and present escapement data for six independent and eight potentially independent populations and two Diversity Strata (National Marine Fisheries Service 2010) within the Central California Coho Salmon Evolutionarily Significant Unit.

Materials and methods

The three intensively monitored life cycle monitoring streams (LCS) (*fig. 1*) were selected for a variety of reasons. Pudding Creek has a fish ladder where fish can be marked and released and has been operated as a LCS by Campbell Timberlands management since 2006. The South Fork Noyo River has coho salmon data relating to the Noyo Egg Collecting Station, fish can be captured and marked there, and it has been operated as an LCS since 2000. Caspar Creek was chosen because of existing salmon monitoring data. In 2005 we built and operated a floating board resistance weir in Caspar Creek 4.9 km from the Pacific Ocean.

The Mendocino coast region extends from Usal Creek to Schooner Gulch (*fig. 1*). We followed Boydstun and McDonald (2005) to define the sampling universe, to create a sample frame (the sample universe broken into sampling units), and to produce a GRTS draw (the spatially balanced random sample). We defined the sampling universe as all coho spawning habitat in coastal Mendocino County.

We estimated escapement using the Schnabel mark-recapture method (Krebs 1989) and conducted redd censuses in our LCS. We marked and released fish with floy tags and recaptures were live fish observations made during spawning surveys. To estimate redd abundance for calculating spawner: redd ratios we used redd count and measurement data collected during spawning surveys following Gallagher et al. (2007). Over and under-counting errors in redd counts (e.g., bias) were reduced following Gallagher and Gallagher (2005). Surveys were conducted fortnightly from early December to late April each year in all spawning habitat in each LCS.

To estimate regional abundance we conducted biweekly spawning surveys in 41 GRTS reaches from mid-November through April each year. Our methods for redd count and measurement data on spawning surveys were the same as for LCS. We used the average annual coho salmon spawner: redd ratios from our LCS to convert bias corrected redd counts into fish number for each reach (Gallagher et al. 2010a). We followed Adams et al. (2010) to estimate regional abundance where the average number of redds in our 41 reaches was multiplied by the total number of reaches in our sample frame. We estimated 95 percent confidence intervals using the Bootstrap with replacement and 1000 iterations.

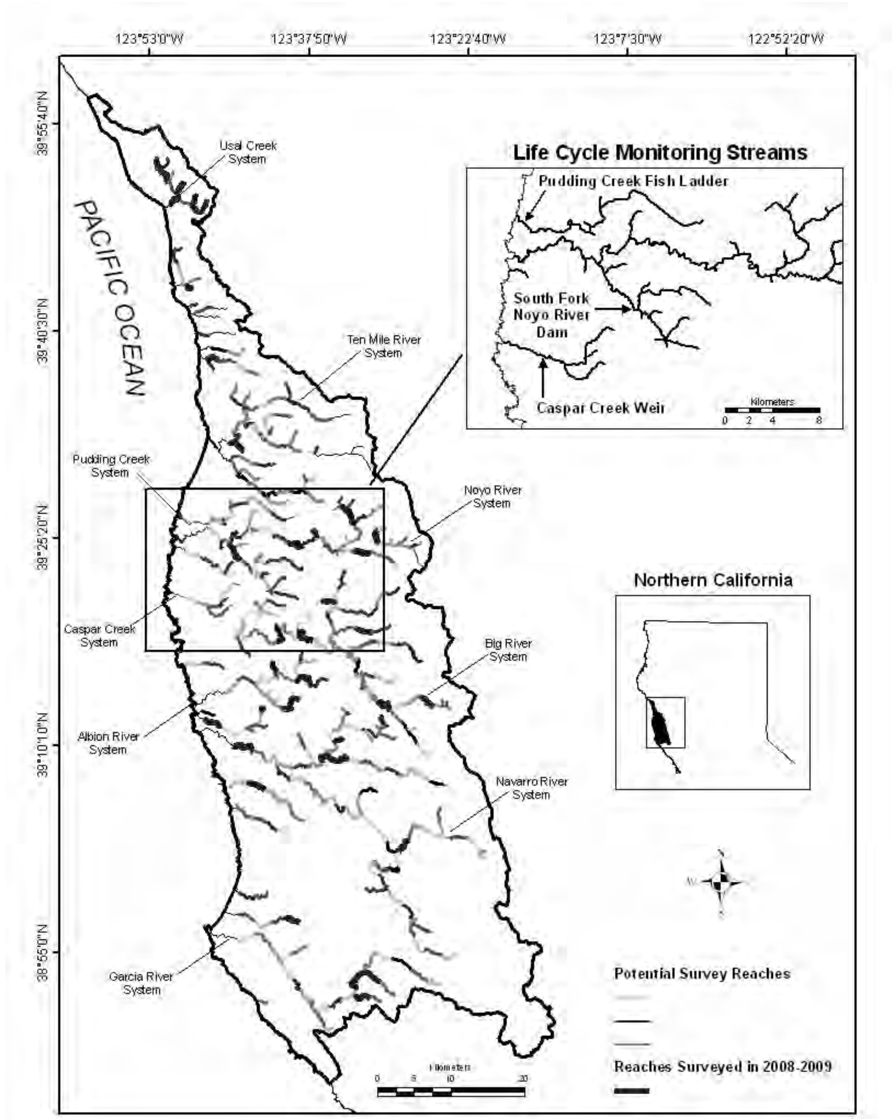

Figure 1—Study area, survey reaches and life cycle monitoring streams in Mendocino County, California.

Results

Each year, nine of the 41 GRTS reaches (21 percent) were unavailable for sampling because landowners denied us permission to enter. These reaches were replaced by the next nine in the list to fill out our required sample size of n = 41 or a 12 percent sample. The GRTS sample resulted in sampling reaches in all independent populations in two coho salmon diversity strata within the CCC ESU.

Each year sampling 41 reaches encompassed the variation in coho salmon redd density within coastal Mendocino County and redd density was not significantly different among streams (*fig. 2*). Because redd density was not statistically different among streams we used the average of all reaches to estimate total redd counts and escapement for the region and for individual populations within the region. We estimated an average of 877 (95 percent CI 377 to 1,515) coho salmon redds and 1,167 (95 percent CI 488-2,068) adult coho salmon in coastal Mendocino County over 3 years (*table 1*). Regional coho salmon confidence limit widths averaged 64 percent with n = 41 and decreased to 47 percent when we included reaches from the LCS's (n =80). Escapement estimates for the two diversity strata and for individual streams had increased confidence limit widths due to smaller sample sizes (*table 1*).

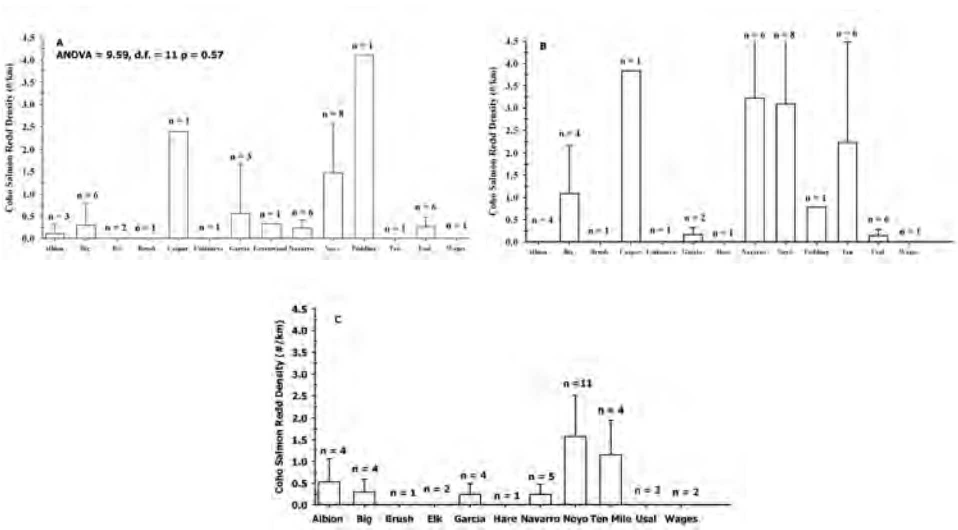

Figure 2—Average coho salmon density by stream for regional surveys in coastal Mendocino County California 2009 to 2011. A. 2009. B. 2010. C. 2011. Numbers above estimates are sample sizes (the number of reaches surveyed). Thin lines are 95 percent confidence limits.

To examine if we could use the regional average redd density to estimate redd abundance for streams we did not survey, we tested LCS redd census and estimates made by multiplying regional average density by LCS stream length with paired t-tests. Coho salmon census redd counts were not significantly different than estimates made by multiplying regional redd density by stream length ($t = 1.079$, $df = 4$, $p = 0.35$, $\alpha = 0.06$).

Confidence limit half widths for our regional sampling were greater than 30 percent (*table1*). From our 2009 and 2010 regional data it appears to attain confidence limits with 30 percent precision and 90 percent certainty following our study design we need to sample 184 reaches (*table 2*). This level of sampling would require sampling more than half of the entire region for coho salmon. Variation around the mean coho salmon redd density peaked at n = 41 and remained constant after n = 58 reaches (*fig. 3*). The coefficient of variation (cv) in coho salmon redd density averaged 221 percent (n = 41) and improved insignificantly with continued sampling (cv = 220 percent, n = 80) over 3 years.

Table 1—*Coho salmon escapement estimates (95 percent confidence limits) for coastal Mendocino County California 2009 to 2011: ns = not surveyed, na = not available, and DS = diversity strata. Precision is the 95 percent confidence limit half widths relative to the mean these data are three year averages.*

Stream	N	Number of Adults			Precision
		2009	2010	2011[a]	%
Mendocino Coast	41	887 (415 to 1545)	898 (555 to 1308)	1575 (534 to 2947)	61
Lost Coast DS	32	672 (295 to 1083)	1059 (515 to 1711)	1318 (328 to 2700)	69
Navarro Point DS	9	158 (41 to 342)	513 (108 to 989	176 (18 to 369)	94
Albion River [e]	3	8 (0 to 22)	0	99 (0 to 297)	148
Big River [e]	6	80 (0 to 210)	134 (20 to 214)	147 (0 to 435)	122
Big Salmon Cr.[b]	2	0	ns	ns	na
Brush Cr.[b]	1	0	0	0	na
Caspar Cr. [c]	6	6	5 (3-9)	30	na
Cottaneva Cr.	1	0	0	ns	na
Garcia River [e]	3	69 (0 to 206)	9 (0 to 18)	65 (13 to 130)	166
Greenwood Cr.[b]	1	9	ns	ns	na
Little River [c]	2	4	2	2	na
Navarro River	6	124 (18 to 124)	452 (159 to 790)	137 (0 to 420)	103
Noyo River	10	294 (82 to 573)	286 (58 to 650)	494 (24 to 583)	79
South Fork Noyo River [c, d]	12	19	63 (42 to 112)	20	na
Pudding Cr. [c]	9	50 (32 to 96)	9 (4 to 27)	na	97
Ten Mile R.[f]	1	0	190 (4 to 454)	295 (0 to 630)	113
Usal Cr.	3	10 (2 to 18)	2 (0 to 5)	7 (0 to 20)	104
Wages Cr. [b]	1	0	0	0	na

[a] Preliminary data.
[b] Only one reach was surveyed in this stream so confidence bounds were not calculated.
[c] Life cycle monitoring station complete census.
[d] Low flows limited the number of fish that passed the weir and spawned above the egg collecting station in 2009.
[e] Four reaches in 2010 and 2011.
[f] Six reaches in 2010 and 2011.

Table 2—*Estimated sample sizes (number of reaches) for five desired levels of precision (width of the 95 percent confidence limits relative to the mean) in coho salmon redd densities for regional monitoring.*

Precision	Confidence limits	
	90%	95%
10%	1635	2370
20%	413	593
30%	184	263
40%	103	148
50%	66	95

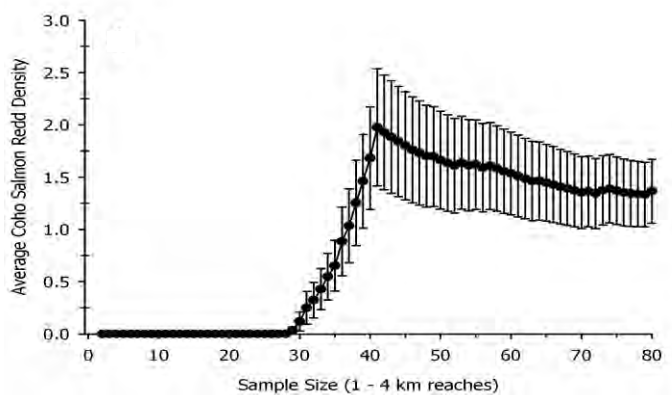

Figure 3—Cumulative mean coho salmon redd density (±SE) plotted against the number of sample reaches surveyed in coastal Mendocino County, California during 2010.

Discussion

Boydstun and McDonald (2005) suggested their example sample frame would need refinement which might reduce the sample frame by 30 to 40 percent. We reduced a list of 2033 stream reaches to 339, an 83 percent reduction by identifying known coho salmon streams (Spence et al. 2008) and using local knowledge to define coho salmon spawning habitat. The sample frame we produced is for Chinook, steelhead, and coho, with species designation for each reach (e.g., soft stratification, Larsen et al. 2008). Soft stratification is simpler and cheaper than having one sample frame for each species because each reach covers multiple species thus reducing logistics and field time.

Adams et al. (2010) suggest a 3, 12, 30 year revisit design based on the life cycles of salmonids present. In 2009 we sampled the first 41 reaches on our GRTS draw. The Action Plan states that 40 percent of the GRTS sample reaches should be assigned as annual samples. During 2010 we sampled reaches 1 to 16 and 42 to 66 and in 2011 we sampled reaches 1 to 16 and 67 to 92. On average 21 percent of selected reaches were not available to sample because landowners denied us permission to enter. All unavailable reaches were on private land were replaced with reaches that were also on private land, reducing this source of bias in our study[3].

For the third consecutive year we produced coho salmon escapement estimates for the entire coast of Mendocino County consisting of two diversity strata within the CCC Coho salmon ESU, six independent populations, and eight potentially independent populations. While the precision of these estimates (95 percent confidence half widths) was lower than expected, we now have estimates, with statistical certainty, of how many salmon escaped in this area. We believe, given the variance in redd density we observed, if we are confident in our regional estimates we can have confidence in individual population estimates despite the large confidence widths.

[3] C. Jordan, NOAA Fisheries, Northwest Fisheries Science Center, personal communication.

In our earlier studies we suggested (Gallagher et al. 2010 b) if redd density variation in the pilot study area was representative of coastal California as a whole, a sample size > 41 reaches for coho salmon should have confidence interval widths of 30 percent and sufficient statistical power for monitoring escapement trends. Our present application of these sample sizes to the entire area of coastal Mendocino County resulted in escapement estimates with larger confidence widths than we expected. We attribute this in large part to low abundance. When we included all reaches surveyed during each year, a systematic rather than design based GRTS sample, precision in our estimates improved. However, the coefficient of variation did not improve with increased sample size and variation about the mean (*fig. 3*) peaked out at n = 41 and did not substantially decrease after about 58 reaches (~15 percent). Redd density (an index of abundance) in LCS was lower between 2009 and 2011 than observed since 2000 and was outside the range of data we used earlier (Gallagher et al. 2010b) to develop sample size estimates. Courbios et al. (2008) found that a larger sampling fraction and higher redd abundance resulted in better accuracy for GRTS. At low redd abundance none of their sampling designs were accurate. In a GRTS sampling design for bull trout in the Columbia Basin, Jacobs et al. (2009) found that accuracy ranged from 15 percent to 35 percent and was dependent on redd distributions within basins and that there was no reduction in accuracy with sample sizes between 10 and 50 sites. Our results are similar in that increased sample size appears to only marginally improve the precision of our estimates.

Crawford and Rumsey (2009) suggest that salmon monitoring programs strive for estimates that have a coefficient of variation (CV) of ± 15 percent. Our regional CVs for coho salmon averaged 221 percent (n = 41) to 220 percent (n = 80) and increased sample size did not substantially improve them. Given the cost to survey one reach for a season ($3,000/ reach, Gallagher et al. 2010b) and the fact that increasing our sampling fraction to 30 percent would result in sampling 184 reaches ($552,000/year), which appears would not greatly improve precision, we recommend continued evaluation of smaller sampling fractions. The use of standardized data collection procedures and trained staff (Gallagher et al. 2007) will continue to contribute to increased precision in regional escapement monitoring. Finally, for regional monitoring at low abundance, managers may have to accept larger uncertainties in escapement estimates. However, management for recovery primarily means listing decisions, and a delisting decision will likely be based on data from sustained higher abundance levels when both precision and accuracy levels would be much improved.

Acknowledgments

This work was funded by the California Department of Fish and Game's Fisheries Restoration Grant Program. Too many individuals to mention by name from the following entities helped with this study: CDFG, Campbell Timberlands Management, NOAA Fisheries Santa Cruz, and the Pacific States Marine Fisheries Commission. Be assured we value your help. A few we are compelled to mention by name include: Stan Allen for administrative support, Wendy Holloway for helping to craft *fig. 1*, and Shaun Thompson, Wendy Holloway, Chris Hannon, and Scott Harris for many long hours in the field. Thanks to Sean Hayes and Brad Valentine for critical review that greatly improved earlier versions of this manuscript.

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Ecology and Management of *Martes* on Private Timberlands in North Coastal California

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Abstract
Green Diamond Resource Company has conducted periodic studies of fishers on its California timberlands since 1994. A graduate study in 1994 and 1995 used track plates to investigate the distribution and habitat associations of fishers. Fishers were detected at 65 percent of the survey segments during both years combined but marten were not detected. A repeated track plate survey conducted a decade after the initial effort revealed the presence of marten west of a known population on public lands. An abbreviated track plate survey effort in 2006 again confirmed the presence of marten in the same area of the ownership and a potential expansion of the occupied area. During 2010 and 2011, we used remote cameras to survey 75 stations in a 2 km^2 hexagonal grid in the northern portion of the ownership. We detected fisher at 45 camera stations, marten at eight stations and both species at six stations. Marten are persisting in areas where we initially detected them in 2004, but additional studies are necessary to investigate habitat use, demographics, dispersal and interactions with competitors like the fisher.

Key words: camera, fisher, managed forest, marten, non-invasive survey, track plate

Introduction

The marten (*Martes americana*) and fisher (*Martes pennanti*) are coexistent mesocarnivores inhabiting the coastal forests of northern California. Within north coastal California, the Humboldt subspecies of marten (*M. a. humboldtensis*) historically occurred in the coast redwood zone from the Oregon border south to northernmost Sonoma County. However, since 1995, surveys conducted in this region suggest that martens no longer occupy most of their historical range (Slauson 2003, Zielinski et al. 2001). Currently, martens are known from only one small population in southern Del Norte and northern Humboldt Counties, which comprise less than 5 percent of its historical range in this part of the state (Slauson 2003). Historical records suggest that martens in northwestern California were closely tied to late-successional coast redwood (*Sequoia sempervirens*) forests (Slauson and Zielinski 2003); however, the one remnant population in this region occurs in an area dominated by Douglas-fir (*Pseudotsuga menziesii*) and tanoak (*Lithocarpus densiflorus*) forest associations (Slauson et al. 2007), with coast redwood associations limited to the western edge of the currently occupied range (Slauson et al. 2007). This population uses two structurally distinct forest types with a common link of

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dense shrub cover; one occurring on serpentine soils and one on more productive non-serpentine soils (Slauson 2003, Slauson and Zielinski 2007, Slauson et al. 2007).

In California, fishers historically ranged from Marin County to the Oregon border, east to Mount Shasta and Lassen Peak and south through the Sierra Nevada to northern Kern County (Grinnell et al. 1937). Fishers currently exist in two disjunct populations in California: one on the west slope of the southern Sierra Nevada and the other in the Klamath Mountains and Coast Ranges of northwestern California (Zielinski et al. 1995). There has long been a concern over habitat loss and alteration due to rapid harvest of mature and old-growth Douglas-fir forests (Rosenberg and Raphael 1986).

Both species have been petitioned for listing under state or federal endangered species acts. The United States Fish and Wildlife Service determined that the west coast population of fisher warranted protection under the Endangered Species Act, but were precluded from being added to the list of threatened or endangered species due to higher priority actions (USFWS 2004). The fisher was officially added to the candidate species list at the time of this ruling. The California Department of Fish and Game recently completed a comprehensive status review of fisher in response to a listing petition (CDFG 2010), but the species was denied protection under the state endangered species act. The Humboldt marten was recently petitioned for federal listing as a threatened or endangered species (Center for Biological Diversity and Environmental Protection Information Center 2010).

Green Diamond Resource Company (GDRCo) has conducted periodic studies focused on fishers inhabiting its California timberlands since 1994. These track plate surveys were suitable for detecting martens, and surveys conducted a decade after the initial effort revealed the presence of marten several km west of the known population on public lands. An impromptu survey effort in 2006 to investigate a potential regional decline in fisher numbers again confirmed the presence of marten in the same area and a potential expansion of the occupied area. The detection of a marten in Prairie Creek Redwoods State Park in 2009 (Slauson and Holden 2009) and about 11 km west of the detections on GDRCo ownership prompted additional surveys on select areas of GDRCo lands in 2010 and 2011. This paper summarizes results of non-invasive surveys for *Martes* conducted on GDRCo's managed forests since 1994.

Study area

We conducted surveys in Humboldt and Del Norte Counties, California on approximately 150,000 ha of timberlands owned primarily by GDRCo. Coast redwood forest dominated the coastal areas and lower elevations. Douglas-fir replaced redwood as elevation and distance from the coast increased. Hardwoods existed as pure stands or common forest components. Red alder (*Alnus rubra*) and maple (*Acer macrophyllum*) dominated in coastal mesic sites, while tanoak, madrone (*Arbutus menziesii*) and giant chinquapin (*Chrysolepis chrysophylla*) occurred at higher, more xeric sites. The entire study area was within 30 km of the Pacific Ocean, and elevation on the study area ranged from 5 to 1400 m. The maritime climate, which produces fog or low clouds throughout the year, has definite wet and dry seasons (NOAA 2001) with mean summer and winter temperatures of 8 and 15 ^{o}C respectively (Zinke 1988). Annual precipitation occurs mostly as rain from

November through April and varied from 95 cm near the coast to 150 cm in the coastal mountains (NOAA 2001). Substantial and persistent snow occurred at elevations above 1000 m. Most of the study area had been subjected to timber harvest over the past century and consisted of second and third-growth forests, ranging in age from recently harvested to 120 years.

Methods

We conducted surveys for mesocarnivores using enclosed track plates (Fowler and Golightly 1994, Ray and Zielinski 2008) in 1994, 1995, 2004, and 2006 and remote cameras from 2010 to 2011. We placed a maximum of six track plates (stations) in suitable habitat at approximately 1 km intervals within 5 km linear segments (Fowler and Golightly 1994). We baited each track plate with a piece of chicken. We re-baited and replaced sooted plates every other day for 22 consecutive days. We surveyed during January to May, but a subset of segments was surveyed during June to August to assess seasonal variation. We calculated detection ratios based on segments by dividing the number of segments with >one detection by the number of segments sampled. These detection ratios were used to assess trends in detection ratios (i.e., index of population size or density) over time and by region.

Prior to 2010, no surveys were conducted with the specific objective of determining the distribution or habitat associations of marten on GDRCo's ownership. However, our field techniques were equally suitable for martens. In 2010, we deployed remote cameras at stations centered on a 2 km^2 hexagonal grid randomly located on the ownership (Zielinski et al. 2007). We randomly selected units to sample, but also focused on areas where marten were detected during prior track plate surveys. We deployed cameras for a minimum of 3 weeks at each station. We baited stations with raw chicken and a commercial trapping lure (Caven's Gusto Lure, Minnesota Trapline Products, Pennock MN) as an attractant. We baited and checked stations weekly with the general exception that some stations were baited and checked every other week due to complications with access due to weather or other demands on field personnel. During the pilot work in 2010, we tested several models of purchased and loaned cameras to assess reliability, ease of use, function and other important factors. In 2011, we began placing two RECONYX (RECONYX, Inc., Holmen, WI) cameras (models HC500 and PC800) at each sample unit. Cameras were located approximately 3 to 5 m and at right angles from the bait tree.

Results

During all surveys in 1994, 1995, 2004 and 2006 we operated a total of 975 individual track plate stations. We obtained 101 and 135 fisher detections in 1994 and 1995 respectively on 26 of the 40 (65 percent) survey segments during both years combined. We obtained a total of 78 fisher detections at 20 of the 40 segments in 2004. In 2006, we obtained 115 fisher detections at 12 of 18 segments surveyed. Mean detection ratios during the four sample periods from 1994 to 2006 varied from 0.40 to 0.67 at the segment level. The mean latency to detection (LTD) for all surveys was 11.6 days. The distributional pattern of detections across the study area indicated that almost all survey segments in the more interior Douglas-fir and mixed redwood-Douglas-fir regions had detections. We detected fewer fishers in the more coastal redwood areas, southern regions near Humboldt Bay and the Eel River drainage and the northern region near the Oregon-California border.

We did not detect marten during the 1994 or 1995 track plate surveys. The repeat track plate survey conducted in 2004 yielded a total of six marten detections at four track plates on two segments in the Klamath region of the ownership. To confirm that the tracks recorded on the sooted track plates were indeed marten, we put a camera trap at select locations and obtained photographic evidence of marten. With the repeat of the track plate survey in 2006, we obtained a total of 13 marten detections at nine track plates on three segments. We also confirmed marten presence in 2006 with a camera trap.

We sampled 75 2 km^2 units with cameras between September 2010 and June 2011. We obtained photographic evidence of fisher at 45 stations (60 percent) and marten at eight stations (10.6 percent). Seven out of eight stations had multiple visits by marten. Fishers were detected at 75 percent of the stations with marten. The mean LTD for marten was 11 days (minimum = 5, maximum = 16 days).

Discussion

Our surveys indicate that fishers were relatively abundant and well distributed throughout the majority of the ownership during the periods of study, suggesting current forest management strategies are compatible with fishers habitat needs. However, Matthews and others (2011) found that indices alone were not adequate for detecting significant changes in a fisher population. Another study over a similar but larger geographic region including Redwood State and National Park and Humboldt Redwood State Park also indicated that fishers were generally less frequently detected in areas closer to the coast (Beyer and Golightly 1996). In addition, this broader survey also showed a pattern with few fishers detected in the northernmost (Smith River watershed) and southernmost (Eel River watershed) portions of the study area. A survey of Redwood National and State Parks provided corroboration relative to fisher habitat use in the redwood region (Slauson et al. 2003). An analysis of track plate surveys throughout old growth and second growth portions of the park indicated that fishers were found more than expected in second growth and less than expected in old growth. However, this study also found that fishers were associated with structurally complex portions of the second growth stands.

Our current knowledge of the marten on the study area is very limited. The lack of detections of martens during most surveys does not prove that they were absent, but the preponderance of negative data certainly indicated that martens were either very rare or absent over most of the study area. With the overall dearth of marten detections, it is obvious that we lacked the data to assess any trends in the marten population, and we are not able to draw any valid conclusions about habitat use. Presumably, the martens detected on GDRCo’s ownership in 2004 and 2006 were dispersers or peripheral residents from the core population to the east on public lands. It may be strictly coincidental, but the potential expansion of martens onto our study area occurred at the same time that the regional population of fishers was temporarily declining. Our surveys in 2004 indicated a potential decline in fisher abundance as evidenced by fewer detections relative to the other survey years. Matthews et al. (2011) documented a decline in relative abundance of fishers from 1998 to 2005 on the neighboring Hoopa Valley Indian Reservation. Although we detected martens at track plates where fishers had been previously detected, no martens and fishers were detected at the same track plate during the same survey period in 2004. Fisher and marten were detected at only one track plate during the same survey period in 2006. While our survey results do not allow for a definitive assessment of the trend in the coastal marten population, it appeared as if martens were dispersing further from their core population during the mid-2000s relative to the 1990s. In addition, a marten was detected approximately 10 km further to the west in Prairie Creek Redwoods State Park in 2009. Regardless of any potential localized expansion of martens within GDRCo’s ownership, it remains clear that the marten population was small and isolated to only a small portion of the ownership.

Our recent camera survey data suggests that fishers are relatively abundant within the Klamath region of the study and that they also overlapped the marten detections more than we observed during the prior track plate surveys. The negative interaction between fishers and martens has been noted in numerous locations throughout their range and the inverse relationship in abundance between the two species may be the result of this interspecific interaction. Although our sample size of marten detections was small (eight), we observed fishers at 75 percent of the cameras where marten were detected. We observed one bobcat (*Lynx rufus*) at a station where we also detected marten and fisher. However, our surveys were not designed to attract bobcats and likely do not accurately represent the abundance of bobcats present within the managed landscape. Forest management activities that encourage growth of other mesocarnivore populations may also be considered a threat to marten populations, as some of these species (e.g., fisher and bobcat) may opportunistically kill martens when encountered (Hamlin et al. 2010). Loss, modification, and fragmentation of habitat are considered significant ongoing threats to the remaining population of martens in northwestern California (Hamlin et al. 2010). Past timber harvest activities have eliminated much of the late-seral forests in coastal northern California, and due to the specialized habitat requirements of martens, such as large diameter live trees, snags, and logs, it would likely require decades of little or no timber harvest in large areas to develop habitat with the necessary structural characteristics to support martens (Hamlin et al. 2010).

Slauson et al. (2003) indicated that the marten should be considered a ‘highly imperiled taxon’ within the redwood region and that Redwood National and State Parks (RNSP) contains both the largest remnant patches and the largest total amount of old growth redwood forests. They concluded that the long-term persistence of martens within the coastal forests of northwestern California will likely require both the maintenance of areas currently occupied by martens and the expansion of their distribution. They further noted that expansion of the marten’s distribution would require the combined effects of the restoration of suitable habitat and functional landscape connectivity to enable recolonization of suitable, but currently unoccupied habitat. However, they went on to note the critical conservation dilemma that the restoration of forest habitats with the structural characteristics necessary to be suitable for martens may take decades while the small isolated extant population of coastal martens was at serious risk of losing genetic variation or elimination due to stochastic demographic or environmental events such as wildfire. This led Slauson et al. (2003) to conclude that while restoration efforts within the coastal parks may eventually facilitate recolonization by martens, other conservation measures may be necessary to re-establish martens due to significant barriers created by large areas of primarily early-seral forest on private lands, the Klamath River and U.S. Highway 101.

Another proactive conservation option that could yield more immediate benefits to the marten population is human assisted dispersal. This approach would be similar to a reintroduction with the exception that limited evidence suggests that marten are currently capable of dispersing to the coastal parks. The marten observed in Prairie Creek Redwoods State Park in 2009 is presumably evidence of a naturally dispersing animal. However, natural dispersal of male and female martens is impeded by considerable ecological and anthropogenic barriers between the existing population and available habitat in the coastal parks. These barriers and the abundant mesocarnivore population on managed lands could constrain natural establishment of a population of martens in the coastal parks. If the martens we detected on private managed forests are reproducing, it may be possible to identify dispersers from this area and deliberately move them westward to suitable habitat on the coastal parks. This strategy is not without complexities that need to be thoroughly explored prior to implementation. For example, this project would likely involve multiple partners from federal, state, tribal and private organizations, a commitment to funding for implementation and monitoring, and some initial basic research such as investigations of dispersal of marten. It would also be prudent to evaluate species reintroduction guidelines in the context of this specific effort (IUCN 1998). Ideally, this collaborative action would increase survival of juveniles relative to that of natural dispersers, it would establish a viable population of martens within coast redwood habitat in the parks, and it could reduce the vulnerability of the population to stochastic events such as disease and wildfire.

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A Permeability Study on Salmonid Spawning Areas in Northern Humboldt County, California[1]

Claire Knopf[2]

Abstract

This research was conducted to determine if local salmonid, specifically coho salmon (*Oncorhynchus kisutch*), Chinook salmon (*O. tshawytscha)*, and the steelhead trout (*O. mykiss)*, redd location is dependent upon the permeability of the streambed. The study took place at four study sites in two coastal drainages in northern Humboldt County, California over a 16 month period (June 2004 to October 2005). This period covered pre and post spawning conditions, as well as changes in seasonal flow conditions. Sampling grids were established at each site to allow for repeatable permeability measurements which ranged in number from 37 to 105. Permeability measurements were taken using a battery operated pump to draw water through a perforated standpipe driven into the streambed at a depth of approximately 25 cm. Study results showed that permeability was not significant in predicting redd location. In addition, results showed that permeability measurements varied in orders of magnitude within as little as 30 cm. Graphic representations of the spawning areas were created using GIS and clearly display the heterogeneity of the streambed. The results of this study support the need for further investigation of spawning gravel assessment methods which can account for extreme spatial variation while minimizing disturbance to the streambed.

Key words: fine sediment, monitoring, permeability, redd, salmon, spawning habitat

Introduction

Increased fine sediment levels in rivers and streams are a concern for many land managers due to possible detrimental effects on salmonid habitat. The need to quantify sediment levels in watercourses serves as a driving force of research and monitoring for both private and public resource managers in California's north coast region. While all life stages of salmonids benefit from cool clean water, embryos and alevins require adequate flow of oxygenated water through the spawning gravel. Permeability, defined as the rate of flow through the substrate, is one parameter that has been used as an indicator of spawning gravel quality.

Excess fine sediment can fill the spaces between gravel in the streambed, or bury it entirely, thereby decreasing permeability, impeding the delivery of oxygen, and reducing the removal of waste from the redd (Cordone and Kelley 1961). Chapman (1988) summarized prior studies showing that the survival of salmonid embryos is positively correlated to permeability ($r^2 = 0.85$). Coble (1961) found that there was no relationship between permeability and survival, but rather a relationship between apparent velocity and survival. The difference is that permeability describes the physical capacity of the substrate to transmit water whereas apparent velocity

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[1] This article has been adapted from the thesis entitled **A permeability study on salmonid spawning riffles in the Little River drainage and Cañon Creek in northern Humboldt County, California,** by C.M. Knopf ; presented December 2010, Humboldt State University, Arcata, California.

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describes the rate of flow of water through a unit area of substrate. While literature shows that redd placement is highly correlated with depth, velocity and bed material (Bjornn and Reiser 1991), there is continued research investigating the complexity of spawning site selection. Ongoing research provides reasons such as stream reach sinuosity, population densities and behavior as to why certain areas are more heavily utilized for spawning.

Sampling methods to determine particle size distribution, such as bulk sampling (McNeil and Ahnell 1960), or freeze coring (Walkotten 1976), are frequently used to quantify substrate composition; however, both methods are destructive to the streambed and are more labor intensive than permeability measurements. Additionally, while bulk sampling measurements can adequately determine the particle size distribution, they do not provide a clear picture of how those particles are arranged in the streambed itself.

The objectives of this study were to determine if streambed permeability explains why certain local salmonids, specifically coho salmon (*Oncorhynchus kisutch*), Chinook salmon (*O. tshawytscha),* and steelhead (*O. mykiss*) create redds in specific locations in the streambed and to create graphic representations which depict the spatial variation of permeability within a spawning area. Permeability measurements for this study were conducted at sites where historic spawning activity has been documented. The hypothesis was that the redd locations and areas of high permeability would be positively correlated.

Methods

Study area

Streambed permeability measurements were conducted at four study sites between June 2004 and October 2005. Study sites were located in two major river drainages, the Mad River and Little River, in northern Humboldt County (*fig. 1*). All study sites were on Green Diamond Resource Company property within commercially managed forests of coast redwood (*Sequioa sempervirens*) and Douglas fir (*Pseudotsuga menziesii*). The estimated total annual precipitation (Rantz 1972) is 165 cm for the Cañon Creek (CC) site in the Mad River drainage and 178 cm for the three Little River sites: mainstem Little River (MLR), Upper South Fork Little River (USFLR), and Lower South Fork Little River (LSFLR). During the study period (June 2004 to October 2005), the rainfall totals from the nearest rain gauges to each of the study sites indicate that the study period had slightly lower than average annual precipitation.

The four sample areas were selected in areas known to have high spawning activity based on results of historical spawning surveys. These watersheds are all known to have spawning coho, Chinook, and steelhead. These sites were also chosen because of similar geology, gradient and substrate. All four sites are low gradient, depositional reaches located within watersheds dominated by formations that consist largely of sandstone.

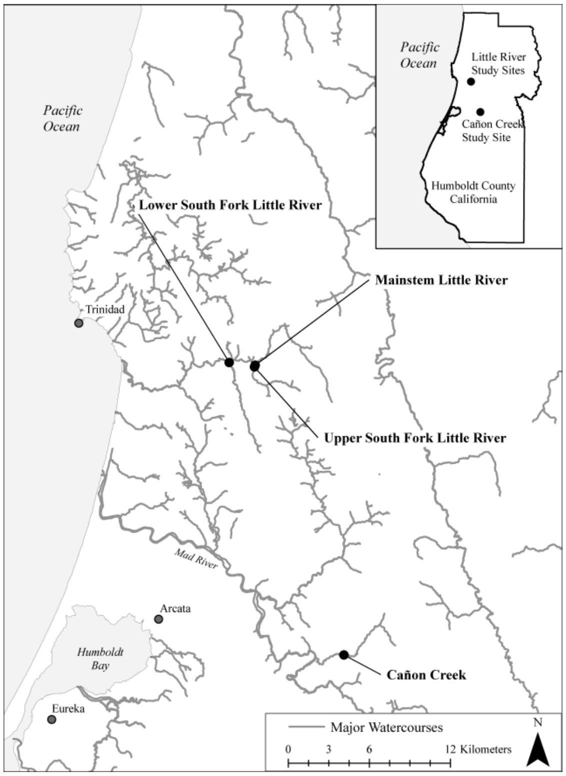

Figure 1—Locations of study sites within Humboldt County, California.

Study design

The method used to measure permeability in this study was based on those developed by Barnard and McBain (1994) and the Mark VI permeability standpipe (Terhune 1958). The sample design varied from the original protocol (McBain and Trush 2000) by greatly increasing the number of points sampled. Rather than using an average of five points at the riffle crest to determine the permeability at a site, a 1.2 m grid was established across the pool tail/riffle crest area. The number of grid points sampled ranged from 37 to 105 depending on the wetted width and total length of the riffle at each site and sampling period in which it was sampled.

Pre-spawning permeability measurements were recorded during summer 2004 at multiple stations throughout the grid, within the wetted width. Each permeability measurement location was marked with a survey flag. Streambed elevation, data point locations, and the wetted perimeters of the study areas were surveyed using a Leica TC307 total station.

Spawning surveys were conducted during the fall/winter 2004 to 2005 run of steelhead, Chinook, and coho. Redd locations were marked with bricks and flags and later surveyed with the total station. Post-spawning permeability measurements were made at grid points and redd locations during early summer 2005. A second post-spawning season measurement of permeability at grid points and redd locations was made in October 2005 at two of the four study sites: MLR and USFLR. These measurements had to occur prior to the onset of rain since increased flows could potentially mobilize deposited sediments. This second measurement (October 2005) was to provide a comparison of seasonal variation at a site and to allow for repeatable measurements in exact locations. In addition, 13 grid points per site were selected at random and designated to have a 30 cm offset permeability measurement point during the October 2005 survey. These offsets were measured in an attempt to determine if permeability varies within very short distances across the study area.

Data analysis

Due to the number of near zero permeability values, a natural log transformation of the data was used to attempt to create a normally distributed data set. Even with the transformation, the data failed normality tests. Logistic regression was used to determine if presence or absence of a redd could be predicted based on the measured variables. The variables used in the regression models were permeability (cm/hr), distance from the riffle crest (m), and distance from the thalweg (m).

Spatial analysis of the data was performed using ArcGIS Spatial Analyst to interpolate permeability values between measured points at each of the study sites. This analysis was done using an inverse distance weighted method (IDW). One analysis used the transformed permeability values, and another used ranked categories. Permeability ranks were used to relate the data back to its biological significance: positive correlation between permeability and survival to emergence (Chapman 1988). The ranks represent three categories, defined as Low (L) (<1,000 cm/hr), Medium (M) (1,000 -10,000 cm/hr) and High (H) (>10,000 cm/hr). Spatial patterns were analyzed to determine if salmonids select areas of high permeability to create redds.

The analysis also investigated the within site and between site variation, the difference between years, seasonal differences, and offset differences. T-tests were used to determine if there were significant differences in permeability between pre and post spawning activity, early and late season measurements and original sampling locations compared to their corresponding offset points.

Results

The mapped interpolated surface (*fig. 2*), shows the location of permeability measurements with their corresponding rates (cm/hr) and the location of redds observed between January and May 2005. The total number of reds marked at each site was (7) CC, (1) LSFLR, (14) MLR, and (9) USFLR. The placement of the redd markers on the map are for comparison only as they were not present during the pre-spawning sampling period.

Logistic regression models were applied to the data set in order to determine if redd location could be predicted based on prior season permeability, distance from the thalweg or distance from the riffle crest. The models indicated that permeability, distance from the riffle crest and distance from the thalweg were not significant predictors in redd placement location. LSLFR was excluded from the models because of the occurrence of only one redd and the presence of an area of very high permeability created by a half-buried log.

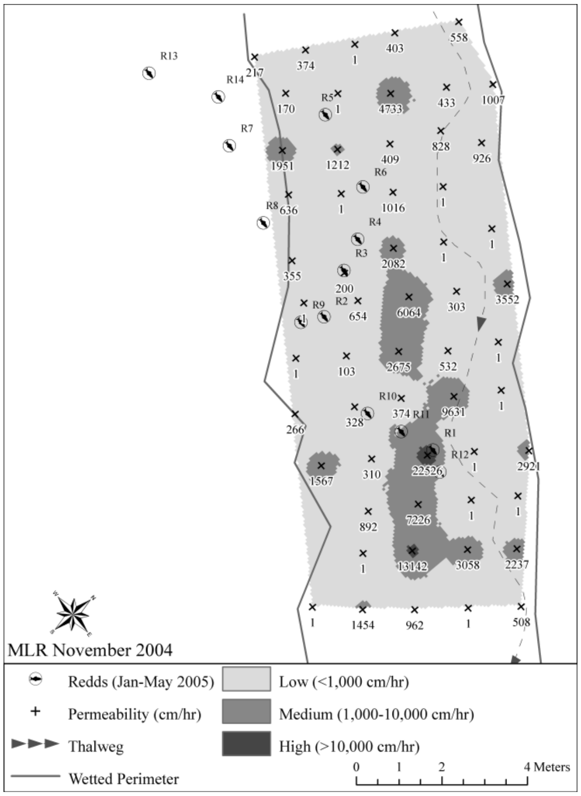

Figure 2—IDW interpolation of pre-spawning substrate permeability at Mainstem Little River study site showing sampling locations and permeability rates in cm/hr.

The models were initially run with the exclusion of low permeability values (<1,000 cm/hr) in an effort to create a normal distribution of the data. This was based on the assumption that fish would not spawn in areas of low permeability due to the limited success of their redds in these areas. Upon further analysis of the data, it was shown that 61 percent of all redds among all sites were built in areas of low permeability; hence it did not seem valid or justified to remove the low permeability values from the analysis.

Although none of the measured variables proved to be significant predictors of redd location, models were developed to provide insight to weak trends in the data. Multiple versions of the logistic regression model indicated that distance from the riffle crest was the most significant factor in predicting the presence of a redd and provided a better result than permeability alone. The prediction model showed that, for this study, the probability of a redd occurrence is more likely as distance from the riffle crest increases. The best fit model, which used categories of permeability, still only explained 52 percent of the variation of the data. The logistic regression models indicate almost no relationship between likelihood of redd presence and permeability.

Spatial analysis of the study sites showed that all three sites were dominated by areas of low permeability prior to the spawning season. There appeared to be no statistically significant change to the site permeability after spawning, or as the flows continued to drop into October. The relationship between the available area of each previously defined category (L, M, H) of permeability and the number of redds present within that category are highly correlated. This distribution further demonstrates the lack of correlation between areas of high permeability and occurrence of redds.

The data were then analyzed to investigate if the fish were able to create their own permeable areas that would persist into the following sampling period. A paired t-test was performed to determine if a change in permeability could be detected between pre-spawning activity and post-spawning activity at individual redd locations. Although the data do show a small difference in the means before and after spawning, there was not a statistically significant difference between pre-spawning and post-spawning permeability ($p = 0.70$, $\alpha = 0.05$). While pre- vs. post-spawning did not show a clear trend, the comparison of early to late summer median values did suggest that permeability decreased between May 2005 and October 2005, though not statistically significant. The difference in permeability values between the original point and the 30 cm offset point measured at MLR and USFLR sites in October differed by category (L, M, H) approximately 50 percent of the time. While the statistical analysis did not detect a difference in the median values, it is clear from the data that there was variation in permeability values within 30 cm.

Discussion

This study used the following assumptions based on literature and established protocols. The first assumption was that the spatial heterogeneity of the gravel in the study sites was low enough that a 1.2 m grid was sufficient to capture the variation in permeability. Another was that the measurements were independent of flow and season. It was evident from the results of this study that the timing of data collection has significant effects on both the ability to actually collect the data and its biological significance. This study also demonstrated that if permeability sampling occurs during times of low flow conditions, then critical spawning locations may be excluded from the data set. The permeability testing for this study was not done while the redds were active to avoid harming incubating eggs. The redd locations were marked as soon as possible and sampled after the fry emerged from the gravel. Because of this, it was impossible to know the conditions while the eggs were buried in the substrate. The monitoring efforts of many governmental agencies and private landowners occur during summer months when weather and work schedules allow. While it is important to monitor organisms in natural conditions, it is difficult to weigh the importance of data collection and the possible impacts that certain methods and timing can have on the organism being studied. It is also important to consider the timing and methods of data collection that will provide scientifically sound and biologically significant data.

In addition to timing or seasonality of data collection, the nature of spawning adds greater uncertainty in analysis of the data. Redd locations were surveyed as a single point, and therefore there are some problems in interpretation of the actual influence of the permeability on redd location. The point at which the redd was marked and surveyed is at best an estimation of the actual location of egg burial. The area of influence of fish activity on or around a redd varies depending on both fish and substrate size (Bjornn and Reiser 1991). Based on estimated measurements from this study, the average area of fish disturbance of the substrate at a redd was approximately 0.88 square meters, but a summary of studies show that redds can average up to 10 square meters (Bjornn and Reiser 1991). Salmonids have the ability to clean the gravel by removing the fine particles while digging their redds. Chinook and steelhead have the ability to reduce the amount of fines within a spawning area by a range of 7 to 23 percent (Everest et al. 1987). It is unknown if the sites that were marked as redds were actually used for spawning, as fish were not present when the redds were marked. Fish can dig “test redds” and can also superimpose redds. While every effort was made to mark each and every redd location, there may have been some that were missed, or were not actual redds, but rather a test redd. Every location marked was assumed to be a single true redd. The sites were visited on a regular basis and redds were marked as soon as they were visible. Due to low visibility in the water during higher flows and the necessity to not disturb the fish during active spawning, the redds were marked as soon as it was feasible. In future studies, redd measurements such as total surface area, pot dimensions, and tailspill dimensions should be observed and recorded.

There were questions that arose with the method and predetermined protocol as the field investigation continued. One issue was the depth at which the measurements were taken. In some cases, the depth that the permeability measurement was made may not necessarily be the depth at which the eggs are buried. The depth in the protocol comes from an average depth of salmonid species; therefore, due to differing geology of the underlying sediments, particle size of the streambed and fish size, the depth of measurement may not accurately represent the permeability of the egg pocket.

The two study sites, MLR and USFLR, in which 30 cm offsets and late summer measurements were taken showed a decrease in permeability between early and late season measurements. The lower permeability in late summer may be attributed to infiltration of fines and organic matter as both surface flow and hyporheic flow decrease. The data also indicated that large variations of permeability can occur within a short distance. Coble (1961) found that permeability values varied at sites only 15 cm apart. Based on results from Kondolf et al. (2008) the zone of influence for each standpipe was less than a 20 cm radius. Because of this variation, and lack of potential spatial dependence, IDW may not have been the appropriate approach to analyze these data, though it appeared to be the best available method, and did provide insight into the complex distribution of the streambed matrix.

Another issue encountered with the data is determining the source of variability, as shown in the permeability maps. The streambed is a very dynamic system and is constantly undergoing change. The spatial variation could be due to hyporheic flow, bed movement, surface flow, fish disturbance, or disturbance due to the sampling method. The variation in fine sediment in streambeds is so large that distinguishing between background and land-use augmented amounts is difficult (Adams and Beschta 1980). It is also hard to determine if the variation is an artifact of the method itself. Kondolf et al. (2008), using the same method, found that up to 68 percent of water that entered the standpipe may actually be surface water leakage along the edge of the pipe. They found that this was common in course gravel installations. Installation of the standpipe into the streambed often opened up pockets free of material allowing for excess water to move along the edge of the pipe.

Conclusion

The results of this study suggest that redd location is not correlated with prior season substrate permeability as determined from a 1.2 m grid. The variation shown within study sites suggests that results are dependent upon where the standpipe is placed within a study area. The large variation suggests that many more sample points would be required to account for, or at least minimize, the variation between measurements. If more sample points are required, other issues arise. The measurements would no longer be independent due to influence of one point on another in close proximity. More sample points would also negate the major benefit of this method in that it is assumed to be a rapid sampling technique. If a large number of samples would be required in order to address variation, then the method is no longer a rapid and inexpensive approach.

This study provides graphic representations of the heterogeneity of the streambed and indicates that redd location is not correlated with permeability. Regardless of the imperfections of the method, study results show the recorded permeability measurements indicate great variation in subsurface conditions. While monitoring permeability rates can describe general bed conditions in specific locations, individual or even average permeability values from individual sites should not be used to characterize spawning gravel conditions for watersheds or even smaller stream reaches.

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Mesocarnivores as Focal Species for the Restoration of Post-Logging Second Growth in the Northern Redwoods

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Abstract

The management of second growth forests to accelerate the restoration of late-successional and old growth characteristics will be one of the greatest challenges for conservation in the redwood region over the next century. In the redwood region, the largest complex of protected areas exists in the north, however >50 percent of these forest reserves are composed of logged, degraded second growth forests. Strategic restoration actions have the potential to accelerate the restoration of old growth forest characteristics and the old growth forest species assemblage that requires these features. Restoration actions in degraded aquatic habitats over the last two decades have been guided by the needs of several salmonid species. Currently there are no guidelines for how to strategically restore second growth forests based on the needs of old growth dependent wildlife species. We developed a focal species approach to provide restoration guidelines based on the spatial and compositional needs of the Humboldt marten (*Martes americana humboldtensis*), a mesocarnivore sensitive to the loss and modification of old growth forest conditions. In addition, two marten predators – the fisher (*Martes pennanti*) and bobcat (*Lynx rufus*) – were selected because they likely expanded their range or abundance, respectively, following the extensive logging of the 1900s. Successful restoration of the old growth forest mesocarnivore assemblage in the redwood region will require an increase in the amount and connectivity of old forest conditions and reduction of road densities which should result in the expansion of the remnant Humboldt marten population and decreases in the range and abundance of the fisher and bobcat.

Key words: *Martes*, Humboldt marten, marten, mesocarnivores, second growth, restoration, focal species, redwood forest management

Introduction

State of forest reserves of the redwood region

More than 95 percent of the primary, old growth, forests of the redwood region have been logged (Fox 1996, Thornburg et al. 2000) and dramatically altered from what it resembled in pre-European settlement times. Forest reserves, defined here as tribal, state, or federal agency administered lands with the primary goal of maintaining or restoring late-successional and old growth forest characteristics, include <20 percent of the redwood region but are composed of >50 percent logged and degraded, second growth forests (Noss et al. 2000). The greatest concentration of forest reserves occurs in the northern redwood region. Currently Redwood National Park includes >50,000, California State Parks >20,000, Yurok Tribe >15,000 and the Six Rivers National Forest >10,000 ac of fog-influenced second growth forest (California State Parks 2010, Keyes 1995). Collectively, the degraded portions of

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these forest reserves provide the greatest opportunity to restore a large-scale example of the fog-influenced forest ecosystem in the redwood region.

Focal species approach for restoration design and biodiversity conservation

Attempts to restore such a large amount of second growth forest to old growth conditions are unprecedented globally. Restoration requires that either historical or desired conditions be defined and methods identified to reconstruct or accelerate the restoration of compositional and spatial elements towards those conditions. While studying the structure and composition of remnant stands of old growth can provide important information to guide the development of silvicultural treatments, this information alone will not adequately address the needs of many wildlife species. A focal species approach is necessary to identify key limiting compositional features and design restoration actions at large enough spatial. The first step in the application of the focal species approach is to select the subset of species most sensitive to the loss and modification of the compositional and spatial characteristics of old growth forests. The habitat needs of the focal species are then used to guide restoration at all relevant spatial scales (Lambeck 1997). By selecting the suite of species most sensitive to the compositional elements and spatial amounts of old growth forests, they act as “umbrellas” for species that are less sensitive.

The focal species approach has already been in practice in guiding restoration actions in the stream and rivers of the redwood region. Salmonids, such as Coho (*Oncorhynchus kisutch*) and Chinook (*O.* tshawytscha), have been focal species and compositional elements, such as suitable spawning gravels and more recently in-stream large woody structures have been targets for restoration where they have been degraded or lost. Meaningful spatial scales for salmonid habitat restoration have focused on entire watersheds to reduce sedimentation sources and improving overall habitat conditions with the goal of providing the amount of habitat to support populations similar to those supported by historical conditions. Restoration actions for terrestrial habitats have been slower to develop, due in part to the lack of information linking terrestrial wildlife to critical habitat characteristics and the growing acceptance that thinning trees in forest reserves is an essential tool for accelerating the restoration of historical and desired conditions (DeBell et al. 1997, Tappeiner et al. 1997). Given the scale of the challenge of restoring large areas of second growth forest with limited resources, a strategic focal species approach will be necessary to optimize the goals of restoring old forest characteristics and the species that most require them. The objective of this paper is to identify a suite of focal species and use them as the basis for developing strategic restoration guidelines for the terrestrial second growth forests of the northern redwood region.

Methods

Process for selecting focal species

I first identified the suite of species that are unlikely to persist in the redwood region without restoration of features of *old growth forests*. Emphasis was given to taxa which are endemic or near-endemic, >75 percent of their distribution in the redwood region. However, taxa at risk (e.g., threatened or endangered) over larger areas were also considered. I did not consider populations that are secure, unless their presence was identified as being a threat to the persistence of a species at risk. Second, I identified the processes that have contributed to the decline in the abundance or distribution of each candidate species (*fig. 1*). Species with similar responses to processes were grouped and for each process, the most sensitive to the process was selected.

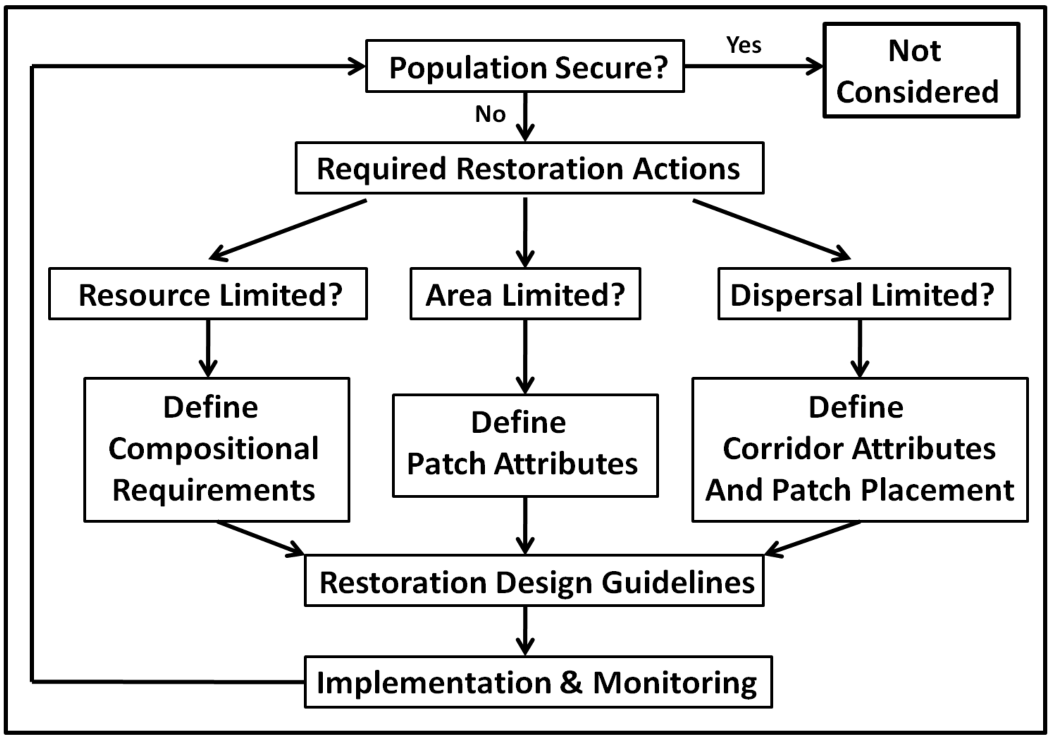

Figure 1—Schematic representation of the steps for the selection of focal species and development of restoration guidelines for reconstructing critical compositional and spatial habitat elements; modified from Lambeck (1997).

Steps for designing restoration guidelines

Landscape restoration and reconstruction will be required for species that are limited by (1) a shortage of critical resources, (2) an inability to move between suitable habitat patches, or (3) insufficient habitat to meet their resource needs or to support a viable population (Lambeck 1997). Once focal species were selected, the first step was to review and synthesize relevant literature on their ecology to identify limiting resources, area limitations, and dispersal limitations. Because everything about a focal species' ecology may not be known, areas of uncertainty were identified. Limiting resources are those that are linked to key life history requirements and lacking them precludes the species occurrence. Area limitations are those that lead to stable occupancy of at least one home range or breeding territory. Dispersal limitations are considered for both within subpopulation movements of individuals and between subpopulation movements.

Results

Focal species selection

Cooperider et al. (2000) identified 18 terrestrial or amphibious vertebrate taxa endemic or near-endemic to the redwood region. Of the nine that occur in the northern redwood region, I conclude only the Humboldt marten (*Martes americana humboldtensis*) is not likely to persist without restoration actions. The Humboldt marten has been extirpated from >95 percent of its historical range and persists in a single population numbering <100 individuals (Slauson et al. 2009b). The remaining eight near-endemic vertebrate taxa are either not strongly associated with old growth forest characteristics (e.g., fog shrew [*Sorex sonomae sonomae*], Pacific coast Aquatic garter snake [*Thamnophis attratus attratus*], California giant salamander [*Dicamptodon ensatus*]), have populations supported on forests managed for timber production (e.g., California tree vole [*Arborimus pomo*]), or have comparatively secure populations in forest reserves (e.g., Del Norte salamander [*Plethodon elongatus*], Wandering salamander [*Aniedes vagrans*]; Cooperider et al. 2000). In addition to the marten, the marbled murrelet (*Brachyramphus marmoratus*) and northern spotted owl (*Strix occidentalis caurina*) are of conservation concern and have significant portions of their populations in the redwood region that likely influence their population status beyond the redwood region. While spotted owls and marbled murrelets are also potential candidates as terrestrial focal species, the remainder of this paper focuses on mesocarnivores.

The loss of old growth forest in the redwood region is the chief process that has driven the decline of the Humboldt marten (Zielinski et al. 2001). While extensive trapping in the late 1800s and early 1900s was the initial driver of population decline for the Humboldt marten (Grinnell et al. 1937), despite the closure of the trapping season for >50 years or approximately 10 marten generations only a single population remains in <5 percent of its historical range. The remnant marten population has been unable to recolonize adjacent areas of second growth forest, or to disperse across second growth landscapes to recolonize suitable old growth patches (Slauson and Zielinski 2003).

In addition to the direct loss of old growth forests, the fragmented state of the remaining old growth forests further threatens population stability and growth for the marten. Highly fragmented habitat is more likely than contiguous old growth habitat to support unstable marten occupancy (Slauson et al. 2010). At the landscape scale, the largest remaining patches of old growth habitat suitable for martens to recolonize are currently too far apart and exist in a second growth matrix not compatible for marten movement and dispersal.

Second growth forests in the northern redwood region up through at least 60 years old support a highly altered mesocarnivore assemblage, dominated by species known to or capable of preying on martens. The conversion of the northern redwood region from primarily old growth to young second growth has facilitated the expansion of the range of the fisher (*Martes pennanti*) and increases in the abundance and distributions of bobcats (*Lynx rufus*), gray fox (*Urocyon cinereoargenteus*). Grinnell et al. (1937) identified the range of the fisher as distinctly inland from the coast and largely on the eastern edge of the redwood region. While some have questioned Grinnell et al.'s (1937) historical range of the fisher in coastal California (DFG 2010), the absence or rarity of fishers in coastal forests is both consistent with their continental distribution (Giblisco 1994, Proux et al. 2000) and present day habitat associations from surveys conducted in the redwood region (Klug 1996, Slauson and Zielinski 2003). In the last century, fishers have expanded their range west and can now be detected near the ocean in second growth forests (Slauson and Zielinski 2003). Surveys within the largest remaining patches of old growth forest, including the area occupied by the remnant marten population, indicate fishers either do not occupy or only use stands near their edges (Slauson and Zielinski 2003). While historical information on range and abundance for bobcats and gray foxes is not as detailed as that for fishers, the pattern of their detections occurring largely in second growth but rarely in the cores of remnant old growth patches mirrors that of the fisher (Slauson and Zielinski 2007; K. Slauson, unpublished data).

Of the six studies reporting 57 predation events on marten in North America, mammalian carnivores were responsible for 75 percent, with bobcats (44 percent), coyotes (19 percent), and fishers (11 percent) the most frequent predators (Bull and Heater 2001, Ellis 1997, Hodgeman et al. 1997, McCann et al. 2010, Raphael 2004, Thompson 1994). Restoration actions will need to also result in the reduction in occupancy and abundance of marten predators in order to be successful in restoring martens and the coastal old growth mesocarnivore assemblage. Therefore, I propose the fisher, a forest specialist, and bobcat, a habitat generalist, as additional focal species.

Resource limitations

Second growth forests are depauperate of two critical resources for martens, dense shrub cover and suitable resting/denning structures. Restoration prescriptions for critical resources should be applied at the stand scale. Larger spatial considerations will follow, see Area Limitations below, such that stand scale restoration prescriptions can be aggregated to meet larger scale needs of focal species.

Dense, spatially extensive shrub cover is a critical component of suitable habitat for the Humboldt marten and >95 percent of all detections (n = 50; Slauson et al. 2007, Slauson et al. 2009b), rest sites (n = 55; Slauson and Zielinski 2009a), and active telemetry locations (n = 235; K. Slauson unpublished data) have occurred in stands that support it. Dense, spatially extensive shrub cover is a common understory structural layer in mature forests in the northern redwood region (Sawyer and others 2000). Martens typically occupy old growth stands with shrub layers that average >70 percent cover, are dominated by long-lived shade tolerant species (e.g., evergreen huckleberry [*Vaccinium ovatum*], salal [*Gaultheria shallon*]), and form continuous structural layers that spread throughout and between adjacent forest stands (Slauson et al. 2007). Shrub layers provide multiple direct and indirect benefits to martens including: berries for food, overhead cover from avian predators, cover from larger-bodied mesocarnivores, and mast that supports prey population. Slauson and Zielinski (2007) measured shrub cover in >800 plots in northwestern California and found that dense shrub cover has been significantly reduced in logged second growth forests <20 km from the coast.

Martens use resting structures daily between forging bouts to provide secure locations to sleep, conserve energy, and in some cases consume prey. Martens show little short-term re-use of structures and thus require many resting structures throughout their home ranges to provide resting locations close to where they are foraging and to provide for seasonal thermoregulatory needs (e.g., platforms in summer for cooling, cavities and chambers in winter to provide refuge from precipitation and warmth; Schumacher 1999, Taylor and Buskirk 1994, Wilbert 1992). Humboldt martens select large live and dead woody structures, including snags, logs, and live trees, that are typically >90 cm in diameter (Slauson and Zielinski 2008). Slauson et al (2010) found second growth stands in the Lost Man creek watershed and Mill creek watershed contain no suitable live tree structures and <10 percent the density of suitable snag structures compared to old growth stands occupied by martens. Further, compared to marten occupied stands, suitable downed log densities were similar in the older second growth stands in the Lost Man creek watershed but were <20 percent of the densities in the Mill creek watershed. Natural recruitment of suitable resting structures will likely not begin for >2 centuries, as most resting structures used by martens were of greater age when living (Slauson and Zielinski 2009a).

Area limitations

In California, martens maintain home ranges of approximately 300 ha to provide for their year-round resource needs (Spencer et al. 1983). While Humboldt martens use stands other than old growth for foraging, a large patch of old growth forest is a prominent component of their home ranges (K. Slauson unpublished data). Humboldt martens are highly selective for patches of old growth habitat >100 ha (Slauson et al. 2007). Patches <150 ha supported unstable marten occupancy (Slauson et al. 2009b). Restoration actions that enlarge existing small patches of old growth habitat may be the most effective in the short term while designing stand scale restoration actions to aggregate into patches of >150 ha will provide for the development of patches that cam become the cores for new home ranges once suitable conditions are restored.

Dense road networks are ubiquitous in second growth areas. The Humboldt marten makes little use of roads (Slauson et al. 2010) and does not occupy areas with high roads densities (Slauson 2003). Roads fragment the dense understory shrub layers and create networks of edge habitat that has no natural precedent in coastal forests. Slauson et al. (2010) found that >75 percent of habitat generalist species detections occurred on roads and were dominated by bobcats and gray foxes. The development of roads created networks of edge habitat that has likely facilitated the increase in distribution and /or the abundance of bobcats and gray foxes in second growth landscapes. Furthermore, encounters between martens and potential predators on roads may lead to higher rates of predation as martens will be farther from escape cover (e.g., tree boles for climbing) than encounters in forest interiors. Without minimizing or eliminating road networks, martens may not become re-established in areas where other suitable habitat characteristics have been restored.

Dispersal limitations

Occupied marten home ranges typically do not occur in isolation from other adjacent marten home ranges, thus >150 ha patches should be aggregated into local clusters of ≥ four patches to provide enough habitat to support adjacent home range establishment by multiple individuals of each sex. While martens are capable of long distance dispersal movements >50 km (e.g., Slough 1989), most martens that disperse and successfully establish home ranges move <5 km (Johnson et al. 2009).

Patch clusters should be located as close as possible to either occupied habitat or adjacent to other clusters < 5 km. Finally, initial restoration actions should be done in areas as proximal as possible to currently occupied habitat to increase the probability that restoration areas can be recolonized by martens once suitable conditions return.

Restoration design guidelines

Restoration actions that will benefit the regeneration of dense shrub cover include reducing overhead cover from trees by thinning. Thinning treatments conducted 30 years ago (see Keyes 2005, Veirs 1986) have restored shrub cover to a median value of 55 percent compared to <10 percent in unthinned control stands (Slauson et al. 2010). Recently thinned stands (2 to 7 years ago) in the Mill creek watershed of Del Norte Coast Redwood State Park have also significantly increased shrub cover to a median of 30 percent compared to <10 percent in unthinned stands. Without canopy thinning, most stands would likely lose shrub cover completely if left to self-thin, requiring re-establishment of shrub species from seed and many more decades before the shrub layer is restored.

Three methods may increase the number of resting structures in second growth habitats: enhancement, augmentation, and deployment of surrogate structures. Because these methods have not been developed for marten in the redwood region, these efforts should be initially undertaken in research and demonstration mode. Enhancement refers to creating suitable resting locations within existing structures such as using chainsaws to create cavities in downed logs and live trees of suitable size (e.g., Brown 2002). Augmentation refers to the transport of suitable structures to sites where they are deficient. This is likely the most expensive option as it requires both the acquisition of large diameter logs and deposition in target sites, necessitating the use of heavy machinery. The deployment of surrogate structures involves the building of artificial resting structures and placement in target stands. In Scotland, biologists have developed a marten den box has successfully provided the European pine marten with resting and even denning locations in forest stands where they are lacking (VWT 2010). Enhancement and the use of surrogate structures are likely the two techniques that if used in combination will provide the most cost effective methods to provide resting structures at large spatial scales. However, augmentation may be necessary in large areas where large diameter structures are particularly scarce.

Implementation and monitoring

One of the most critical phases of restoration projects is the monitoring that follows implementation to provide critical feedback to guide future restoration actions. While actions intended to restore critical habitat elements where they have been lost is founded in the ecology of the most sensitive species, success is not guaranteed. Therefore, post implementation monitoring will be critical for evaluating restoration actions and to provide any refinement for future restoration actions.

Discussion

The degraded state of much of the second growth forests in reserves cannot be left unmanaged due to the presence of many characteristics that are far outside the natural range of variability. Tree densities and species compositions depart greatly from those present in mature and old growth forests (Keyes 1995, Veirs 1986). Just as the threats of sediment delivery lead to actions to decommission dense roads networks in these forests for restoring salmonid populations, terrestrial habitats too will need restoration actions to reconstruct critical compositional and spatial elements to accelerate the restoration of habitat for sensitive the terrestrial fauna of redwood forests. In combination with silvicultural objectives and aquatic focal species restorations actions, an approach using terrestrial focal species provides a comprehensive and strategic basis for designing restoration actions with the best chance for restoring old growth forest communities and ecosystems northern redwood region.

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Northern California Redwood Forests Provide Important Seasonal Habitat for Migrant Bats

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Abstract

Bats are known to roost in redwood forests year-round, but their activities outside the summer season are poorly understood. To improve understanding of the use of redwoods by resident and migrant bats, we conducted 74 mist net surveys between February 2008 and October 2010. Captures were dominated by Yuma myotis (*M. yumanensis*) in the summer and silver-haired bats (*Lasionycteris noctivagans*) in the winter. During November-February, silver-haired bats, accounted for 78 percent of 23 captures and male:female sex ratio was (9:9). By contrast, during June-August, silver-haired bats accounted for 13.8 percent of 269 captures and sex ratios were highly male skewed (34:3). In combination with other regional information, this indicates that female silver-haired bats migrate to redwood forests. To infer summer locations of bats captured in redwood forests, we analyzed stable isotopes of hydrogen, carbon, nitrogen, and sulfur in their fur. Despite spatial segregation between male and female silver-haired bats during presumed molt period, we did not find differences between the sexes in range of isotope values in their fur. Nor were their values different from Yuma myotis. Our findings highlight some of the challenges in using stable istotope analysis to infer migratory pathways in bats.

Key words: bats, *Lasionycteris noctivagans*, *Lasiurus cinereus*, *Myotis yumanensis*, redwoods, stable isotopes, migration, winter

Introduction

Studies of bats in redwood forests have focused largely on their roosting ecology, in particular their use of basal hollows (Fellers and Pierson 2002, Gellman and Zielinski 1996, Rainey et al. 1992, Zielinski and Gellman 1999, Zielinski et al. 2007). A novel result of two of these studies was documentation of bat activity during winter (Gellman and Zielinski 1996, Zielinski and Gellman 1999). Autumn–spring is the period when temperate-zone bats engage in important activities that critically affect their annual energy balance such as migration, mating, and hibernation (Weller et al. 2009).

The silver-haired bat (*Lasionycteris noctivagans*, hereinafter LANO) is a tree-roosting species that is thought to undertake seasonal migrations between its summer and winter habitat (Cryan 2003). Male and female LANO are geographically segregated during the summer months throughout much of North America (Cryan 2003). In Douglas fir forests of northwestern California (hereinafter NWCAL) LANO was one of the most frequently captured species during summer; but captures were > 95 percent males (TJW unpublished data). The limited capture work in

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NWCAL from October-March, suggested that female LANO were more abundant during this time period (Zielinski and Gellman 1999). However, sample sizes were too low to accurately estimate sex ratios by season. Therefore, an important objective of this study was to verify whether the proportion of female LANO increased during autumn–spring in redwood forests of NWCAL. If so, it would suggest that female LANO move to the redwoods for the purposes of wintering, and presumably, mating.

We investigated the use of stable isotopes to help determine movement patterns and infer the summer range of LANO individuals that used NWCAL during winter. Hydrogen isotopes (δD) are commonly employed to infer migratory movements of vertebrates by exploiting characteristic latitudinal and elevational structure in the isotopic composition of precipitation (Bowen et al. 2005, Hobson and Wassenaar 1997). Because tissues such as hair and feathers are metabolically inert, stable isotope ratios reflect the geographic areas where they were synthesized rather than where they are collected (Hobson and Wassenaar 1997). Bats generally molt their fur once per year during late-summer (Constantine 1957) and δD analysis has been used to estimate minimum distances traveled between the summer range and location at time of fur collection (Cryan et al. 2004).

We explored the use of carbon ($\delta^{13}C$), nitrogen ($\delta^{15}N$), and sulfur ($\delta^{34}S$) isotopes to provide additional information and potentially improve the accuracy of geographic inference (Rubenstein et al. 2002, Wunder et al. 2005). We were particularly interested in evaluating the efficacy of $\delta^{34}S$ because of the proximity of our study area to the Pacific Ocean (Zazzo et al. 2011). The ocean represents a uniform reservoir of sulfate with a sulfur isotopic composition of about 21‰. Due to sea spray effects, we expected bats that grew their fur in redwood forests to have higher values of $\delta^{34}S$ than individuals that grew their fur further from the ocean.

Our use of stable isotopes to infer seasonal movements of bats focused on quantifying the range of values for each isotope within the local population and comparing it against expectations of their movements. We speculated that fur samples from female LANO would have a wider range of isotope values than for males because they are largely absent from NWCAL during summer. However whether male LANO were year-round residents of NWCAL is uncertain. Therefore, we attempted to constrain inference by quantifying baseline levels of isotopic variation in a summer resident species, Yuma myotis (*Myotis yumanensis*, hereafter MYYU), while remaining cognizant of inter-species differences (Britzke et al. 2009). The seasonal movements of MYYU are largely unknown (Verts and Carraway 1998) but based on size and ecomorphology, MYYU is not expected to travel long distances between summer and winter habitat. Accordingly, we expected narrower ranges of isotopic values in year-round resident MYYU than female LANO. If male LANO were summer residents of NWCAL, we expected their range of isotopic values to be similar to MYYU. If instead, the range of isotopic values in male and female LANO were similar to each other but different from MYYU, this might indicate that both sexes are moving to the redwoods from summer habitat elsewhere.

Our study had three interrelated goals: 1) characterize year-round bat activity in redwood forests of NWCAL; 2) determine whether female LANO use redwood forests as wintering habitat; 3) evaluate the use of stable isotopes to infer summer locations of LANO and MYYU in NWCAL.

Study area

Our study was conducted in redwood forests of Mendocino, Humboldt, and Del Norte Counties in NWCAL (*fig. 1*). We selected 18 sites in or adjacent to redwood forests based on ease-of-access and the potential to provide conditions conducive to bat capture during all seasons of the year. All sites were stream reaches ≤ 300 m long that generally retained surface flow throughout summer and early autumn, yet were still wadeable during high flows in winter. Capture sites were located on public (California State Parks, Redwood National Park) and private lands at distances of 1.4 to 31.4 km from the Pacific Ocean (*fig. 1*).

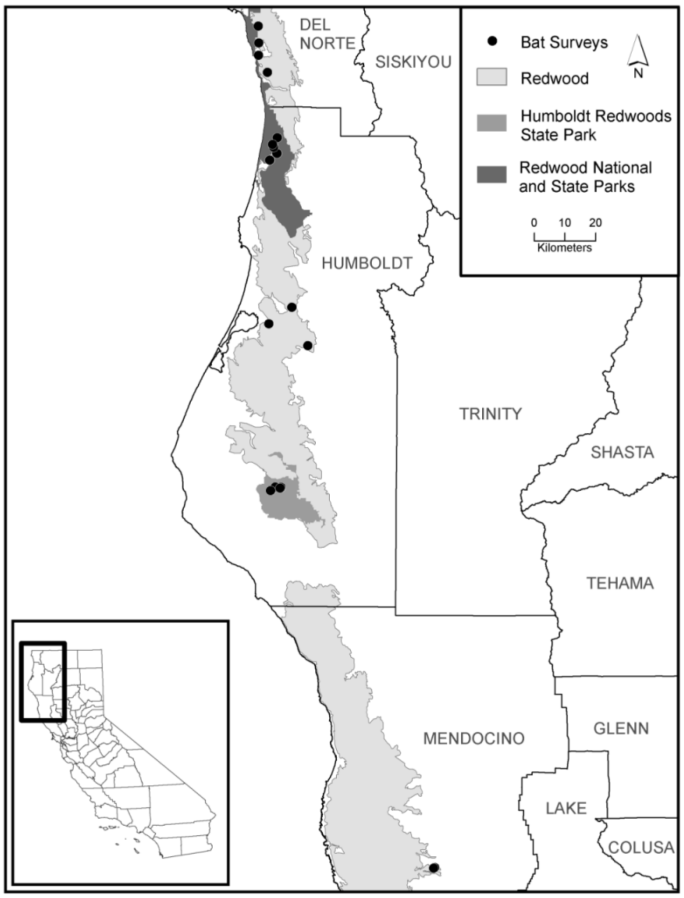

Figure 1—Location of bat captures sites relative to redwood forest habitat in northwest California.

Methods

Field

We captured bats by setting two to four (mean= 3.6 SD=0.55) 2.6 m high mist nets over the creek at each site. We selected the number and length (6 to 12 m) of mist nets to best suit the physical characteristics of each site. Mist nets were opened at sunset for a minimum of 2 hours or until at least 1 hour had passed since the last capture. Individuals were identified to species based on examination and measurement of external morphological features; MYYU was identified using the methods of Weller et al (2007). Sex was determined via external inspection (Racey 2009). We clipped 0.3 to 6 mg of fur from the dorsum of LANO and MYYU individuals for subsequent stable isotope analyses. All bats were released on site the same night.

Stable isotope analyses

Fur samples were triple-washed in a 2:1 chloroform/methanol solution, air dried, and weighed into silver (δD) and tin capsules (δ^{34}S and δ^{13}C/ δ^{15}N). Samples were allowed to air equilibrate to ambient laboratory conditions for ≥ 2 weeks prior to δD analyses (Wassenaar and Hobson 2003). All isotopic measurements were made by continuous flow isotope ratio mass spectrometry, with results reported in delta (δ) notation, expressed as parts per thousand (‰) relative to internationally accepted scales; δD data are specific to non-exchangeable hydrogen. Mean differences between duplicate samples from the same individual were ± 4.1‰ for δD (n = 29), ±0.2‰ for δ^{13}C (n=6), ± 0.1 ‰ for δ^{15}N (n=6), and ± 0.3‰ for δ^{34}S (n=11); quality control and assurance was verified by repeated analyses of an in-house keratin standard, as well as primary standards analyzed as unknowns.

Data analyses

We tested whether monthly sex ratios of LANO or MYYU differed from 50:50 using the conditional test (Przyborowski and Wilenski 1940). The conditional test assumed that the observed count followed a Poisson distribution for both sexes and months with counts for sexes within a month being independent. We used a 2-factor Analysis of Variance (ANOVA) with species and sex as fixed effects to evaluate differences in isotope values. The ANOVA assumed separate variances for each species and residuals for each isotope and sex-species combination were normally distributed. We evaluated differences in least-square means using a t-test with Tukey-Kramer adjustment of p-values; all tests performed were 2-tailed.

Results

Seasonal activity patterns

Between February 2008 and October 2010 we conducted 73 mist net surveys and captured a total of 476 bats, with a mean of 6.5 (SE = 0.78) per survey. Overall we captured 10 species of bats including *Eptesicus fuscus* (*n* = 15), *Lasiurus blossevillii* (*n* =2), *L. cinereus* (*n* = 36), LANO (*n*=116), *Myotis californicus* (*n* = 27), *M. evotis* (*n*=2), *M. lucifugus* (*n*=22), *M. thysanodes* (*n*= 7) *M. volans* (*n* =7), and MYYU (*n* = 129). An additional 113 individuals were identified only as either *M. lucifugus* or MYYU. Bats were captured in every month surveyed though overall capture rates were higher during May-July (*fig. 2*). MYYU, or individuals identified as either MYYU or *M. lucifugus*, comprised 65 percent of 269 bats captured during June-August. From May-September. females composed 36 to 63 percent of monthly captures of MYYU (*fig. 3*), and sex ratios did not differ significantly from 50:50 in any month (range: p=0.092 in Oct to p>0.999 in May). LANO was the second most frequently captured species and capture rates were relatively consistent throughout the year (*fig.* 3). LANO comprised 79% of 34 bats captured during November–March. Sex ratios of LANO were male-biased in March-July (all p≤0.031) and to a lesser extent in August (p=0.065). Sex ratios of LANO were not different than 50:50

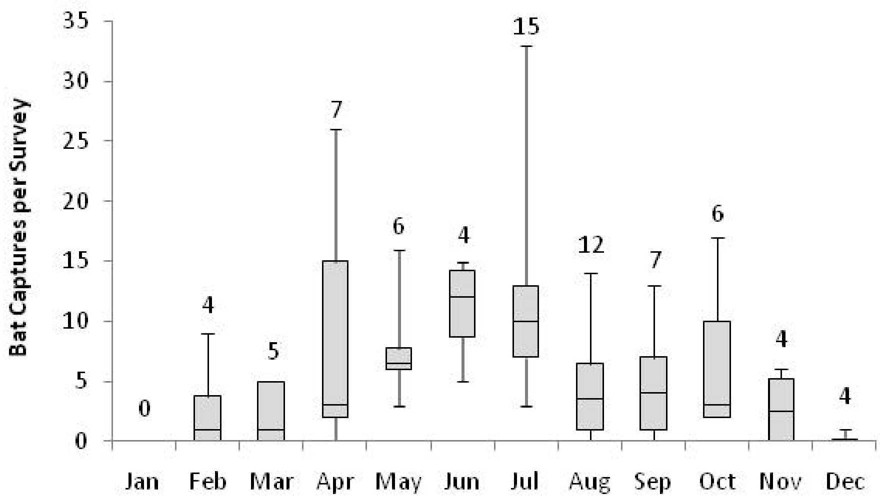

Figure 2—Number of bat captures per survey (all species combined) by month in redwood forests of northwestern California 2008 – 2010. Boxes represent 1st and 3rd quartiles, horizontal line is median, and whiskers represent maximum and minimum captures per survey. Number above whiskers is total number of surveys conducted in each month.

in any other month (all $p \geq 0.500$). Capture rates for male LANO were lowest (0.29 – 0.77 captures/survey) during August-October, however they comprised 67 to 100% of LANO captures during these months (fig. 3). Though not a primary focus of this study, 36 hoary bats were captured (6 from 11 April-16 May and 27 from 25 September-5 October); all but two hoary bats were males.

Stable isotopes

We analyzed stable isotope content in fur samples collected from bats during 2008 and 2009. We found no difference in δD values between male and female LANO (t_{65}= -0.57, p = 0.571, *table* 1). Although MYYU fur samples averaged 5 ± 3‰ higher than LANO (*table* 1), the difference was not significant ($F_{1,114}$= 2.98, p = 0.087). The range of δD values in LANO fur from Jun-Aug (-94 to -52‰), when male:female sex ratio of samples were 13:3, was not substantively different from Nov-Feb (-103 to -55‰) when sex ratios of samples were 8:8. Further, neither differed from the range of values for MYYU during June-August (-111 to -49‰), or throughout their April-November activity period (-111 to -41‰).

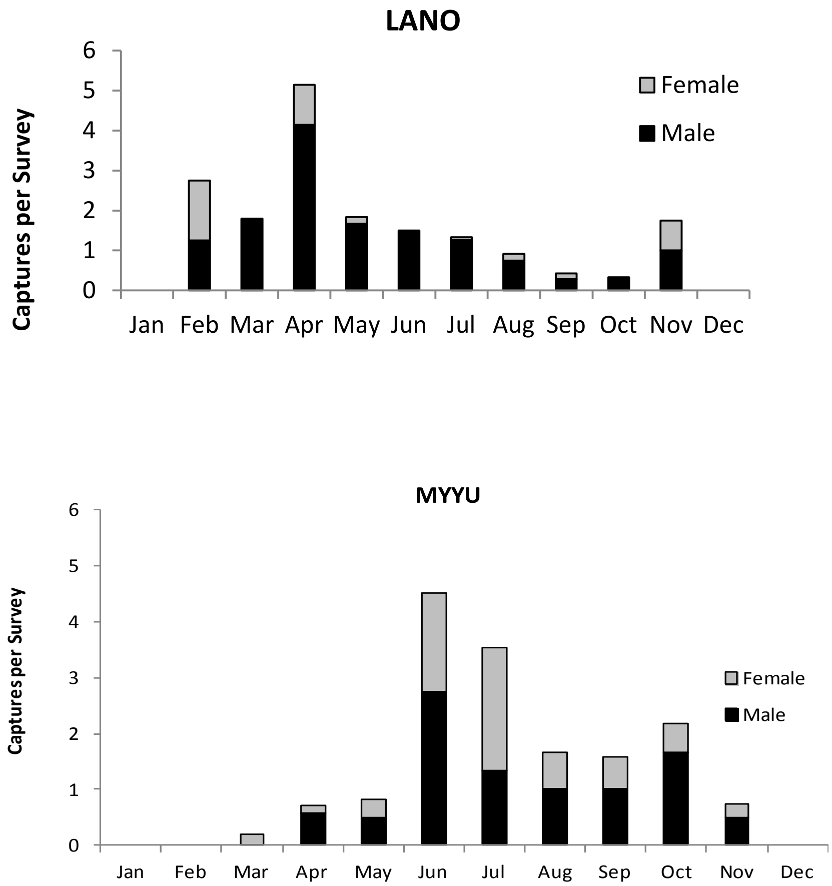

Figure 3—Mean monthly number of captures per survey and sex ratios for Yuma Myotis (MYYU) and silver-haired bats (LANO) captured during 74 mist net surveys in redwood forests of northwestern California 2008 to 2010.

Fur of LANO was higher than MYYU in $\delta^{13}C$ ($F_{1,49.2}$= 5.86, p = 0.019; *table* 1) though mean difference between species was small (0.8 ± 0.3‰). We found no difference in $\delta^{15}N$ between LANO and MYYU ($F_{1,70.9}$= 2.77, p = 0.101). On average LANO fur was 1.9 ± 0.7‰ higher for $\delta^{34}S$ than MYYU ($F_{1,91.1}$= 6.60, p = 0.012, *table* 1) and patterns differed between the species according to site where fur was collected. MYYU exhibited a visual trend toward lower $\delta^{34}S$ with increasing distance from the Pacific Ocean, but LANO did not (*fig. 4*). Neither sex (all $p \geq 0.240$), nor the interaction of species and sex (all $p \geq 0.092$), explained significant portions of the variation for any of the four isotopes.

Discussion

Seasonal activity patterns

Our study confirmed that bats are active year-round in redwood forests of NWCAL but that the species responsible for this activity differ markedly on a seasonal basis. MYYU was the most frequently captured species and followed a

Table 1—*Mean stable isotope values (‰) in fur by sex, for Yuma myotis (MYYU) and silver-haired bats (LANO) captured during all seasons of the year in redwood forests of northwestern California 2008–2009.*

Element	Species	Sex	*n*	Mean	Range
Hydrogen (δD)	LANO	M	51	-75	-113 to -39
		F	16	-78	-94 to -53
	MYYU	M	34	-75	-111 to -53
		F	22	-67	-105 to -41
Sulfur (δ^{34}S)	LANO	M	42	6.6	-3.5 to 12.9
		F	15	6.2	1.6 to 9.2
	MYYU	M	32	3.5	-4.1 to 12.7
		F	21	5.6	-0.8 to 10.7
Carbon (δ^{13}C)	LANO	M	31	-23.0	-25.0 to -21.5
		F	11	-23.1	-24.8 to -21.9
	MYYU	M	24	-23.9	-29.6 to -21.5
		F	12	-23.8	-26.2 to -21.2
Nitrogen (δ^{15}N)	LANO	M	31	6.4	4.9 to 7.8
		F	11	6.8	4.2 to 9.1
	MYYU	M	24	7.0	5.0 to 10.4
		F	12	7.1	5.6 to 8.6

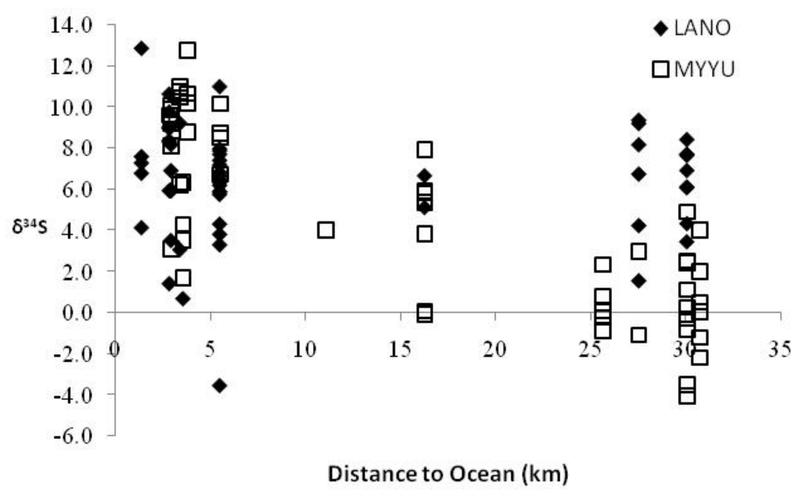

Figure 4—Sulfur isotope ratios (δ^{34}S) in fur of silver-haired bats (LANO) and Yuma myotis (MYYU) versus distance of sample site from the Pacific Ocean during surveys in redwood forests of northwest California 2008 to 2009.

pattern typical for a summer-active resident bat species with captures peaking during June and July and LANO was the most frequently captured species during winter. Hoary bats made significant, if episodic, visits to redwood forests presumably as they migrated through the area.Previous studies with guano traps in basal hollows established that bats are active throughout the winter in NWCAL, though at much lower levels than during summer (Gellman and Zielinski 1996, Zielinski and Gellman 1999). However, guano traps likely index of activity by *Myotis* spp. and big brown bats because LANO are not likely to make significant use of basal hollows (Zielinski et al. 2007). Both sexes of MYYU are summer residents of redwood forests and they likely hibernate during winter, although hibernation sites are unknown throughout their range (Verts and Carraway 1998). Although winter captures of LANO have previously been recorded in redwood forests (Zielinski and Gellman 1999), our results establish that such activity is common. Further our work is in agreement with previous reports that captures of LANO in redwood forests of NWCAL are nearly all male during summer while sex ratios are roughly equal from November-March (Zielinski and Gellman 1999). This strongly indicates that female LANO migrate elsewhere for parturition and pup-rearing before returning to the northwest coast of California to overwinter.

The closest summer records of female LANO are from the upper Sacramento River Canyon (Rainey 1994), Whiskeytown National Recreation Area, and Lassen National Forest (Morrell and Duff 2005), all > 150 km inland. Whether LANO that winter in redwoods of NWCAL spend summers in these areas, or are drawn from a wider range of geographic locations, is unknown. The attractiveness of NWCAL may be explained by relatively moderate winter temperatures that are conducive to tree-roosting (Cryan 2003). Areas, such as redwood forests of NWCAL, which maintain ambient winter temperatures of 5 to 10 °C may be energetically optimal for LANO (Dunbar 2007). Further, cool summer temperatures in redwood forests may enable male LANO to maximize use of torpor (Weller et al. 2009). The presence of both male and female LANO beginning in autumn and continuing throughout winter suggests that redwood forests may be important locations for mating in this species. Hence redwood forests of NWCAL could prove a vital area for maintenance of LANO populations at regional- or, perhaps, continental-scales.

Stable isotopes

Ours is the first study to report stable isotope values in the fur of either MYYU or LANO. The range of values for all four isotopes was similar among both sexes and species. Contrary to our expectations we did not find that isotope values in female LANO differed from MYYU or male LANO. Instead, mean values were similar and LANO females exhibited the narrowest range of values for δD, $\delta^{13}C$, and $\delta^{34}S$ (*table 1*), although this group had the smallest sample size.

The 40 to 70 ‰ range in δD we observed is similar to ranges of four species in the eastern U.S. (Britzke et al. 2009) and a sedentary population of big brown bats in Colorado (Cryan et al.[3]). The wide range of δD values we observed could be associated with bats making relatively short regional movements (e.g., to Sacramento Valley) across a steep gradient in δD values from the Pacific Coast (Bowen et al. 2005). As such, the range of δD values may not differ much between male and female LANO or MYYU individuals. We observed a 70‰ range of δD values for MYYU captured over 1.4° range in latitude. This is similar to the apprximately 60‰ range of δD over a 2° range in latitude reported for little brown bats (*M. lucifugus*;

[3] Cryan, P.M.; Stricker, C.A.; Wunder, M.B. **Biologically structured isotopic variation in a sedentary population of bats.** Unpublished draft supplied by authors.

Britzke et al. 2009). This level of within-area variability in δD is troubling because it could equally have been interpreted as movements over multiple degrees of latitude (Wunder et al. 2005, Cryan et al. unpublished draft); which seems particularly unlikely for MYYU. These findings underscore the challenges inherent in using isoscapes to characterize migratory movements of animals because assignment of tissue origin requires characterization of the isoscape at multiple spatial and temporal scales (Wunder 2010).

The range of stable isotope values present in local bat communities has only recently been the subject of study (Britzke et al. 2009, Cryan et al. [see footnote 3]), and it is possible that it may be great enough to preclude identification of all but long distance seasonal movements (e.g. > 500 km). A primary impediment to the use of stable isotopes in bat fur is poor understanding of molt (Britzke et al. 2009). A single molt in bats is expected during July and August (Constantine 1957, Cryan et al. 2004) but molt may be protracted over a month, even at a single locale (Constantine 1957). In addition, timing of molt may vary by latitude, species, or sex (Constantine 1957). The consequences of these molt characteristics are magnified if seasonal movements are concurrent with new fur growth (Britzke et al. 2009). Further impediments include poor resolution of δD baselines relative to the scale of daily or seasonal movements (Wunder and Norris, 2008), lack of controlled experiments linking diet and water to isotope values in mammalian consumers (Martinez del Rio et al. 2009), and inherent system noise.

Although used successfully in other studies (Rubenstein et al. 2002) the combination of δD with $\delta^{13}C$ and $\delta^{15}N$ did not appreciably improve our understanding of the seasonal movement patterns of LANO or MYYU. The range of values we observed were relatively similar among the four species-sex groups we evaluated and to a population of big brown bats in Colorado (Cryan et al. unpublished draft). We believe the most parsimonious explanation for the wide range in $\delta^{13}C$ and $\delta^{15}N$ for both MYYU and LANO is that they are habitat and dietary (Ober and Hayes 2008) generalists.

We conjectured that $\delta^{34}S$ may provide longitudinal inference because of our study area's proximity to the Pacific Ocean (Zazzo et al. 2011). Indeed we observed a pattern in MYYU fur with lower $\delta^{34}S$ values as distance from coast increased; a pattern not observed in LANO. Taken together these observations are consistent with the hypothesis that capture locations reflect location of fur growth for MYYU but not LANO. If confirmed, this pattern suggests that the $\delta^{34}S$ gradient is quite steep inland from the ocean and that $\delta^{34}S$ may be a useful tool for evaluating local- versus regional-scale movements of bats.

Conclusions

As a result of year-round capture activities we determined that the bat communities in redwood forests of NWCAL are dominated by MYYU in the summer, LANO in the winter, and may also provide important stopover habitat for hoary bats during fall and spring migration. Female LANO appear to migrate from the redwoods during spring and return to them during autumn. Male and female LANO are only together during autumn through spring when mating occurs, further emphasizing the importance of redwood forests of NWCAL to LANO populations at regional- and perhaps continental-scales.

Despite spatial segregation between male and female LANO during their presumed molt period, we did not find differences between the sexes in range of isotope values in their fur. Further, the range of isotope values in the presumably migrant LANO did not differ from the presumably resident MYYU. Although differences in mean values were observed between the species for $\delta^{34}S$ and $\delta^{13}C$, broad ranges in individual dietary preferences may obscure inferences about summer ranges of MYYU and LANO in our study area (Cryan et al. unpublished draft). Nevertheless, we noted a promising trend between $\delta^{34}S$ and distance to coast for MYYU which may prove useful in future work.

Acknowledgments

We thank M. McKenzie for assisting with bat captures and sample preparation and C. Gulbransen for conducting the isotope analyses. J. Baldwin provided statistical assistance. Bill Zielinski provided helpful comments on an earlier draft of this manuscript and Paul Cryan provided helpful guidance throughout. Funding for this work was provided by Save the Redwoods League and USDA Forest Service Pacific Southwest Research Station. Use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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Measurements of Key Life History Metrics of Coho Salmon in Pudding Creek, California[1]

David W. Wright[2], Sean P. Gallagher[3], and Christopher J. Hannon[4]

Abstract

Since 2005, a life cycle monitoring project in Pudding Creek, California, has utilized a variety of methodologies including an adult trap, spawning surveys, PIT tags, electro-fishing, and a smolt trap to estimate coho salmon adult escapement, juvenile abundance, juvenile growth, winter survival, and marine survival. Adult coho salmon escapement and smolt abundance are closely related when spawner abundance is high. However, with decreasing escapement, smolt abundance appears to stabilize. Adult abundance has declined dramatically, from 1200 in 2005 to 20 in 2010. Corresponding smolt abundance initially declined from 24,000 to approximately 15,000 but maintained at that level despite continued declines in escapement. Winter survival ranged from 17 to 80 percent, and appeared to increase as the juvenile population decreased. Juvenile growth rates within the upper watershed were low during summer and significantly higher in spring and fall. PIT tag detections indicated that thirteen percent of all coho outmigrants were two year old smolts that were smaller at tagging than one year old outmigrants, but larger than one year old fish at outmigration. Over three complete coho life cycles (2006 to 2010), ocean survival appeared to strongly affect adult returns. In all years since 2006, ocean survival for Pudding Creek spawners was notably lower than reported by others in streams throughout the Pacific Northwest in previous years.

Key words: coho salmon, escapement, life history, smolt, Mendocino County

Introduction

Pudding Creek is a small coastal tributary in Mendocino County, northern California, known to support populations of coho salmon (*Oncorhynchus kisutch*) and steelhead (*O. mykiss*). As a component of a larger regional project encompassing most streams and rivers in Mendocino County, the Pudding Creek Coho Salmon Project goals include: 1) estimation of adult escapement, 2) estimation of summer juvenile abundance, 3) estimation of outmigrant production, and 4) characterization of life-history patterns. This report characterizes life history traits of coho salmon inhabiting Pudding Creek.

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[1] This paper was presented at the redwood science symposium: coast redwood forests in a changing California: a symposium for scientists and managers, June 21-23, 2011 Santa Cruz, California.
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Methods

Project area

The project area includes all fish-bearing stream reaches (approximately 22 km) within the Pudding Creek watershed (*fig. 1*). Most of the watershed is presently managed for timber production, and over 70 percent of the watershed is owned by the Hawthorne Timber Company, LLC and managed by CTM. Vegetation in the area is primarily comprised of coast redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*). Red alder (*Alnus rubra*), willow (*Salix* spp.), and big-leaf maple (*Acer macrophyllum*) are the primary constituents of the riparian canopy, which is nearly continuous throughout the stream network. Pudding Creek drains approximately 45 km^2. Streambed gradient is generally < two percent throughout the mainstem reaches.

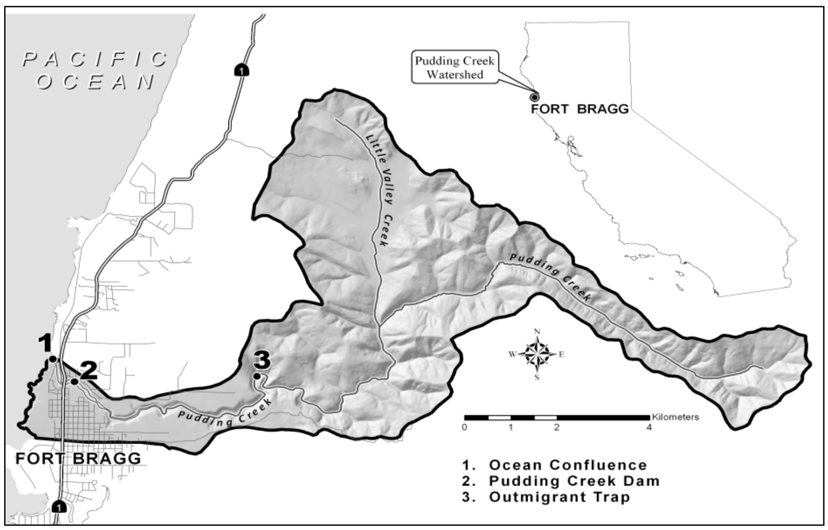

Figure 1—The Pudding Creek watershed project area.

The Pudding Creek watershed includes a dam and an associated impoundment located 0.7 km upstream of the confluence with the ocean (*fig. 1*) which historically was a water supply for the now defunct Georgia Pacific mill site. The impoundment has a surface area of nearly 0.09 km^2 and a volume of approximately 25 hectare-meters. The resulting reservoir inundates nearly 4.3 km of the stream network.

Due to the multiple life-history stages of coho salmon present in the Project Area, several study methods were used to characterize their abundances and life-history patterns, which follow.

Adult trapping and spawning estimates

Adult coho salmon are trapped and tagged at the Pudding Creek Dam fish ladder. Spawning surveys upstream of the pond produce estimates of spawning numbers using mark-recapture methods. The "recapture" component is the re-sight of fish tagged at the dam made by surveyors in the field.

Outmigrant trapping

Outmigrating fish are captured in a rotary screw trap (RST) with a 5 ft diameter cone. The trapping station is 6.6 km from the ocean confluence and 1.6 km from the top of pond habitat formed by the dam (*fig. 1*). The trap is operated from February until mid June. Fish are identified, implanted with PIT tags, and size (fork length) and weight are measured. Smolt population estimates are generated using the Darroch Analysis with Rank Reduction (DARR) method (Bjorkstedt 2000). During springtime low flow periods, juvenile salmonids are also captured at the Pudding Creek Dam on an opportunistic basis.

Juvenile growth modeling

The basic model assumption is that the weight of any individual as a function of time, w_t, satisfies the stochastic differential equation

$$\frac{dw_t}{w_t} = g_t dt + \sigma\, dB_t ,$$

where g_t is the instantaneous growth rate at time t and B_t is a standard Brownian motion. Under these assumptions, the growth over a finite time interval $[t_1, t_2)$ is

$$\ln(w_{t_2}) - \ln(w_{t_1}) = \int_{t_1}^{t_2} g_t\, dt + \sigma(t_2 - t_1)^{1/2} \varepsilon ,$$

where ε is a standard normal deviate.

Assuming that the same g_t and σ apply to all individuals, the probability of observing N pairs of recoveries, where the k th recovery record includes the times t_{k1} and t_{k2} of the recoveries and the weights w_{k1} and w_{k2} at these times, is

$$L(\sigma, g) = \prod_{k=1}^{N} \frac{1}{\sqrt{2\pi\sigma^2 (t_{k2} - t_{k1})}} \exp\left(-\frac{\left(\ln(w_{k2}) - \ln(w_{k1}) - \int_{t_{k1}}^{t_{k2}} g_t dt\right)^2}{2\sigma^2 (t_{k2} - t_{k1})} \right).$$

For the generated results, the growth function g was taken to be piecewise constant, depending on the year and season:

$$g = \begin{cases} g_{\text{Y, winter}} & \text{from October of year Y through February of year Y} + 1 \\ g_{\text{Y, spring}} & \text{from March through May of year Y} \\ g_{\text{Y, summer}} & \text{from June through September of year Y} \end{cases}$$

Counting σ, this gives a statistical model with fourteen unknown parameters, from Spring 2006 through Spring 2010[5].

Late-summer abundance surveys

Juvenile abundance during the summer rearing period is assessed using an electro-fish sampling protocol based on a Generalized Random Tessellation Stratified (GRTS) sample draw. At 10, approximately 50 m selected reaches, juvenile

[5] The data extend into June of 2010; this was folded into the “Spring 2010” period.

abundance is estimated with depletion electro-fish sampling. PIT tags are implanted in unmarked fish.

PIT tags

All juvenile salmonids greater than 65 mm captured during summer surveys or during outmigrant trapping, are implanted with 11 mm full duplex (FDX) PIT tags.

PIT tag arrays

To detect fish at the Pudding Creek Dam that have been PIT tagged at upstream locations, multiple antennae arrays have been affixed around the dam flashboards. These arrays record a time stamp for each fish detected.

Results

Adult escapement

Of the two methods traditionally used to estimate adult escapement, the mark-recapture technique has proved most reliable. Thus, fish per redd estimates are shown only for 2002 and 2003; the years before mark-recapture experiments were initiated (*fig. 2*). Overall, fewer coho returned to spawn in water year 2010 than in previous project years. For comparison, escapement from eight regional streams is shown for the same time series (*fig. 3*)[6].

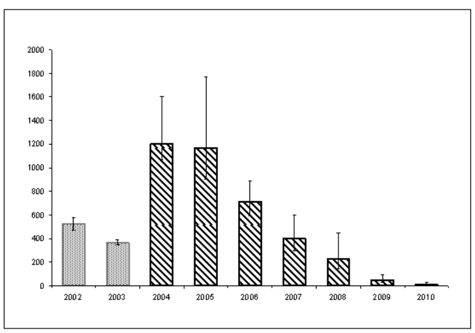

Figure 2 – *Estimated coho salmon escapement based on mark-recapture (slash) and one redd per female (grey) methods with 95 percent confidence intervals (WY 2002 to 2010)*

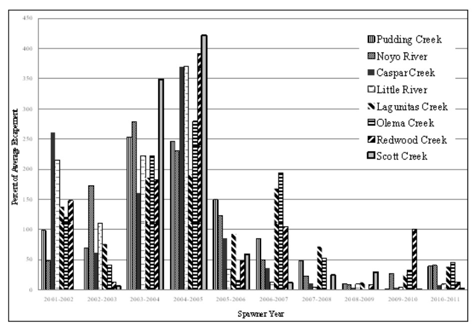

Figure 3 – *CCC ESU coho salmon escapement estimates over the past 10 years. Percent of average escapement is derived from the average of escapement estimates for each watershed over the past 10 years of data (WY 2002 to 2011)*

Outmigrant production

Coho smolts were more abundant in 2006 than in all other monitoring years with a population estimate of approximately 24,000 fish. From 2007 to 2010, estimates varied from approximately 13,000 fish to 16,000 *(fig. 4).* When escapement estimates are plotted with the corresponding natal lineage of smolt abundance (*fig. 5*), smolt abundance does not appear closely correlated with the abundance of the parental brood-stock.

[6] Reichmuth, Fishery Biologist, Inventory and Monitoring Program, Point Reyes National Seashore. Unpublished.

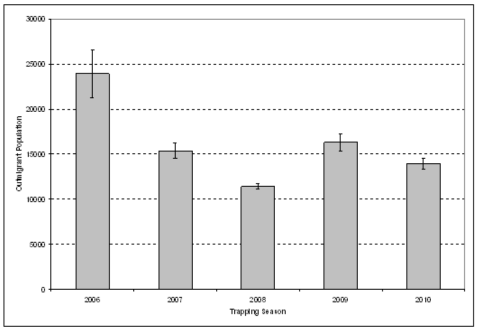

Figure 4 – *Coho salmon outmigrant production point estimates for Pudding Creek, California (WY 2006 to 2010).*

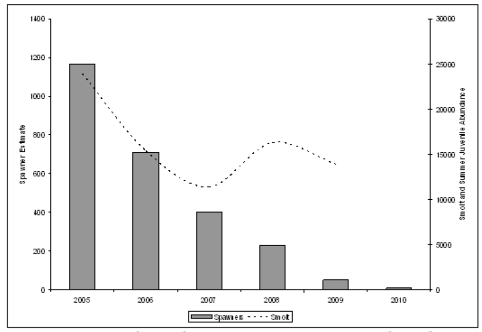

Figure 5 – *Coho salmon spawner estimated and the corresponding outmigrant estimate for Pudding Creek, California (WY 2006 to 2010).*

Juvenile growth

The recapture of PIT tagged juvenile salmonids produced information on seasonal instream growth. Growth seasons were defined by the period between sampling regimes, or by the period in which the sampling regimes occurred. Over five years of sampling (2006 to 2010), coho salmon growth in Pudding Creek appeared greatest in spring (February - May. Fall/winter (October - March) produced moderate growth, and summer (May - September) appeared to be a period of limited growth (*fig. 6*). In some years, growth was negative over the summer period.

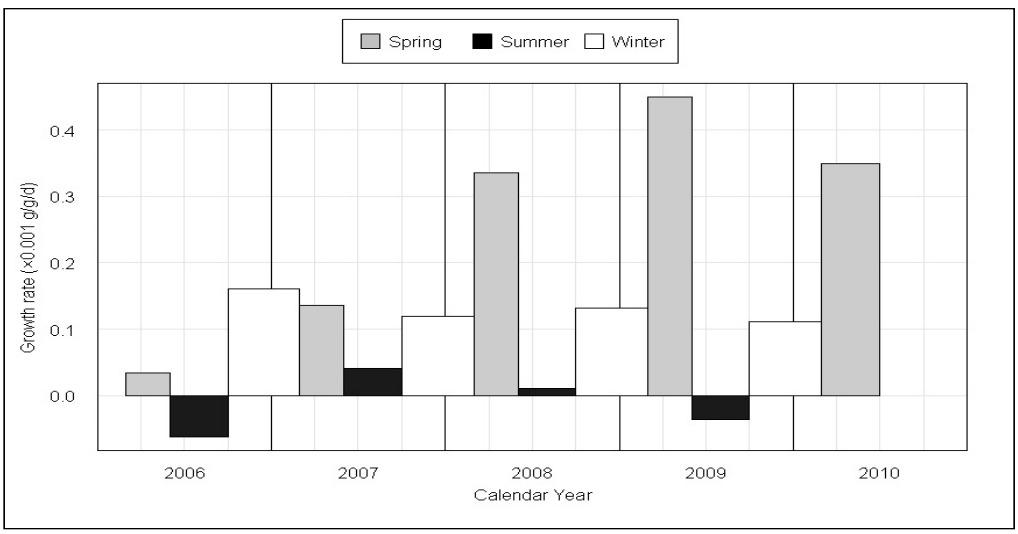

Figure 6—Growth rates of Coho salmon (*Oncorhynchus kisutch*) in Pudding Creek. Fit of individual growth data from PIT recoveries to fourteen parameter model.

Based on smolt PIT tagged at the RST and recaptured at the dam, in 2009 and 2010 significant growth occurred in the 6.1 km reach during outmigration. Respectively, mean coho smolt growth 10.6 mm (n = 268, s.e. = 0.62) and 11.8 mm (n = 41, s.e. = 1.62).

Coho salmon ocean survival

We examined six coho salmon cohorts (brood years) to estimate *spawner to spawner ratios* (2002 to 2010) – the ratio of the number of spawners to the number of their progeny that return as spawners. Two cohorts had ratios greater than 1 (more progeny returned), and four had a ratio less than 1, suggesting poor yield from the spawning efforts after the 2003 water year (*table 1*). Outmigrant sampling combined with adult recaptures of the same cohort facilitates estimation of specific marine survival (*smolt to spawner estimates*) by two methods: Comparison of estimates between outmigrant and escapement indicates that ocean survival from 2006 to 2010 ranged from 0.08 to 0.95 percent (*table 2*). Ocean survival based on PIT tag returns from these cohorts ranged from 0.04 to 0.44 percent. PIT tag return survival estimates remained consistently lower for all cohorts (*table 2*).

Table 1—*Adult escapement estimates and spawner-to-spawner ratios for six cohort classes of coho salmon.*

Years	2002 to 2005	2003 to 2006	2004 to 2007	2005 to 2008	2006 to 2009	2007 to 2010
Spawner-to-Spawner (95% CI)	2.23 (2.05-2.91)	1.93 (1.77–2.21)	0.33 (0.28–0.37)	0.2 (0.17-0.25)	0.07 (0.05-0.11)	0.02 (0.01-0.05)

Table 2—*Coho salmon ocean survival (smolt to spawner estimates), calculated from population estimates and tag returns.*

Ocean Years	2006 to 2008	2007 to 2009	2008 to 2010
Comparison of Population Estimates (Percent)	0.95	0.33	0.08
PIT Tag Returns (Percent)	0.44	0.14	0.04

Coho salmon over winter survival

The proportion of juveniles PIT tagged during summer abundance surveys and recaptured at the RST indicates instream survival from late summer to spring outmigration (*table 3*). Over winter survival varied from 0.17 to 0.8. However, due to the low number of tags recaptured, confidence intervals were large, in some cases indicating unrealistic upper bounds as in 2007 and 2010.

Table 3—*Over winter survival.*

Years	2007	2008	2009	2010
Over winter survival (95 percent CI)	0.71 (0.24-1.18)	0.17 (0.13–0.22)	0.61 (0.45–0.77)	0.8 (0.57-1.02)

Juvenile size at outmigration

The modal size class of coho salmon was 90 to 100 mm from 2006 to 2008. In 2009, the modal size class was smaller (80 to 90 mm) and the frequency distribution appeared to be wider. The 2010 modal size class was 100 to 105 mm, and the frequency distribution was relatively narrow (*fig. 7*). In 2006 and in 2008 a bimodal distribution was observed, with the secondary peak at approximately 140 mm. This pattern was also observed to a lesser degree in 2010. In 2007 and 2009, the bimodal distribution was not apparent. The secondary modes in the small size classes (< 60 mm) are newly emergent YOY.

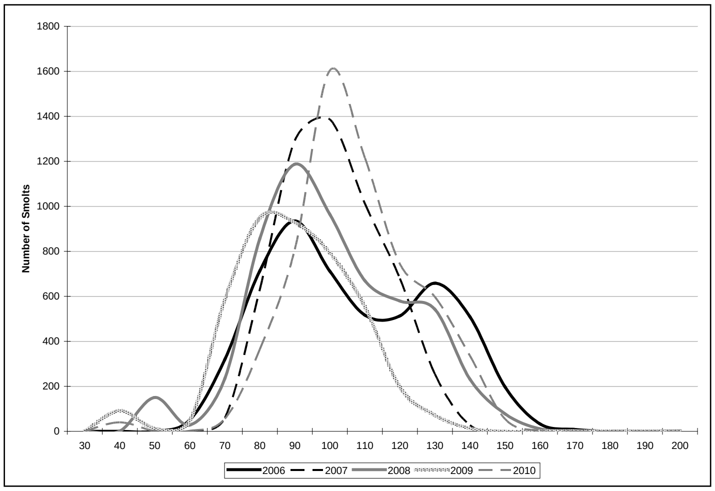

Figure 7—Smoothed histogram of coho salmon fork lengths at the outmigrant trapping station in Pudding Creek (2006 to 2010).

The mean value of annual juvenile cohort fork lengths since 2006 (*table 4*) from the RST captures indicate that the 2010 smolt were largest and the 2009 cohort were smallest.

Table 4—*Mean juvenile coho fork lengths with 95 percent confidence intervals for Pudding Creek* (*2006 to 2010*).

Year	2006	2007	2008	2009	2010
Mean Fork Length	101.9	96.5	88.1	87.7	104.2
(CI)	(0.5)	(0.4)	(0.5)	(0.5)	(0.5)

Coho juvenile two year life history

The use of PIT tags, the RST, and the PIT tag antennae array at the dam indicated evidence of a coho two year instream life history and several characteristics that appear to be associated with this behavior. Since 2009, 13 percent of all coho salmon detections at the dam array have been 2 year old juveniles. Since 2006 when downstream trapping was initiated at the RST, we have recaptured coho in their second year on multiple occasions (*fig. 8*). The recaptures at the RST also indicate that second year life history behavior is not annually consistent (*fig. 8*).

The median fork length for first year smolt (92 mm) at initial capture was greater than the group that outmigrated in their second year (76 mm). At recapture, the median value of second year smolts was greater (129 mm) than first year smolts (104 mm). A Mann-Whitney Rank Sum Test (not normally distributed) indicates a significant difference between the median values of the two groups at first capture ($P= <0.001$). A t-test of the normally distributed recaptures indicates a significant difference between the mean values of the groups ($P=<0.001$).

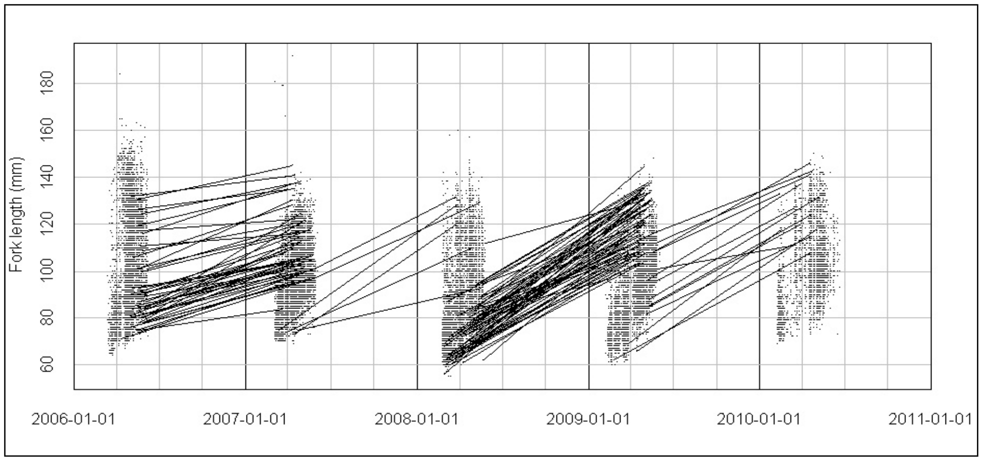

Figure 8—Length of individual Coho salmon at the Pudding Creek RST vs. time. The length of each fish measured in each spring survey is shown as a dot. Line segments join successive recoveries of the same individual PIT tag in spring surveys. All recaptures occurred at RST.

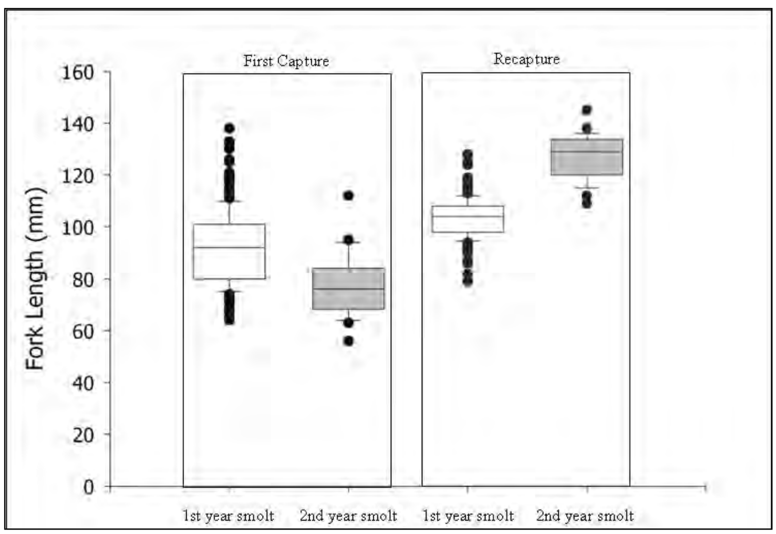

Figure 9—Fork lengths for first year smolts compared to second year smolts at first capture and recapture at the RST.

Discussion

Spawning abundance

The current analysis of escapement and outmigration for coho salmon suggest the Pudding Creek population is following the trend found in other streams throughout California (*fig. 3*), which further suggests the causes for the immediate decline may be at the regional scale.

Juvenile abundance

At the RST, coho salmon outmigrant numbers have remained consistent since 2007 after a significant decrease from 2006 (*table 1*). A comparison of coho salmon spawners to their outmigrant offspring the following spring suggested the outmigrant population was dependent on the number of spawners from 2006 to 2008, a pattern not reflected in 2009 and 2010 (*fig. 5*).

The outmigrant trap is approximately 6 km from the ocean confluence, with spawning, rearing, and overwintering habitat located between it and the ocean. Thus, the trap may not actually estimate the true outmigrant population. PIT tag recaptures in returning adult coho salmon suggest that the outmigrant population may be considerably higher than our stated estimates (*table 2*). However, this is contingent on whether chronic mortality of PIT tagged fish is significant. To test for chronic mortality, in 2010, 50 percent of the tagged fish were tagged with a right maxillary clip rather than a PIT tag. The tag rate of these two groups in adults returning during water year 2012 will indicate whether chronic mortality is an issue.

Over winter survival

Based on the spring recaptures at the RST of fish originally PIT tagged in summer, over winter survival in Pudding Creek appears to increase as juvenile abundance decreases (*table 4, fig. 4*). We believe this is the result of reduced density dependent competition among rearing juveniles for habitat and feed.

Marine survival

Coho salmon spawner-to-spawner and outmigrant-to-spawner ratios were low for the WY 2008, WY 2009 and WY 2010 runs (*tables 1 and 2*). A poor spawner-to-spawner estimate indicates low production at any point in the coho salmon life cycle; however, poor smolt to spawner estimates clearly point to low survival at sea (*table 3*). Our smolt to spawner estimates are much lower than observed in California coastal streams (e.g., greater than six percent in Waddell Creek [Shapovalov and Taft 1954]) and in many streams further north (greater than 10 percent in Puget Sound [Mathews and Buckely 1976]), However, our results are consistent recent rates of less than one percent in Oregon streams (Lewis 2004).

Size

By mean fork length, smolts in Pudding Creek were largest in 2010 (*table 5*) and smallest in 2009. The summer of 2009 was extremely dry, which may account for the diminished size. Although 2006 and 2010 smolts were comparatively large, they were small compared to those in Waddell Creek, CA (Shapovalov and Taft 1954). In 2009 and 2010, mean fork length increased by approximately 11 mm as fish migrated the final 6.1 km between the RST and the dam (near the ocean confluence). These additional 11 mm would make the Pudding Creek mean FLs as measured at the RST more comparable to those reported in the 1930s Waddell Creek study. However, the Waddell Creek weir was located 2.2 km from the ocean confluence. Waddell smolts probably also grew between trapping site and the ocean. We compared the length of Pudding Creek coho smolt captured between 2006 and 2008 to those captured in a similar monitoring project on Prairie Creek, Humboldt County, CA[7]. The main land-

[7] W. Duffy, California Cooperative Fisheries Research Unit, 2011, unpublished data.

use difference between these two otherwise similar watersheds is that Prairie Creek has not undergone significant timber harvesting. Pudding Creek, on the other hand, has been managed for timber harvest for over a century. Coho outmigrants from Pudding Creek (97.5 mm) are somewhat larger than those from Prairie Creek (96.8 mm; $p = 0.011$ for the two-sided heteroscedastic t-test). This information suggests that the characteristics of coho size at outmigration are variable by watershed. Coho forklengths were comparable to the size range reported by Bradford et al. (1997) throughout Oregon and Washington.

Growth

Coho salmon in Pudding Creek grow well in spring, and moderately in fall-winter. Summer appears to be a period of low growth, which is likely due to diminished carrying capacity stemming from the annual summer drought. These findings support the notion that stream enhancement projects intended to augment pool habitat (e.g., through the introduction of large wood into the channel) may be effective to increase juvenile salmon survival and growth over summer.

Based on smolt captures at the dam that were tagged at the RST, on average coho smolt grew approximately 11 mm while traveling the last 6.1 km of stream, which is nearly 10 percent of their total growth within the last 100 days of stream residency. This suggests the importance of lower channel habitat for juvenile growth.

Coho juvenile 2 year life history

A second year life history likely supports three important life history functions. First, it allows undersized fish with little potential for marine survival a second year to increase size. Second, it is one pathway for genetic mixing between the cohort classes. Finally, a two year life history can help ensure adequate recruitment when poor spawning occurs by contributing juveniles from one year to the next.

The mechanism for second year life history in juveniles appears to be, in part, the slower downstream movement exhibited by smaller smolts. We can determine the length of time smolt spend between the RST and the dam and compare this period to fork length measurements taken at the RST (*fig. 10*). The analysis of these data indicate that larger fish move rapidly through the stream network, while smaller fish take considerably more time in downstream migration. Considering that streamflow generally does not allow downstream migration in Pudding Creek past early June, which is approximately a 100 day window for migration, smaller fish will more likely be prevented from migrating to the ocean.

In Pudding Creek coho salmon that stayed a second year were significantly smaller than the rest of their cohort upon initial capture (*fig.* 9). We found the mean fork length at first capture of these fish to occur at approximately 75 mm, which is also the size class that takes longest to outmigrate (*fig. 10*). However, when they outmigrated the following year they were significantly larger than that year's cohort. This is consistent with the findings of Bell and Duffy on Prairie Creek, CA (2007).

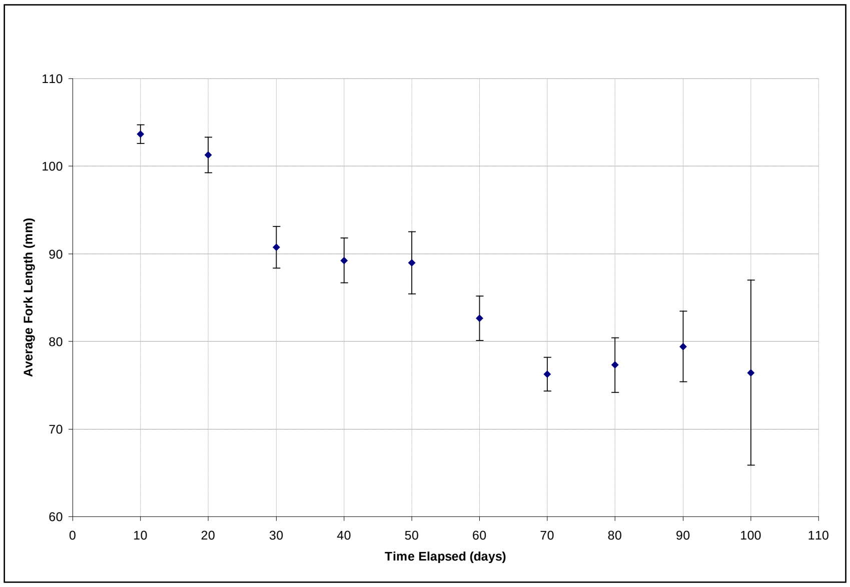

Figure 10—Time elapsed between original capture at RST and subsequent detection at dam vs. initial measured fork length. Error bars are 95 percent confidence intervals.

The dam and impoundment raise many issues regarding the affects on salmonids, and presently we do not understand the degree to which it benefits or hinders them. On one hand, the impoundment may benefit coho because it acts as over-wintering refugia for juveniles during high winter flows, and as summer rearing habitat. However, dam and impoundment may be detrimental due to low dissolved oxygen levels in the pond during summer from decaying non-native vegetation, fish passage issues, predation, and the division of the coastal lagoon. Decisions regarding the future of the dam structure will soon be necessary due to changing regional socio-economic conditions. More resources should be allotted to research on the dam as it relates to salmonids before these decisions are made.

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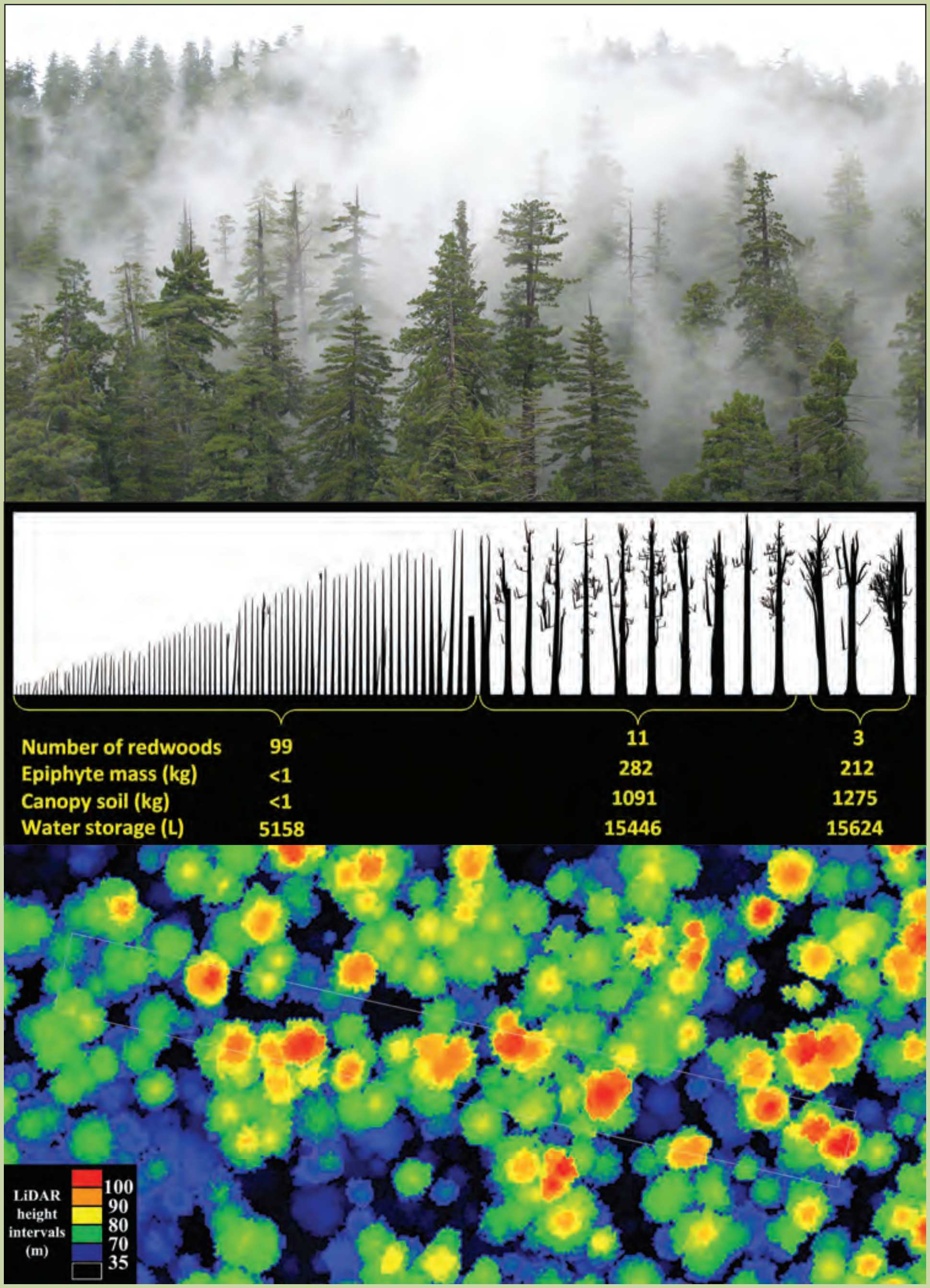
Silviculture and Restoration



Coast Redwood Live Crown and Sapwood Dynamics

John-Pascal Berrill,[1] Jesse L. Jeffress,[1] and Jessica M. Engle[1]

Abstract
Understanding crown rise and sapwood taper will help meet management objectives such as producing long branch-free boles for clear wood and old-growth restoration, or producing sawlogs with a high proportion of heartwood. Coast redwood (Sequoia sempervirens) tree crown ratio data were collected 20 years after partial harvesting in a 65-year-old second growth stand. Crown ratio correlated with stand density and relative tree height, and differed between stand interior and edge locations. The sum of tree crown lengths in each plot gave crown length/ac which exhibited an asymptotic relationship with stand density (i.e., crown length/ac peaks at higher densities). Maximum crown length/ac can be used to constrain simulations of crown rise by limiting the increase in crown length following thinning. Sapwood cross-sectional area was sampled at breast height and at the base of the live crown (BLC) of redwood trees growing in northern, central, and southern portions of redwood's natural range. Variability in sapwood area taper prevented detection of significant differences between regions, but on average sapwood tapered most rapidly among northern trees and least among southern trees. We present a robust sapwood taper model derived from the power function that has the desirable property of predicting zero taper for trees with BLC at breast height. Models predicting sapwood area at breast height from DBH allow prediction of sapwood from forest inventory data. Results indicated that dominant and codominant trees in the northern region had more sapwood than trees of similar size growing in the hotter, drier southern region.

Key words: crown ratio, crown rise, heartwood, leaf area, live crown ratio, sapwood taper, *Sequoia sempervirens*

Introduction

Contemporary management objectives for young coast redwood stands include old-growth restoration and clear wood and heartwood production. Distinguishing characteristics of old-growth forests include long branch-free boles. These can be re-created by artificial pruning, or by manipulating stand density to control crown rise. Crown height and size affect tree growth which largely comprises sapwood and heartwood development. Understanding the impacts of stand density on crown rise and its relationship with tree size, sapwood, and heartwood content permits us to design density management regimes for aesthetic and wood quality objectives. For example, it may be worthwhile to sacrifice tree diameter development to induce crown rise and produce valuable knot-free heartwood with narrower growth rings.

Crown rise in monospecific plantations of loblolly pine (*Pinus taeda*) and Sitka spruce (*Picea sitchensis*) has been predicted successfully using estimates of average tree height and either tree count per unit area or average inter-tree distance (Valentine

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et al. 1994). The same variables correlated with crown length in old-growth spruce-fir forests in British Columbia (Antos et al. 2010). Tree size and stand density measures, and topographic site factors were useful predictors of crown ratio throughout the forests of Austria, where crown ratio differed between species and was most affected by competition (Hasenauer and Monserud 1996). Norway spruce (*Picea abies*) crown ratio and crown length varied between age groups and were affected by thinning (Kantola and Mäkelä 2004).

Estimates of tree crown leaf area for individual trees and forest stands are useful predictors of tree growth and forest productivity in terms of tree growth efficiency (defined as the ratio of stem wood production to total leaf area), net primary productivity, and carbon sequestration. Growth efficiency can be used as a reference for a stand's susceptibility to disease or insect attack or for other trends in forest health (Maguire and Batista 1996). The relationship between redwood tree growth, growth efficiency, and tree leaf area is affected by stand structure and the tree crown's position within the stand (Berrill and O'Hara 2007a). Therefore redwood stand density and structure can be manipulated to affect leaf area, stand growth, and tree size at harvest (Berrill and O'Hara 2007b, 2009). The pipe model theory (Shinozaki and others 1964) proposes that a given amount of foliage is supported by a proportional cross-sectional area of 'pipes', i.e., a close functional relationship exists between sapwood area (SA) and leaf area (LA). Sapwood area at the base of the live crown (SA_{BLC}) is more strongly correlated with leaf area than sapwood area at breast height (SA_{BH}) (Waring et al. 1982). A taper model predicting SA_{BLC} from measured SA_{BH} will improve leaf area estimates (Waring et al. 1982). The existing sapwood taper model for redwood was constructed using data for second-growth trees of varying size in different canopy positions within stands in the central portion of redwood's natural range (Stancioiu and O'Hara 2005).

We examined crown ratio data for redwood trees in the central portion of redwood's natural range and sapwood area among redwood trees in the southern, central, and northern parts of the range. Our objectives were to:

(i) Develop models that predict redwood crown ratio in managed and unmanaged stands.

(ii) Construct an equation that predicts SA_{BLC} for the southern, central, and northern regions, and test for regional differences in sapwood taper.

(iii) Model SA_{BH} as a function of diameter at breast height (DBH) to enable prediction of SA_{BH} and SA_{BLC} from traditional forest inventory data.

Methods

Crown ratio data were collected in 234 permanent sample plots within the 260 ac Railroad Gulch Harvest Demonstration Area on Jackson Demonstration State Forest (JDSF), near Ft. Bragg in Mendocino County, north coastal California. After clearcutting in 1920, a mixture of redwood, Douglas-fir (*Pseudotsuga menziesii*) and tanoak (*Notholithocarpus densiflorus*) regenerated naturally, supplemented by planting of redwood seedlings. In 1982, tenth-acre plots were established on a grid pattern across the 260 ac demonstration area. Then, the area was divided into 14 blocks: individual tree selection was applied in four 20 ac treatment blocks, group selection applied in four 20 ac blocks, and a combination of both regeneration methods was applied in four 20 ac blocks. Two smaller blocks were reserved as ‘no treatment’ control areas. The plots have been measured three times at 10 year intervals. Tree measurements included DBH, total height, and height to BLC.

Sapwood data from the southern portion of redwood’s range were collected at five sites between Santa Clara County and Napa County. These were relatively hot and dry redwood sites, being located near the inland limits of redwood’s range (*fig. 1*). Sapwood data for the central region were obtained from a previous study on JDSF (Stancioiu and O’Hara 2005). Two study sites were chosen for sampling in the northern region: Arcata Community Forest and Freshwater Forest, near Humboldt Bay in Humboldt County. The northern sites are shrouded in fog for much of the growing season, and have an average mid-summer daytime high temperature of only 64 ^{0}F. Summer fog generally dissipates during the daytime at the central sites in Mendocino County, and average summer high temperatures exceed the value of 67 ^{0}F reported for the weather station at the coast. At all study sites, most rainfall occurs between November and March. Mean annual precipitation is approximately 40 in at the northern and central sites and at the inland Napa County site (Las Posadas State Forest). However, we categorized the Napa site as ‘southern’ because it receives less fog and its summer temperatures are much higher (86 ^{0}F in July), much like the other southern sites (76 to 88 ^{0}F). Annual precipitation averaged approximately 21 in at Gilroy near sampling sites at Bodfish Canyon and Mt Madonna in Santa Clara County and 26 in at Roberts and Redwood Regional Parks in Alameda and Contra Costa Counties, respectively (http://www.weather.com).

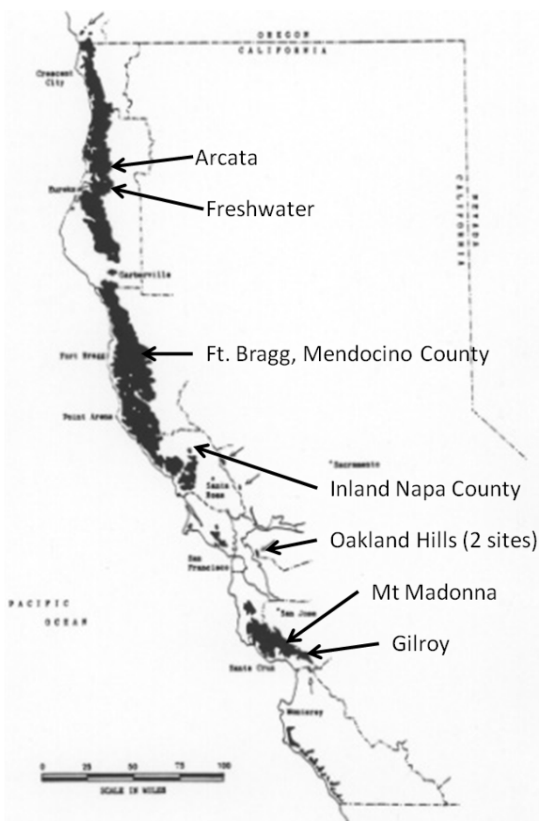

Figure 1—Natural range of coast redwood and study site locations for northern, central, and southern regions.

Dominant and co-dominant redwood trees approximately 80 to 130 years old and about 110 ft tall were selected for sapwood measurement in the northern and southern regions: 19 trees from the two northern sites and 18 trees from the five southern sites. Trees were climbed using a flip line and spurs. Total height and height to BLC were measured, where the lowest living vigorous branch defined BLC (whereby excluding epicormic branches found lower on the stem). Diameter at BLC and DBH were measured with a steel tape. Two increment cores were taken at both breast height and BLC. Bark thickness measurements were taken at the site of each increment core. The point of transition from translucent live sapwood to dry sapwood adjacent to the red-colored heartwood was marked on each core. Live sapwood radial bandwidths including the vascular cambium were measured using calipers. Data for 10 trees in the central region were obtained from a destructive sample of cross-sectional discs cut from felled trees (Stancioiu and O'Hara 2005).

Analysis of crown ratio involved separating tree size and crown data from the most recent measurement at Railroad Gulch into damage classes: top damaged, forked, and undamaged (no top damage or forking). Differences in crown ratio between damage classes justified exclusion of damaged trees. Tree data were also separated into two crown class groups: dominant and codominant trees, and intermediate and suppressed trees, and summarized for each group. Differences in crown ratio between groups justified separate analysis of crown class groups. Too few dominant tree data were available to model data for individual crown classes. Data for height to BLC, crown length, crown ratio, and height – diameter ratio were regressed against redwood stand density index (SDI). Estimates of redwood SDI were obtained using an SDI-adjustment procedure where a new plot area was calculated for the redwood component growing in mixture within a sample plot. The ratio of redwood component SDI to plot SDI gave an estimate of the proportion of total growing space occupied by redwood within each plot. This ratio was applied to the plot area, giving an estimate of area occupied by redwood. This 'SDI-adjusted' redwood plot area was used to calculate stand-level estimates of average tree size, basal area, and SDI for redwood. Generalized linear mixed models were constructed to account for nesting of trees within plots within harvest treatment blocks. The models were fitted to crown ratio data for individual redwood trees from each crown class group using PROC GLIMMIX in SAS statistical analysis software (SAS Institute 2004). The effects of topographic and species composition variables on crown ratio were tested. A dummy variable for stand edge versus interior locations, and relative height – the height of a tree relative to the crown class group average – were also tested as explanatory variables. Tree height and crown data from previous measurements at Railroad Gulch were not amenable to crown ratio modeling, but pre-harvest species composition and stand density estimates were included in the crown ratio analysis. Random effects in the mixed models were plot and treatment block. Crown length was summed in each plot, giving total redwood crown length/ac. Crown length/ac estimates were binned in increments of stand density (for example SDI 50 to 150, 150 to 250, and so forth), and an approximation of their upper limit (99^{th} percentile) regressed against SDI using PROC REG in SAS. This relationship was assumed to represent an approximate upper limit to average tree crown length for redwood in managed and unmanaged stands, for use constraining crown rise simulations within realistic limits.

Preparing data for sapwood analysis involved calculating heartwood diameter by subtracting the average live sapwood bandwidth from the total diameter inside bark

(DIB). Both DIB and heartwood diameter were converted to area. Sapwood cross-sectional area (SA) for breast height and BLC were determined by subtracting heartwood area from basal area inside bark. Sapwood taper equations were fitted to sapwood data from each region using the non-linear regression analysis procedure PROC NLIN in SAS. The taper models predict SA_{BLC} as a function of SA_{BH} and breast height crown ratio (BHCR, defined as crown length/ (total height – 4.5ft)) for each region, and for all regions combined. Equations that predict SA_{BH} as a function of tree basal area were also fitted to sapwood and tree size data from each region using the linear regression analysis procedure PROC REG in SAS.

Results

Crown rise

Of the 2,894 second-growth redwood trees within 234 plots at Railroad Gulch, 69 percent did not show signs of forking or top damage, and 41 percent were classified as either dominant or codominant. Forty-four percent of intermediate and suppressed trees were forked or exhibited top damage. Among dominants and codominants, forked trees had the highest average crown ratios. Trees with top damage had lower crown ratios than other trees in the intermediate and suppressed crown class group (*table 1*). Therefore, trees with forks or top damage were excluded from analyses of crown ratio. Redwood tree and crown size varied according to position within the canopy (*table 2*). On average, edge trees were larger, more tapered, and had longer crowns than trees growing inside the stand (*fig. 2*). Average crown ratio among dominant and codominant trees in each plot decreased with increasing crowding (SDI). Dominant and codominant trees in plots with higher redwood SDI had higher crown base heights and less taper, but maintained similar live crown lengths to redwood trees growing in less crowded conditions (*fig. 3*).

Table 1—*Damage summary: count and crown ratio of second-growth redwood trees classified as either dominant or codominant (D+C), or intermediate or suppressed (I+S), with and without forking or top damage, at Railroad Gulch, JDSF.*

	Number of trees			Crown ratio		
	D+C	**I+S**	**All trees**	**D+C**	**I+S**	**All trees**
Forked trees	40	52	92	0.48	0.35	0.41
Trees with top damage	129	732	861	0.42	0.30	0.32
Forked and top damage	156	754	910	0.44	0.30	0.33
No forking or top damage	1040	944	1984	0.43	0.36	0.39
All trees (incl. damaged)	1196	1698	2894	0.43	0.33	0.37

Table 2—*Tree size and crown summary data for second-growth redwood trees without forking or top damage at Railroad Gulch, JDSF. Standard deviations listed in parentheses beside means for each crown class.*

Crown class	n	DBH (in)	HT (ft)	H:D ratio[a]	BLC (ft)[b]	Crown ratio
Dominant	28	33.6 (9.2)	131.1 (18.1)	49.2 (09.9)	64.3 (14.3)	0.50 (0.11)
Codominant	1012	23.5 (5.9)	117.1 (20.6)	61.9 (11.9)	66.6 (16.0)	0.43 (0.11)
Intermediate	445	14.0 (4.8)	77.1 (23.1)	68.3 (16.0)	46.0 (17.5)	0.40 (0.15)
Suppressed	499	7.0 (4.2)	38.5 (17.7)	73.4 (23.5)	26.0 (12.3)	0.31 (0.15)

[a]H:D ratio = ratio of total height (ft) to diameter at breast height (ft).

[b]BLC = height to base of live crown.

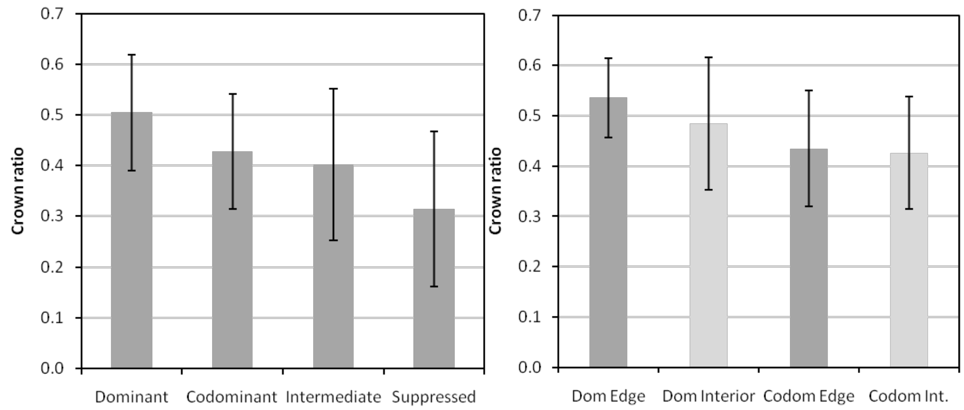

Figure 2—Average crown ratio for each crown class (left), and for dominant and codominant second-growth redwood trees growing on stand edges and inside the stand (right) at Railroad Gulch, JDSF. Error bars depict standard deviation for each group mean.

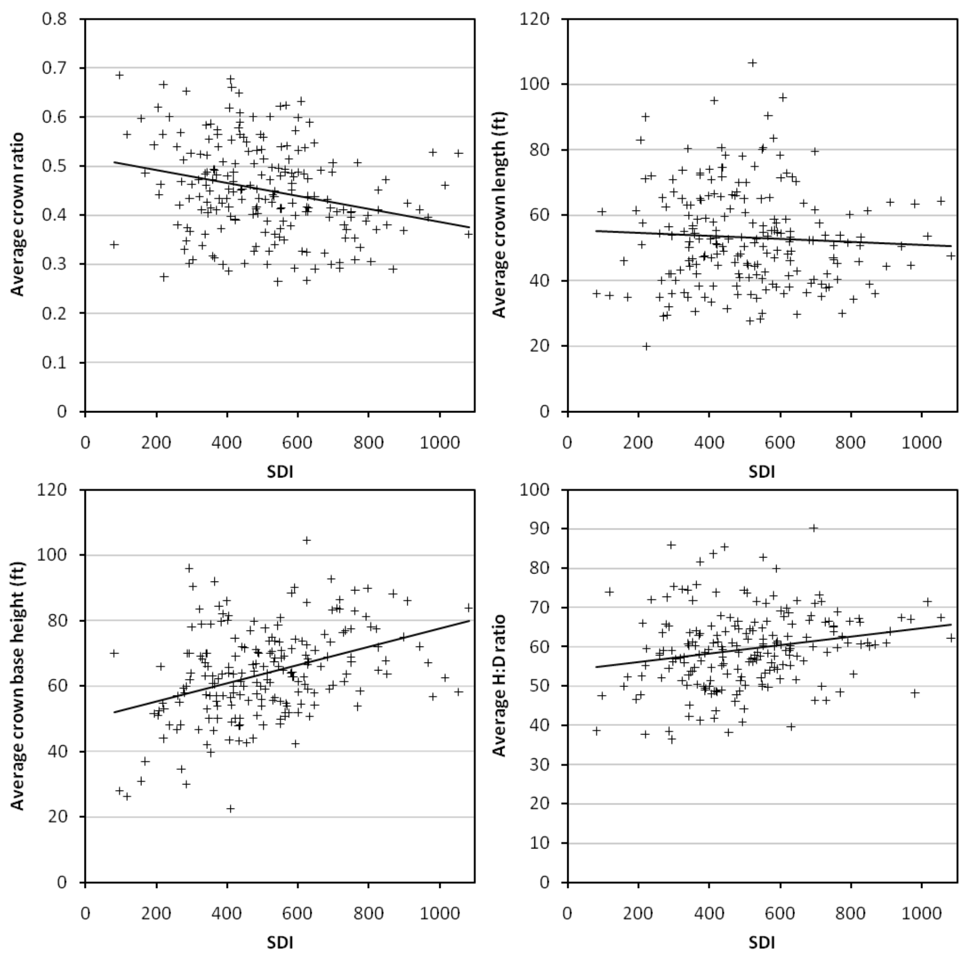

Figure 3—Average crown and stem taper data for dominant and codominant second-growth redwood trees and redwood stand density index (SDI) in 203 plots at Railroad Gulch, JDSF.

Species composition, aspect, and site index did not significantly affect redwood tree crown ratio. Dominant and codominant redwood crown ratios were slightly lower in plots with more Douglas-fir basal area, and slightly higher in plots with aspects tending towards northeast and in plots with higher site index, but these effects were not significant (Pr > t 0.49, 0.13, 0.46, respectively). Redwood crown ratio was unaffected by pre-harvest and current percent hardwood basal area and percent slope. Crown ratio was higher with greater relative height (height relative to average height of trees in each crown class group), and lower with increasing redwood SDI (*fig. 4*). The best crown ratio models in terms of AIC (goodness-of-fit criterion favoring models with fewer variables) included only tree location (edge or interior) and SDI (*table 3*). Model predictions indicated that tree location was a more important determinant of crown ratio among intermediate and suppressed trees than among dominants and codominants. The effects of relative height were important among trees in both crown class groups.

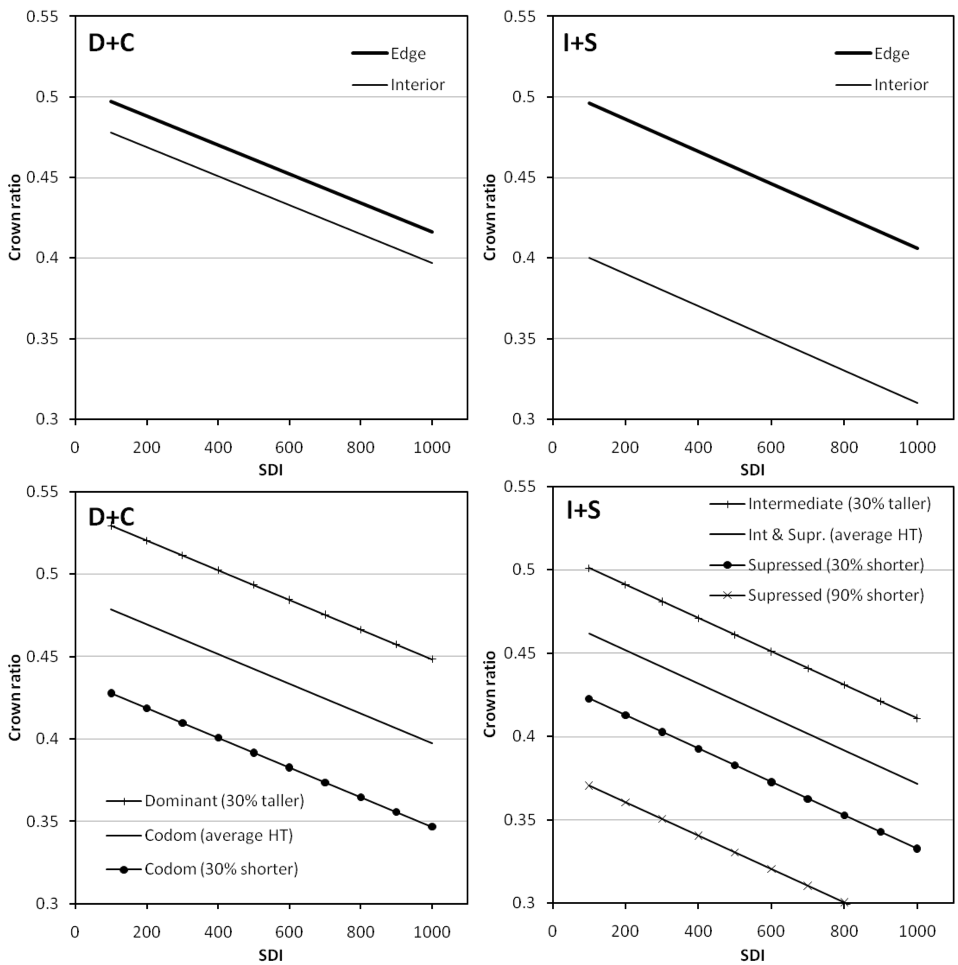

Figure 4—Generalized linear mixed model predictions of redwood crown ratio among dominant and codominant (D+C) trees and intermediate and suppressed trees (I+S) in stand edge and interior locations (above), and for interior trees with different relative heights (below), at Railroad Gulch, JDSF.

The approximate upper limit of the per-acre sum of redwood tree crown lengths in each plot attained 13,000 ft/ac at SDI 1000. The relationship was best described by a second-order polynomial equation (*fig. 5*).

Table 3—*Generalized linear mixed effects models of redwood crown ratio as a function of tree location (stand edge, interior) and stand density index (SDI), with/without relative height (above/below), for dominant and codominant (D+C) and intermediate and suppressed (I+S) crown class groups at Railroad Gulch. Dummy variable for edge location is added to intercept; interior location is latent variable (intercept only).*

Data	Model fit	Parameter	Coefficient	s.e.	d.f.	t	Pr>|t|
D+C	χ^2 = 10.19	Intercept (β_0)	0.31800	0.03426	13	9.28	<.0001
(n=1028)	AIC=-1709	Edge (d_1)	0.01820	0.00926	987	1.96	0.0497
		Relative height (β_1)	0.16960	0.02980	987	5.69	<.0001
		SDI (β_2)	-0.00009	0.00002	987	-4.47	<.0001
I+S	χ^2 = 15.15	Intercept (β_0)	0.34130	0.02655	13	12.86	<.0001
(n=900)	AIC=-1011	Edge (d_1)	0.08437	0.01452	859	5.81	<.0001
		Relative height (β_1)	0.13050	0.01857	859	7.03	<.0001
		SDI (β_2)	-0.00010	0.00003	859	-3.35	0.0008
D+C	χ^2 = 10.52	Intercept (β_0)	0.48700	0.01721	13	28.29	<.0001
(n=1028)	AIC=-1682	Edge (d_1)	0.01917	0.00940	988	2.04	0.0417
		SDI (β_1)	-0.00009	0.00002	988	-4.38	<.0001
I+S	χ^2 = 16.02	Intercept (β_0)	0.41040	0.02498	13	16.43	<.0001
(n=900)	AIC=-969	Edge (d_1)	0.09567	0.01482	860	6.46	<.0001
		SDI (β_1)	-0.00010	0.00003	860	-3.59	0.0004

Relative height = tree height/average height for crown class group.
χ^2 = generalized chi-square for model.

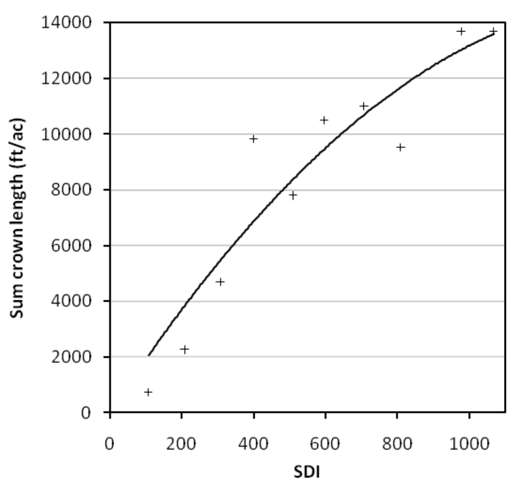

Figure 5—Relationship between 99th percentile of crown length/ac in plots grouped by SDI and the average SDI for each group, where crown length/ac = 19.832 *SDI* - 0.0067 *SDI*^{2}.

Sapwood taper

Northern and southern sample trees were larger than trees from the central region where a wide range of tree sizes were sampled across all crown classes (*table 4*). Different taper model coefficients for region-specific models indicated that, on average, northern trees exhibited greater sapwood taper between breast height and

BLC (*table 5*, *fig. 6*). However, variability in sapwood taper within regions prevented detection of significant differences in taper between the different regions, justifying a combined model for the entire redwood range. When re-arranged to predict SA_{BLC} as a function of SA_{BH}, the non-linear power model gave a mean prediction error of 4.4 in^2 SA_{BLC} for n = 47 trees from northern, central, and southern regions.

Table 4—*Summary data for redwood sapwood sample trees by region.*

Region	Statistics	DBH (in)	Height (ft)	BHCR[a]	SA_{BH} (in^2)	SA_{BLC} (in^2)
Northern	Mean	31.1	115.8	0.73	179.5	122.7
(n=19)	s.d.	7.5	24.6	0.11	69.4	47.8
	Max.	51.4	158.8	0.93	304.6	204.2
	Min.	19.4	75.1	0.49	84.0	54.7
Central	Mean	13.6	78.1	0.68	59.3	41.2
(n=10)	s.d.	10.3	42.6	0.13	73.3	52.9
	Max.	36.5	148.3	0.84	248.3	179.5
	Min.	3.7	27.9	0.50	8.3	6.9
Southern	Mean	29.1	111.8	0.56	113.4	69.3
(n=18)	s.d.	6.9	21.6	0.12	45.4	34.1
	Max.	43.4	163.0	0.76	215.4	137.0
	Min.	18.5	71.5	0.37	52.7	27.2

[a]BHCR = breast height crown ratio; crown length/(total height - 4.5 ft).

Table 5—*Summary statistics for sapwood taper model* $SA_{BLC} = SA_{BH} * (1-(1-BHCR)^{a})$.

Location	Coefficient	s.e.	Appr. 95% confidence limits	
			Lower	Upper
Southern	1.1549	0.1123	0.9180	1.3917
Central	1.0776	0.0771	0.9031	1.2521
Northern	0.9020	0.0737	0.7471	1.0569
Combined	1.0362	0.0565	0.9224	1.1499

Regional coefficients for linear models fitted to tree basal area and SA_{BH} data were separated by greater than two standard errors (north: β_1 = 0.20611 s.e. = 0.015; south: β_1 = 0.15058 s.e. = 0.012), indicating that, on average, northern trees contained relatively more sapwood than southern trees for a given DBH. The models for northern and southern trees were compared against the model presented by Berrill and O'Hara (2007a) for 891 redwood trees >4 in DBH growing in all canopy positions within even-aged and multiaged stands on JDSF. Predicted sapwood at breast height for dominant and codominant sample trees from the northern and southern regions exceeded the published SA_{BH} – BA model predictions (*fig. 6*).

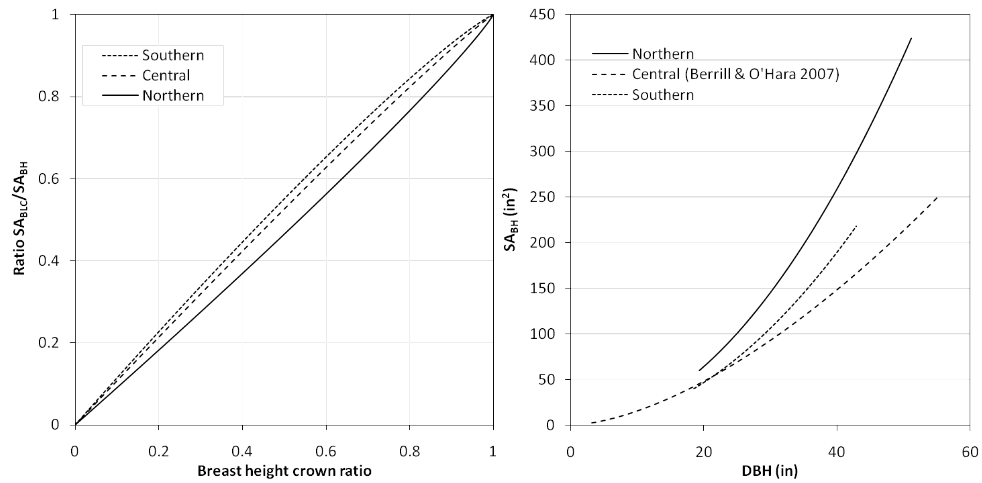

Figure 6—Sapwood taper models for northern, central, and southern regions SA_{BLC}/SA_{BH} = 1-(1-$BHCR$)^{a} (left), and SA_{BH} – DBH relations for northern and southern regions SA_{BH}=0.155 a π (1.27 DBH)^{2} and existing model for the central region (Berrill and O'Hara 2007a) (right).

Discussion

Greater incidence of top damage sustained by relatively short redwood trees at Railroad Gulch reminds us that care must be taken to protect residual trees and advanced regeneration during partial harvesting. Damage rates reported in *table 1* include damage from natural causes and harvesting, and could be used to estimate number of replacement trees required in younger cohorts under multiaged management. Retaining more replacement trees in younger cohorts leaves less growing space for other trees within multiaged stand structures (Berrill and O'Hara 2009).

The crown ratio models apply to redwood trees in the oldest cohort of uncut and partially cut second-growth stands around age 85 years at Railroad Gulch. Plots with lower SDI had total crown lengths/ac below the curve for maximum crown length (*fig. 5*) suggesting that maximum crown length had not been attained 20 years after partial harvesting. Therefore the crown ratio model slope coefficients for SDI may be too low, and should be revised after future re-measurements. The models can be applied to simulate crown rise in managed stands using a bi-conditional approach where crowns do not rise until sometime after partial harvesting when crown length/ac approaches the upper limit (*fig. 5*), after which time crown ratio can be predicted from SDI (*table 3*). The relationships between stand density and tree taper, crown length and height, and crown ratio among dominant and codominant trees (*fig.* 3) may be explained by 'plasticity' in redwood physiology. Trees in crowded plots may have responded with increased allocation of energy to height growth, at the expense of diameter growth, resulting in reduced taper. While tempting to infer that redwood trees can maintain a long live crown under crowded conditions, it is more likely that winners and losers had emerged; that loss of crown length resulting from crowding (*fig. 4*) eventually led less competitive trees to relinquish their dominance within the canopy.

Redwoods regulate water use poorly (Burgess and Dawson 2004). We found that, on average, southern trees in hotter, drier climates had less SA_{BH} for a given tree size but a lower rate of sapwood taper above breast height (*fig. 6*). These differences may reflect on adaptation to conserve water resources in drier climates by restricting sap flow. If the current trend of decreasing fog (Johnstone and Dawson 2010) continues, central and northern redwood trees may respond to increased evaporative demand by altering sapwood dimensions and crown morphology. Our sapwood taper sampling controlled for tree size and crown class between northern and southern regions, focusing on large dominant and codominant redwood. Faster growth in the north meant that sample trees were younger than trees of equivalent size in the south. Apparent differences in sapwood taper between regions (*fig. 6*) could be age-related. A large sample would be needed to examine effects of tree age, size, and position within the stand.

We present a non-linear power model of sapwood taper that provides a more realistic depiction of sapwood taper than the existing model, $SA_{BLC} = 0.7460SA_{BH} - 3.8293(H_{BLC} - 1.37)$ (Stancioiu and O'Hara 2005). The published model predicts negative values for SA_{BLC} for trees with small SA_{BH}, while our model makes realistic predictions for trees of all SA_{BH} and BHCR. Predicting SA_{BLC} as a function of BHCR in the non-linear model has the desirable property that when BHCR =1, the BLC is at breast height and, therefore, SA_{BLC}=SA_{BH}, making negative predictions for SA_{BLC} impossible. We also developed models to facilitate implementation of the taper model by predicting SA_{BH} from DBH (*fig.* 6), but these only apply to large dominant and codominant trees (*table 4*), whereas the model presented by Berrill and O'Hara (2007a) applies to a wider range of tree sizes and growing conditions. When applied in combination, crown ratio and sapwood models can be used to predict SA_{BLC} from traditional forest inventory data. First, SA_{BH} can be predicted from DBH. Then, the sapwood taper model can be applied to the estimate of SA_{BH}, and BHCR calculated from tree height and BLC data or crown ratio model predictions, giving predicted SA_{BLC}, which in turn allows for prediction of tree leaf area and stand LAI.

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Influence of Tree Spatial Pattern and Sample Plot Type and Size on Inventory Estimates for Leaf Area Index, Stocking, and Tree Size Parameters

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Abstract

Sampling with different plot types and sizes was simulated using tree location maps and data collected in three even-aged coast redwood (*Sequoia sempervirens*) stands selected to represent uniform, random, and clumped spatial patterns of tree locations. Fixed-radius circular plots, belt transects, and variable-radius plots were installed by simulation. Bootstrap sample means, coefficient of variation (CV), and 95 percent confidence intervals were calculated for sample estimates of eight important stand parameters. Percent CV models depicted sample precision and enable calculation of minimum sample size for forest inventories. Precision differed between stand parameters e.g., dominant height and mean top height estimates were most precise; in many cases four times as precise as density estimates. Precision was affected more by spatial pattern than plot type, and generally ranked: uniform > random >> clumped. Density, average diameter, and average height estimate precision was especially sensitive to spatial pattern, and generally poorer in variable-radius plots. However, variable-radius plots generally produced the most precise estimates of basal area, volume, and leaf area index. Quantitative descriptions of sample stand structure and spatial pattern provide a basis for comparison or application of results to other forests with similar characteristics.

Key words: bootstrap confidence intervals, coefficient of variation, precision, Ripley's K, sampling simulation, *Sequoia sempervirens*

Introduction

Forest inventory data are increasingly being used for objectives other than estimating wood product yields. Objectives include assessment of ecosystem structure and function, carbon storage and sequestration, and "ground truthing" remotely sensed data. Users of forest inventory data are often interested in a variety of parameters (e.g., stem density, stocking, leaf area index) and demand different levels of accuracy and precision of sample estimates. Factors such as forest community composition, size, and structure influence sample design when the objective is to obtain an adequate sample with minimal sampling effort. Comparing estimates of precision across a wide range of variability in tree size or spatial patterns in real stands shows how each variable affects precision. The influence of sample plot type and size, and spatial pattern of tree locations on the accuracy and precision of key stand parameter estimates has not been examined in regenerating stands of coast redwood (*Sequoia sempervirens*) in north coastal California.

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Redwood is shade tolerant and long lived. Clumping of stump sprout regeneration around cut stumps results in aggregated spatial patterns. As a consequence of these and other factors, regenerating redwood stands can have wide tree-size distributions and complex spatial arrangements of trees. Since spatial pattern is expected to affect precision of estimates obtained from sample plots (Lessard et al. 2002, Martin 1983), it may be advantageous to account for spatial pattern when sampling by altering plot size or the number of plots, or by stratifying sampling according to differences in spatial pattern. However, less is known about how much or how differently spatial pattern can affect inventory estimates for different parameters obtained from different plot types and plot sizes.

Sampling efficiency in forest types other than redwood has been examined extensively, but only for a restricted number of comparisons e.g., between plot types or stand parameters. When viewed collectively, patterns emerge from these results: Fixed-radius circular plots generally yielded the most accurate estimates of stand density and were more time-efficient than other fixed-area methods. Variable-radius plots were most time-efficient, but tended to underestimate density. Belt transects were least time-efficient (Sparks et al. 2002). Bormann (1953) tested different fixed-area plot shapes, and found that longer belt transects produced more precise estimates than shorter rectangular plots of the same area when sampling across changing contour, soil, or vegetation, and when sampling sporadic species. Lessard et al. (2002) showed that, when sampling an average of *n* trees per plot, density estimates from fixed-radius plots were more precise than estimates obtained from *n*-tree distance sampling. However, fixed-area sampling was generally more time-consuming than distance sampling methods. Kint et al. (2004) found that distance sampling was generally more efficient than plot sampling for estimation of structural indices (i.e., nearest-neighbor spatial statistics; species distribution patterns; tree-size differentiation) in forest stands. 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). Sampling can be more efficient when more numerous smaller plots are installed, however below a certain plot size, estimates are prone to bias and many more plots are required to achieve a given level of precision (Lynch 2003).

The objective of our study was to examine how sample estimates for eight important stand parameters were affected by sample plot type and size, and the spatial pattern of tree locations. Inventory estimates for traditional forest stand parameters (basal area, volume, density, average tree diameter and height, and dominant height), mean top height (an alternate height metric), and leaf area index were obtained by simulating the installation of three different plot types – each having a wide range of plot sizes tested - in three redwood stands with different spatial patterns. Leaf area index (LAI) was examined because it is emerging as a key variable in forest ecosystem research and modeling (Asner et al. 2003, Berrill and O'Hara 2007) and management (O'Hara 1998). Dominant height and mean top height were examined because of their role in site index determination. Site index estimates are 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). Larger trees are selected based on conventions such as 'top height' or 'dominant height'. Top height is the average height of the *n* largest diameter trees per unit area. Theoretically this metric is not affected by thinning methods that target smaller trees for removal. Dominant height is the average height of dominant trees, or dominant and codominant trees, or trees above a certain proportion of the dbh distribution. It can be sensitive to changes in density when based on proportions of the dbh distribution. Sampling simulation results for these eight stand parameters were obtained in stands with similar structural characteristics but different spatial patterns (i.e., uniform, random, or clumped) to examine effects of spatial pattern on sample estimates.

Methods

Redwood-dominated stands were sampled on Jackson Demonstration State Forest (JDSF) near Ft. Bragg, Mendocino County, California. Redwood stands on JDSF regenerated naturally, occasionally supplemented by planted seedlings, after the old growth forest was progressively removed mainly by diameter-limit cutting or commercial clearcutting between 1880 and 1950. In three stands, a well-stocked area dominated by redwood was selected for measurement. A large rectangular sample block was established: (i) in a dense stand located on an upper slope (0.2 ha); (ii) in a less dense stand on a lower slope (0.5 ha), and; (iii) in a stand comprising widely-spaced clumps of redwood trees located on an alluvial flat (0.4 ha). All trees >10 cm diameter at 1.37 m breast height (dbh) were mapped, and measured for dbh, total height, and height to live crown base – defined as stem height above which the tree crown was generally continuous on one or more sides. Two breast height increment cores gave sapwood data for tree leaf area estimation. Pith-to-bark cores collected from dominant trees gave approximate breast height age of the sample stands.

Tree leaf area was predicted for each tree using tree-size and sapwood cross-sectional area data and equations presented by Berrill (2008) and Stancioiu and O'Hara (2005). Tree volume was estimated from dbh and height (Wensel and Krumland 1983). Tree data, and volume and leaf area estimates were summarized for each large rectangular sample block. The spatial pattern of tree locations in each block was characterized using the Ripley's K-function (Ripley 1981). Since a uniform pattern was not detected in any block, a low thinning was simulated in the largest block with the objective of reducing density in sprout clumps to create a more uniform pattern and concurrently reducing tree-size variability (to better align it with variability in the other two blocks).

Sampling within the three large rectangular blocks (each representing either uniform, random, or clumped patterns) was simulated for three plot types across a range of plot sizes. Circular fixed-radius plots, variable-radius plots, and fixed-width transect plot types were tested. Toroidal edge correction was applied to each large rectangular block (Haase 1995). Thirty randomly located plot centers were generated using the S-Plus Spatial Module (Insightful Corp. 2005) for sampling within each block. Trees were classified as 'inside plot' when their stem center point fell within the boundaries of fixed-area circular or transect plots or within the critical radius R_i for variable-radius sample plots. Trees inside plot boundaries were identified by calculating an inter-point distance between plot center and each tree location defined by pairs of x and y coordinates and comparing the inter-point distance against the defined plot radius or critical radius.

Circular fixed-radius plots of areas 0.005, 0.01, 0.015…0.1 ha were established at each plot center point. Randomly located plot centers defined the easting (the x of an x,y pair of location coordinates) for the centerline of north-south fixed-area belt transects running the entire length (same x, vary y approx. 50 m) of each large block. The transect width was varied to create rectangular belt transects with sample areas of 0.005, 0.01, 0.015…0.1 ha. Tree size and leaf area data for trees in each fixed-area circular and transect plot were summarized to the stand-level giving basal area (BA), volume, LAI, density, arithmetic average dbh and average height. Dominant height was calculated as average height of trees with dbh above the 75th percentile of the dbh distribution in each sample plot. Mean top height was average height of the largest 100 stems/ha in terms of dbh in each sample plot.

Variable-radius plots were established on each randomly located plot center. A range of BA factors (BAF) were applied. Basal area per hectare was obtained by multiplying the count of sample trees by the BAF. The tree factor gave the number of trees per hectare represented by each sample tree (Husch et al. 2003). Stand density was the sum of tree factors for sample trees. Volume per hectare was calculated from tree volume and the tree factor for all sample trees. Leaf area index (LAI) was calculated from tree leaf area LA_i and the tree factor TF_i for all sample trees, such that: $LAI = \sum(LA_i TF_i)/10000$. The weighted average height of trees in the upper 25 percent of the diameter distribution, weighting by the tree factor, gave dominant height. The approximate diameter distribution was obtained by replicating each sample tree dbh a total of TF_i times. Dominant trees had a dbh greater than the 75th percentile of the replicated distribution. An approximate mean top height was obtained by taking the TF_i-weighted average height of trees representing the largest 100 stems/ha.

For each plot type, re-sampling with replacement from the summary data for each plot size gave bootstrapped means, variances, and 95 percent confidence intervals for each stand parameter (Crawley 2002). To enable application of our findings in stands with different densities (having different number of trees per plot for a given plot size), results were presented in terms of average number of trees per plot instead of plot size, and bootstrap means and confidence intervals were presented as percent deviations from the population summary data for the large blocks. Results were summarized to give the minimum average number of trees per plot needed to obtain 95 percent confidence intervals for sample estimates within ±5 percent or ±10 percent deviation from the estimate. Percent deviations of the bootstrap sample means and confidence intervals from the population summary data were plotted against the average number of trees at each plot size or level of BAF, depicting sample mean estimates and precision. Coefficient of variation of the mean (CV) – the ratio of bootstrap sample standard deviation to the sample mean – was also calculated in percent terms to enable direct comparison between stands with different spatial patterns that each had different stand parameters. To facilitate interpretation and future implementation of results, percent CV (Y) was regressed against average number of trees per plot (X) for each plot type sampling each spatial pattern. Five equations were tested: Power: $Y = aX^b$ Logarithmic: $Y = a + b\ln(X)$ Exponential: $Y = ae^{bX}$ Type III exponential: $Y = ae^{b/X}$ and Schumacher: $Y = ae^{-bX^c}$ (Schumacher 1939).

Fitted models were compared in terms of goodness-of-fit (AIC) and residual bias. Percent CV predicted using the best model was plotted against average number of trees per plot for each plot type and spatial pattern, depicting differences in sample precision. Data were analyzed using S-Plus statistical analysis software (Insightful Corp. 2005).

Results

The three sample stands had relatively similar stocking in terms of BA and LAI, but different densities, average tree diameter, and total standing volume. Low thinning simulated in Block 1 removed stems ≤ 40 cm dbh, all Douglas-fir, and reduced density in redwood sprout clumps. Average tree size increased from 55 cm to 78 cm dbh. Dominant height, calculated as the average height of trees in the top 25 percent of the dbh distribution, increased from 54.4 m to 58.0 m as a result of low thinning. Mean top height, the average height of the 100 largest dbh trees per ha, was less affected by thinning. It only decreased from 56.2 m to 55.4 m. The resulting pure redwood stand had a uniform spatial pattern at scales up to 7 m. Thinning to create a uniform pattern reduced density by 65 percent and BA by 40 percent in the lower slope stand, and the range and variability of tree sizes became comparable to the stands on the upper slope and alluvial flat with random and clumped spatial patterns respectively (*table 1*).

Table 1—*Summary data for three sample blocks on Jackson Demonstration State Forest.*

Stand parameter/descriptors	Block 1[1]	Block 2	Block 3
Topography	Lower slope	Upper slope	Alluvial flat
Sample block area (ha)	0.5	0.2	0.4
Species >10 cm dbh	Redwood	Redwood	Redwood
Approximate age (years)	100	85	85
Number of trees sampled	90	230	292
Density (stems/ha)	180	1150	730
Average dbh (cm)	78.4	39.8	45.8
Standard deviation dbh (cm)	21.1	19.9	23.8
Minimum dbh (cm)	41.7	10.7	10.0
Maximum dbh (cm)	138.6	112.7	100.0
Standard deviation height (m)	7.96	10.7	13.7
Minimum height (m)	31.3	9.6	5.5
Maximum height (m)	63.8	50.1	53.3
Dominant height (m)	58.0	43.4	44.1
Mean top height (m)	55.4	44.5	45.7
Basal area (m^2/ha)	93.0	179.0	152.5
Volume (m^3/ha)	1401	1961	1684
Minimum tree leaf area (m^2)	88.7	0.5	0.3
Maximum tree leaf area (m^2)	771.7	616.8	596.6
Leaf area index (LAI m^2/m^2)	5.6	10.7	10.8
Stand density index (SDI)	442	957	758
Site Index – base age 50 yr (m)[2]	39.9	31.8	32.4
Spatial pattern[3]	Uniform	Random	Clumped
Scale of uniformity/clumping	0-7 m	-	0-12 m

[1]Data for Block 1 after simulated thinning in stand with minor Douglas-fir component, clumped spatial pattern < 7 m, 520 stems/ha, 55 cm av. dbh, 157.2 m^2/ha BA, 2223 m^3/ha volume, and LAI 9.9 m^2/m^2.
[2]Wensel and Krumland (1986).
[3]Spatial patterns described using Ripley's K with L-function transformation.

Stem maps and Ripley's K with L-function transformation depict spatial patterns in the three stands (*fig. 1*). The *L(d)* statistic for the upper-slope stand stayed within 'envelope' limits of complete spatial randomness despite the presence of a few multiple-stem clumps of redwood. Significant clumping was detected in the alluvial flat stand at scales < 12 m, while complete spatial randomness beyond 12 m suggested that widely spaced clumps were randomly distributed throughout the stand.

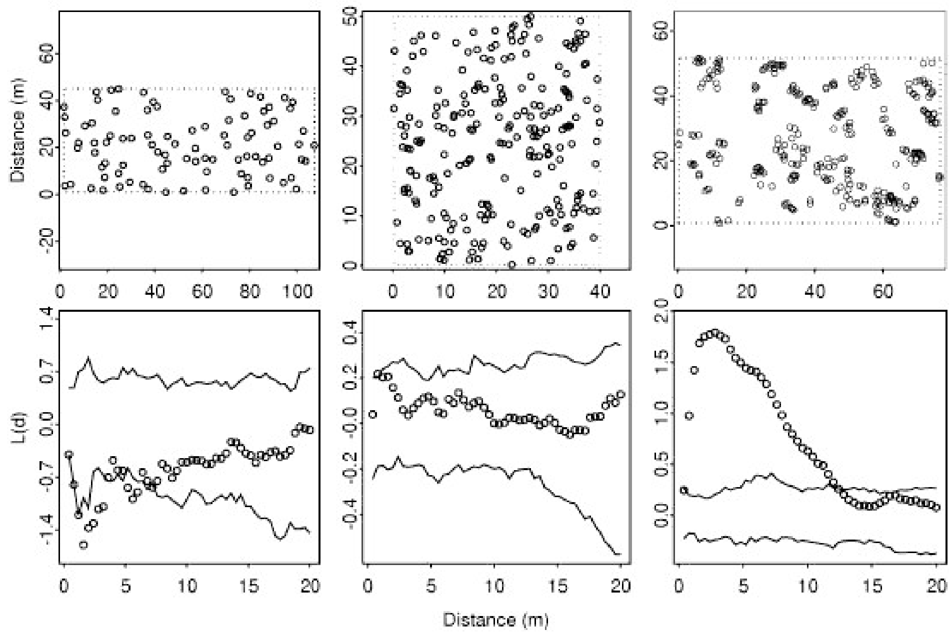
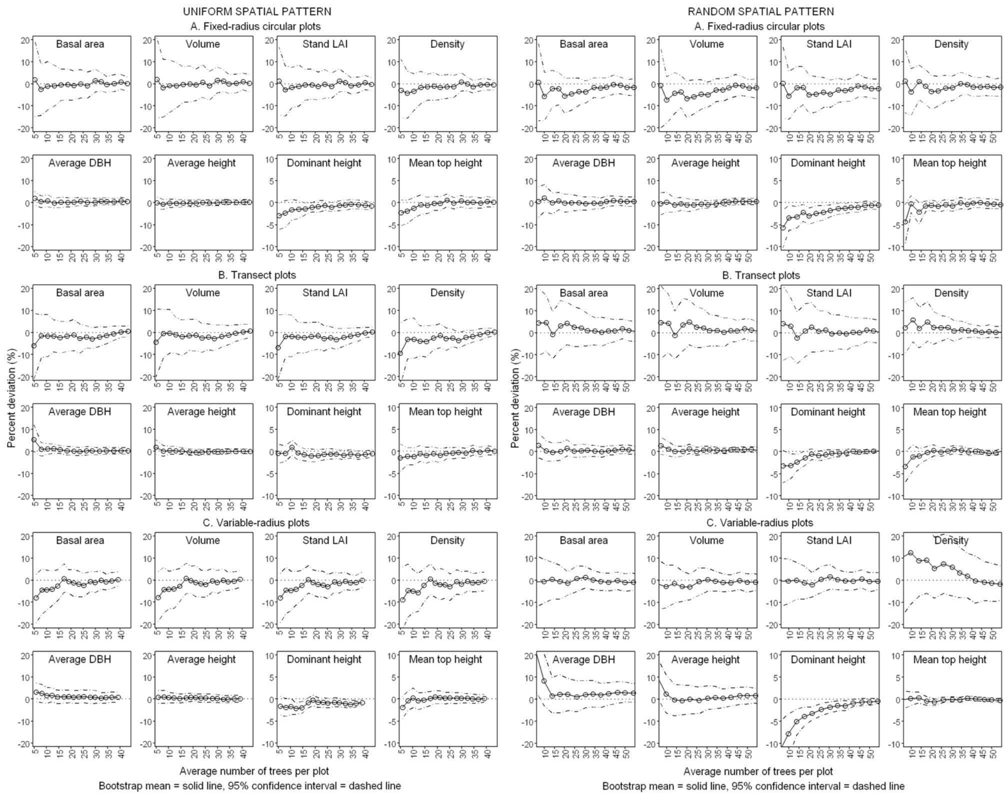


Figure 1—Stem maps and Ripley's K with L-function transformation for the lower slope (left), upper slope (center), and alluvial flat (right) stands. Circles represent tree locations in stem maps (above) and the L(d) statistic (below); lines represent upper and lower limits of complete spatial randomness. Uniform pattern indicated by L(d) falling below lower limit; clumped pattern when L(d) above upper limit.

Sample simulation results showed that sample estimates from small plots deviated from the population mean. Differences between the estimate for the large sample block and the average estimate from n = 30 randomly-located sample plots were evident when fewer than 20 trees per plot were sampled in the uniform thinned stand and when sampling fewer than 30 trees per plot in the stand with a random spatial pattern. Results indicated that some differences between population and sample estimates can be expected irrespective of plot type or number of trees per plot when using 30 randomly-located plots to sample clumped stands (*fig. 2*).

Sample variance for all stand parameters decreased as plot size increased. These increases in sample precision are depicted as 'tapering' of 95 percent confidence intervals around the bootstrap mean of sample plot estimates. However, the marginal benefit – in terms of greater precision – of expanding fixed-area plots or lowering the BAF in variable-radius plots to capture more trees generally decreased as number of trees per plot progressively increased. This effect was most pronounced when the average number of trees per plot increased beyond 30 trees for estimates of BA, volume, and LAI, and beyond 20 trees for dominant and mean top height. Results for density generally indicated that important improvements in precision were still being obtained as plots were expanded to capture around 40 trees per plot (*fig. 2*).



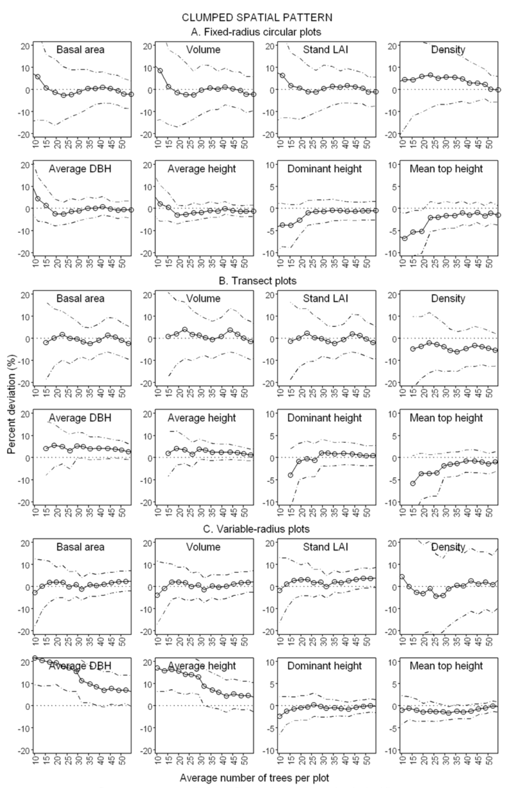


Figure 2—Bootstrap sample means and confidence intervals (α = 0.05) for stand parameters in terms of percent deviation from large-block ‘population’ values using fixed-radius circular plots (A), belt transects (B), and variable-radius plots (C) in redwood stand with uniform spatial pattern (*upper left*), random spatial pattern (*upper right*), and clumped spatial pattern (*lower left*).

Sample precision varied between stand parameters, plot types, and spatial patterns. There was no ‘best plot type’ that gave the most precise estimates for all stand parameters in all stands. Density estimates were especially sensitive to plot type. The 95 percent confidence intervals for density estimates were narrowest for circular fixed-radius plots and 50 m north-south transects. Transects gave the most precise estimates of density when an average of 30 to 40 trees per plot were sampled. Variable-radius plots generally produced more precise estimates of stand BA, volume, and LAI, especially in the clumped stand.

Plot type had little effect on precision of estimates in the uniform stand where ≤10 trees per plot were needed to obtain a 95 percent confidence interval of ±5 percent deviation from estimates of average dbh, average height, dominant height, or mean top height in all plot types (*table 2*).

Table 2—*Average number of trees per plot needed for bootstrap confidence intervals (α = 0.05) to equal ±5% or ±10% deviation from parameter estimates in stands with uniform, random, and clumped spatial patterns, sampled using fixed-radius circular plots, belt transects, and variable-radius plots. C.I. = confidence interval; LAI = leaf area index; Av. = average. Dom. = dominant; MTH = mean top height.*

Spatial pattern[1] and plot type	C.I.±10% deviation[2]				C.I.±5% deviation[2]			
	Average number of trees per plot							
	BA	Volume	LAI	Density	Av. dbh	Av. ht	Dom. ht	MTH
A. Uniform								
Circular plot	12	13	10	8	<8	<8	10	9
Belt transect	9	9	8	<8	<8	<8	<8	<8
Variable radius	8	9	8	10	<8	<8	<8	<8
B. Random								
Circular plot	11	12	12	11	13	8	12	9
Belt transect	22	22	23	12	9	<8	12	12
Variable radius	9	9	9	42	43	34	16	<8
C. Clumped								
Circular plot	30	33	30	38	40	28	35	40
Belt transect	26	29	27	31	29	18	40	40
Variable radius	15	14	15	>50	>50	>50	26	17

[1]Uniform pattern: Block 1 thinned to 180 stems/ha; random pattern: Block 2; clumped pattern : Block 3.
[2]Average difference (percent deviation) between sample estimate and upper/lower bounds of 95% (α = 0.05) confidence interval for estimates from simulated sample of n = 30 plots.

Sample estimates for the clumped stand were almost always least precise. Variable-radius plots produced much poorer estimates of density in stands with random and clumped spatial patterns. Variable-radius plots needed an average of 42 trees per plot to achieve a 95 percent confidence interval of ±10 percent deviation from density estimates in the stand with a random spatial pattern. This level of precision was not achievable sampling the clumped stand with over 50 trees per plot in 30 randomly located variable-radius plots (*fig. 2*). However, regression models depicting relationships between coefficient of variation (percent) and average number of trees per plot indicated that spatial pattern barely affected the precision - in terms of coefficient of variation – of variable-radius plot estimates of BA, volume, and LAI. Spatial pattern also had less effect on variable-radius plot dominant height and mean top height estimate precision than estimates from fixed-radius circular plots or belt transects. In general, sample precision was affected more by spatial pattern than plot type (*fig. 3*). Models describing coefficient of variation as a function of the number of trees per plot for each plot type and spatial pattern (*table 3*) had average prediction errors (RMSE) of only 1.1 percent (max. error 2.3 percent).

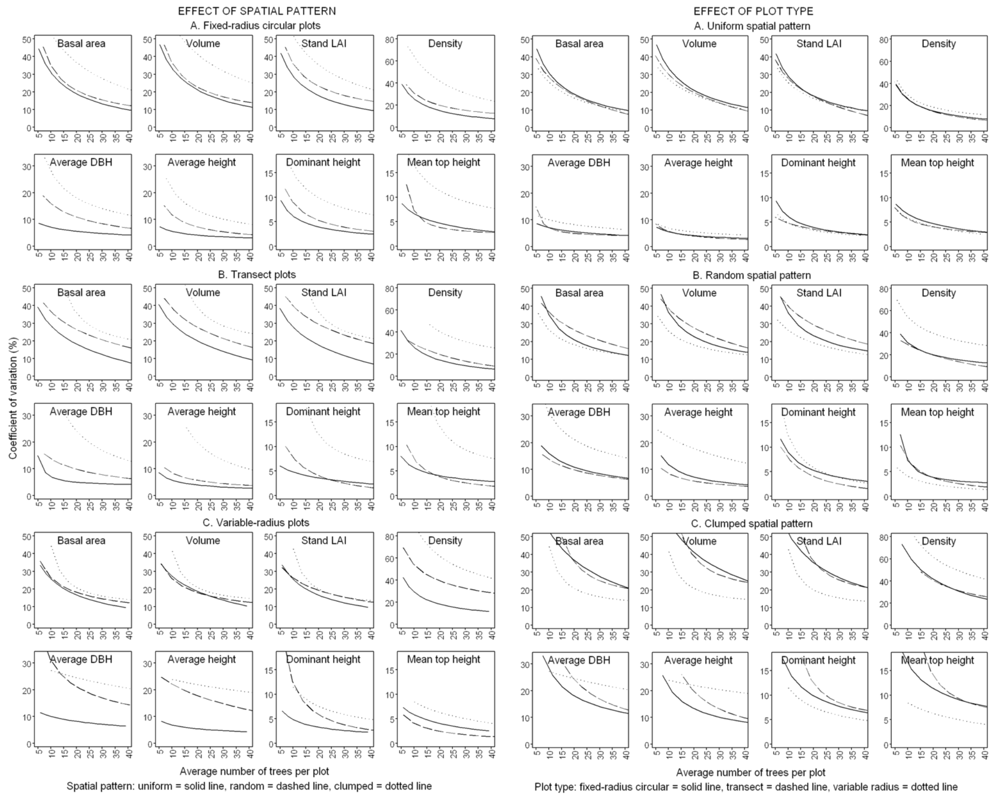


Figure 3—Effect of spatial pattern (*left*) and plot type (*right*) on coefficient of variation (percent) for sample estimates from n = 30 fixed-radius circular plots, 50-m belt transects, and variable-radius plots in stands with uniform, random, or clumped spatial patterns.

Discussion

Sample precision is affected by stand structure (e.g., Gray 2003) and spatial pattern of tree locations (e.g., Lessard et al. 2002). Confounding effects of structural differences were reduced by sampling even-aged redwood stands with similar stocking (BA, LAI). Thinning in Block 1 (the older stand) reduced tree size variability to levels comparable with the two other (younger) stands (*table 1*). Thinning had a greater impact on the estimate of dominant height than mean top height, suggesting that measures of top height are more robust metrics than dominant height for site index estimation when diameter distributions of even-aged stands are altered by low thinning. However, both top height and dominant height estimates are sensitive to plot size due to tree selection effects; estimates typically increase with increasing plot size (Garcia 1998; Magnussen 1999). Similarly, estimates from smaller plots may be more sensitive to spatial autocorrelation of tree size i.e., from microsite differences and genetics of sprouting species such as redwood. The plot size effect was evident in this study when increasing bootstrap means for dominant height and mean top height of redwood were observed with increasing plot size in most sampling simulations irrespective of plot type or spatial pattern (*fig. 2*).

Table 3—*Models of coefficient of variation (%) as a function of average number of trees per plot for eight parameter (parm.) estimates from three plot types sampling three spatial patterns.*

Parm.	SP[1]	PT[2]	F[3]	*a*	*b*	*c*	Parm.	SP[1]	PT[2]	F[3]	*a*	*b*	*c*
BA	U	F	S	164.79	0.71	0.37	*DBH*	U	F	S	31.36	0.90	0.22
	U	T	L	62.53	-14.76	-		U	T	E3	3.54	7.02	-
	U	V	S	90.41	0.47	0.43		U	V	L	16.11	-2.63	-
	R	F	P	194.33	-0.74	-		R	F	S	91.07	0.90	0.29
	R	T	L	69.94	-14.48	-		R	T	S	50.37	0.61	0.33
	R	V	P	94.09	-0.55	-		R	V	P	110.11	-0.55	-
	C	F	S	125.09	0.26	0.52		C	F	P	117.94	-0.63	-
	C	T	E3	12.47	21.04	-		C	T	P	334.59	-0.88	-
	C	V	E3	9.46	15.68	-		C	V	S	31.77	0.03	0.75
VOL	U	F	S	161.96	0.68	0.37	*HT*	U	F	P	14.43	-0.41	-
	U	T	L	63.90	-14.66	-		U	T	P	20.14	-0.54	-
	U	V	L	55.64	-12.32	-		U	V	P	15.04	-0.34	-
	R	F	P	176.34	-0.68	-		R	F	P	60.49	-0.71	-
	R	T	L	74.74	-15.67	-		R	T	P	32.53	-0.58	-
	R	V	P	85.55	-0.52	-		R	V	S	31.58	0.07	0.69
	C	F	S	162.56	0.38	0.43		C	F	P	102.23	-0.68	-
	C	T	E3	15.47	18.37	-		C	T	P	395.90	-1.00	-
	C	V	E3	10.23	14.19	-		C	V	S	27.53	0.03	0.69
LAI	U	F	S	209.91	0.96	0.32	*DHT*	U	F	P	27.27	-0.65	-
	U	T	L	61.78	-14.74	-		U	T	L	8.85	-1.75	-
	U	V	S	125.13	0.71	0.35		U	V	P	17.68	-0.56	-
	R	F	P	155.94	-0.63	-		R	F	P	50.99	-0.76	-
	R	T	L	74.57	-15.05	-		R	T	S	43.48	0.60	0.46
	R	V	S	80.76	0.49	0.36		R	V	P	147.31	-1.08	-
	C	F	S	231.58	0.69	0.33		C	F	P	59.63	-0.60	-
	C	T	E3	12.94	20.53	-		C	T	E3	3.95	22.97	-
	C	V	E3	9.10	15.71	-		C	V	P	48.45	-0.62	-
SPH	U	F	S	312.47	1.32	0.28	*MTH*	U	F	S	25.82	0.63	0.33
	U	T	S	160.80	0.72	0.40		U	T	P	17.12	-0.48	-
	U	V	P	140.84	-0.68	-		U	V	S	16.03	0.36	0.45
	R	F	P	133.72	-0.63	-		R	F	E3	2.04	12.76	-
	R	T	S	46.95	0.06	0.87		R	T	P	68.00	-0.96	-
	R	V	P	155.02	-0.46	-		R	V	S	142.08	2.28	0.19
	C	F	S	258.37	0.58	0.38		C	F	P	56.39	-0.54	-
	C	T	P	272.54	-0.64	-		C	T	E3	4.17	24.21	-
	C	V	P	329.08	-0.56	-		C	V	L	15.75	-3.16	-

[1]Spatial pattern: U = uniform; R = random; C = clumped.
[2]Plot type: F = fixed-radius circular; T = 50-m belt transect; V = variable-radius plot.
[3]Functions: P = power; L = logarithmic; E3 = type III exponential; S = Schumacher.

The marginal benefit - in terms of precision - of sampling additional trees (by increasing plot size) decreased with increasing number of trees sampled (*fig. 2, 3*). This result implied that smaller plots or higher BAF should be favored in terms of statistical efficiency. In practice, BAF are typically selected to capture around 5 to 12 trees per plot (Avery and Burkhart 1994). However, simulation results indicated that below a threshold of approximately 10 redwood trees per plot, sample variance and confidence intervals for the sample mean could increase dramatically. The effect of spatial pattern on precision of some sample estimates was greater in smaller plots (*fig. 3*). The CV models (*table 3*) can be used to define approximate minimum sample size for various forest inventory strategies. For example, a fixed-radius plot size capturing an average of 20 trees in a clumped stand is predicted to generate BA estimates with a coefficient of variation of 36.8 percent, therefore the number of plots needed to obtain an estimate within ±5 percent of the population value at the 0.80 probability level ($t_{0.2,\infty} = 1.282$) is approximately $n = ((1.282*36.8)/5)^2 \approx 89$ (Avery and Burkhart 1994). Alternatively, the number of variable-radius plots (that also capture an average of 20 trees per plot) needed to obtain BA estimates with same level of precision and confidence is approximately 28 (based on predicted CV of 20.7

percent). However, around 250 such variable-radius plots would be needed in the clumped stand to obtain density estimates within ±5 percent of the population value at the 0.80 probability level; density estimates from 28 variable-radius plots capturing an average of 20 trees per plot were predicted to be within ±15 percent of the population value at the 0.80 probability level. These examples illustrate the sensitivity of sample precision to changes in plot type and between stand parameters.

The general trend in all plot types was of decreasing precision as spatial pattern changed from uniform to random to clumped (*fig. 3*). These findings are consistent with results presented by Lessard et al. (2002) showing that the expected variance of estimates from a Poisson-distributed random spatial pattern will always be less than estimates from a negative binomial-distributed approximation of a clumped pattern of tree locations. However, our results indicated that the effect of spatial pattern differed markedly between some stand parameters and its effects can interact with plot type. For example, spatial pattern most affected precision of redwood density estimates from variable-radius plots and precision of average dbh and height estimates in all plot types. Precision of BA, volume, and LAI in variable-radius plots capturing >15 trees was least affected by spatial pattern. Variable-radius plots generally made more precise estimates of ‘total’ stand parameters such as BA, volume and LAI where larger trees make a greater contribution towards the estimate. Average dbh and height estimate precision was poorest in variable-radius plots, except in smaller plots within the clumped stand where fixed-area plots performed poorly. Variable-radius plots consistently produced the least precise estimates of density for a given sample size. This problem was most pronounced in the stands with random or clumped spatial patterns (*fig. 3*). Hebert et al. (1988) and Schreuder et al. (1987) also found variable-radius plots to make marginally more precise estimates of BA, but less precise estimates of density when compared with fixed-radius plots. Schreuder et al. (1987) found fixed-area sampling most efficient for stand density and number of small trees, and variable-radius sampling best for density in larger dbh classes. In differentiated stands or multiaged stands with numerous small trees and fewer large trees, fixed-area plots or transects can capture more small trees and fewer large trees than are needed. This problem can be mitigated by installing a series of concentric plots each sampling a different size class or stand component i.e., cohort or canopy layer (Spurr 1952), by implementing a multistage design (Thompson 2002), or by using variable-radius plots.

The accuracy and precision of estimates can be affected by sampling and non-sampling error. Sampling errors arise because only part of the population of interest was sampled. They tend to decrease with increasing sample size, and can only be avoided by complete enumeration (Avery and Burkhart 1994). Non-sampling errors include measurement errors and incorrect sample plot establishment. These errors tend to have a greater impact in smaller plots, and persist with increasing sample size (Thompson 2002). Belt transects have a greater proportion of edge, and therefore a greater probability of sampling bias from incorrect plot establishment when ‘borderline’ trees are present (Bormann 1953). Variable-radius plots are susceptible to systematic errors caused by incorrect calibration, hidden or leaning trees, and errors in slope correction (Avery and Burkhart 1994). Another potential non-sampling error is bias introduced by inappropriate volume tables or allometric equations (Chave et al. 2004). Our analysis ignores the uncertainty and propagation of errors associated with predictions of redwood tree volume and leaf area and therefore underestimates the uncertainty around these estimates.

Sampling simulations were conducted using data collected in 85 to 100 year old redwood stands where density and tree size variability remained high despite many decades of competition. The range of spatial patterns and tree sizes should encompass conditions found in many size-differentiated natural stands of redwood. Stand attributes have been quantified using detailed stand summary data (*table 1*) and Ripley's K analysis (*fig. 1*), giving three-dimensional characterizations of sample stand structure. Application of results in stands with different densities or tree sizes is facilitated because we reported coefficient of variation and confidence intervals in percentage terms, and average number of trees per plot instead of plot size. Narrower confidence intervals around sample estimates can be expected in stands with greater uniformity of tree size or spatial pattern, such as younger stands and plantations.

Conclusion

Redwood forest managers and researchers interested in precise density estimates (e.g., to design and evaluate silvicultural treatments, monitor changes during stand development, or for spacing experiments), should favor fixed-area plots over variable-radius plots in stands with random or clumped tree spacing. Variable-radius plots were advantageous when sampling BA, volume, and LAI. Choice of plot type hardly affected precision of dominant and mean top height estimates, except in the clumped stand where estimates from variable-radius plots were more precise. In general, larger plots will be needed to achieve a desired level of precision when sampling stands with a higher degree of clumping. Larger plots are less likely to underestimate dominant height or top height. Top height was less affected by thinning and is recommended over dominant height in managed stands. Sampling in redwood forests should be preceded by careful definition of objectives and consideration of stand structure and spatial pattern before making decisions about plot type and sample size.

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Management Practices Related to the Restoration of Old Forest Characteristics in Coast Redwood Forests

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Abstract

A standardized, interactive, interview process was used with practicing Registered Professional Foresters asking a suite of questions to ascertain their management approaches to coast redwood (*Sequoia sempervirens* [D. Don] Endl.) stands that could best be transferred to other projects and lands interested in recruiting older forest characteristics. The assimilated results provided management insights that provide comparisons of similarities and vagrancies between properties, management styles and approaches. The survey included the properties and practitioners along the north-south geographic axis of coast redwoods to account for the diversity and range of physical and edaphic heterogeneity that is expressed within the forest type.

Forest managers generally agreed that uneven-age management provided the basis from which to begin the discussion further recognizing that a fully stocked stand was equally important and may require a "transitional" period depending on the initial condition of the stand in question. Furthermore, it was generally recognized that a positive financial return for the forestland owner was vital in order to secure their support throughout the period of restoration while advancing the stands to incorporate and retain larger size class cohorts and other structural elements associated with older forests.

In all cases the desire to increase the distribution of tree sizes classes (the inverse "J" curve) while simultaneously retaining existing conditions peculiar to older forests (i.e. tree trunk hollows, broken tops, reiterative branches, and others), affords the forest manager the ability to set the stand on a particular management pathway while affording the flexibility to adjust management decisions over time and capture market opportunities. Existing, archived datasets of unaltered old growth stands are used for comparisons in order to assist those interested in recreating old forest conditions.

Key words: *Sequoia sempervirens*, coast redwood, restoration, old forest characteristics, management practices

Introduction

The management for old forest characteristics in coastal redwood forests is a topic of interest among conservation interests and policy leaders in California. Unfortunately, a limited knowledge base regarding the structure and composition of older forests is available to land managers to assist them in making management decisions to achieve a specified future desired condition. Much of the scientific inquiry investigating the characteristics of redwood forests is relatively new and promises to provide useful and enlightening information in the years to come (Giusti 2004, Sillet and Van Pelt 2000).

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Historic and contemporary forestry practices have simplified the structural components of coast redwood forests (O'Dell 1996). Recruiting the structural elements commonly found in older forests is recognized as an important management objective in younger forest stands to address issues of biological diversity and forest integrity (Mladenoff 1993, Spies 2002). Simply stated, older forest characteristics are a function of time and chaotic events such as storms, wind, fire, and landslides whose legacies shape forest structure and composition and which may be difficult to mimic through traditional management practices. Interest in recruiting old forest characteristics is not limited to the redwood region and has been document for other forest regions (National Research Council 2000). As land ownership patterns and objectives continue to shift away from corporate ownership it is generally expected that this interest will continue to grow and find application in other forest types throughout the West.

Redwood characteristics through its range

The coast redwood forest, being dispersed along a north-south axis, has characteristics affected by geographic location and localized climate and edaphic conditions (Sawyer et al. 2000). Any comparison between redwood stands from differing geographic locations should be viewed as reference points for guidance purposes. Furthermore, numerical comparison between stands should recognize that differences exist in structural attributes between trees growing on upland sites vs. trees growing on alluvial flats and that comparisons should be limited to similar site characteristics. As illustrated in table 1 stands growing on alluvial deposits generally have higher tree densities then those stands on adjacent upland sites. This illustrates the need to recognize that no one description will define or articulate the structural elements of any particular site and any reference data should be used conservatively in management discussions.

Table 1—*Stand density comparison between old growth coastal redwood stands from different geographic locations with differing climatic and edaphic conditions (Giusti 2004).*

Stand Age	Old Growth	Old Growth	Old Growth	Old Growth
Location	Redwood Creek	Little Lost Man Creek	Bull Creek	Montgomery Woods State Reserve
County	Humboldt	Humboldt	Humboldt	Mendocino
Tree Size	TPA	TPA	TPA	TPA
> 24" dbh	No data	20	40	16
> 32" dbh	No data	15.7	40	------
> 40" dbh	11.3	12	35	15

Redwood forest characteristics vary greatly throughout the range. In general, the presence of hardwoods increases with distance from the coast. Zinke (1988) demonstrated how moisture gradients affect species associates from Sitka spruce-grand fir-hemlock in moist areas, redwood mixed with other conifers, redwood mixed with hardwoods, to Douglas-fir- hardwoods, to grassland-oak woodlands in the driest locations. The northern limit of the redwood forest is determined by winter conditions, particularly from frost damage to young trees (Baker 1965).

Generally older forests in the northern part of the range are the largest, most continuous due to the higher amounts of winter rainfall and summer fog which help to moderate the effect of the rainless summers (Veirs 1996). As the sites become hotter and drier and elevation increases, a more favorable environment is created for Douglas-fir, and it can be found sharing the upper canopy with redwoods.

Dagley and O'Hara (2003) report that near its southern extent redwood can be found 5 to 19 km inland with Douglas-fir, tanoak, and madrone (Zinke 1988). Moving inland it can be found in isolated patches up to 32 km from the coast while further south redwood is found only in very moist canyon bottoms, such as along the Big Sur River. Both Baker (1965) and Zinke (1988) have written that the limit to the southern range seems to be restrained by soil moisture. However, high summer temperatures may also help determine redwood's southern boundary (Barbour et al. 2001).

There does exist some limited descriptive fieldwork for non-tree attributes of older redwood forests. Sholars (in Sawyer 2000) has cataloged 47 species of lichens as part of the Bryophytic component for an old growth forest. Similarly, Largent (in Sawyer 2000) has identified over 323 species of fungi associated with old growth redwoods. Others (Thornberg, personal communication) have taken their leads and are now pursuing field studies to determine the opportunities to inoculate previously logged stands that are missing these non-vascular botanical components as a means of recruiting missing botanical elements to mimic older forest stands.

Structural characteristics of older forests in the northern tier of the redwood range

Baseline conditions for coast redwoods are found in limited quantities. In 1969 the U.S. Department of Interior initiated a project to identify and purchase what is now known as Redwood National Park in the northwestern corner of California. As a result, stand structure data was collected in old growth redwood stands proposed for acquisition (Hammon, Jensen & Wallen, 1969 a, b, c, d). The synthesized information from the surveyed sites provides an opportunity to compare size class distributions in un-logged forests (*fig. 1*) and represents one of the few existing data sets for unaltered stands.

The resultant data from the approximately 9,000 aggregated acres cruised demonstrates a relatively similar size class distribution over the surveyed area. Given the acreages of the cruised parcels, one can surmise that both upland and riparian sites were sampled. The average tree density of the three ownerships was 32 trees per ac, with tree sizes below 50 in dbh dominating. In each case, trees >50 in dbh were common and averaged >2 trees/ac. Similarly, each property averaged approximately one tree >90 in dbh per ac.

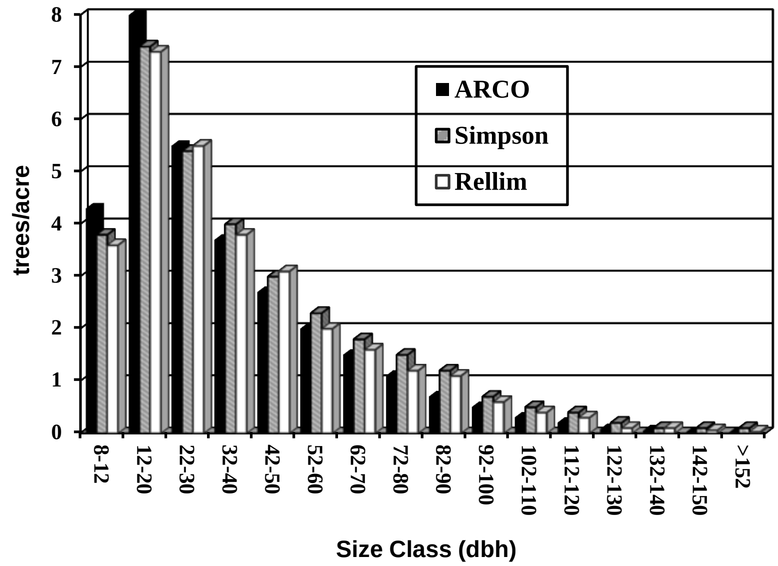

Figure 1—Size class distribution of old growth redwood trees; ARCO, Simpson and Rellim timber companies. Del Norte County. 9,378 acres. ≈ 32 trees/acre. Source: HJW 1969 a, b, c, d.

Though each property shared similar traits when comparing size class distribution, they had strikingly different conifer composition. Of the properties surveyed by HJW in 1969 the Rellim property had the least percent by volume of redwood and the highest percent by volume of Douglas-fir (*fig. 2*).

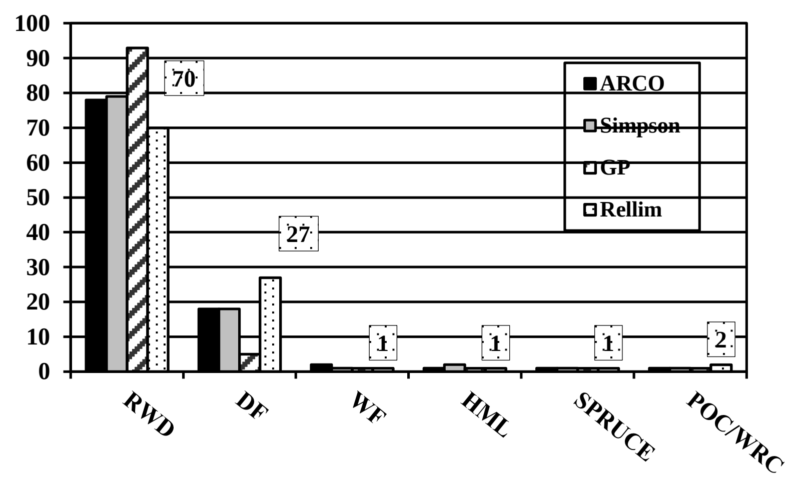

Figure 2—Relative percent by volume of tree species on ARCO, Simpson, Georgia-Pacific and Rellim Redwood Companies. Source: HJW 1969 a, b, c, d.

Combined, these data demonstrate the need to recognize that not all redwood stands within the range are created equal. Understanding the location along the north-south axis in addition to proximity to the coast with its influences is an important consideration when attempting to identify redwood associates and approximate species composition ratios.

Methods

Objective

This project was designed to "identify and articulate the applied knowledge that exists from practitioners who have experience with silvicultural operations that might affect forest restoration efforts." This was accomplished by identifying knowledgeable individuals widely recognized by their peers and others who have been part of an on-going discussion regarding recruitment of old forest characteristics or who have successfully or are attempting to retain or recruit old forest characteristics into their managed stands. Those interviewed represented managers of non-industrial, industrial and public lands.

Survey design

The project conducted range-wide surveys of managers with various management experiences in order to capture and synthesize their collective experiences at restoring old-forest characteristics in coast redwoods. The preliminary management survey review included identification of public and private landowners and land managers either (a) managing ancient coast redwood forests, or (b) conducting silviculture directed towards restoration of old-forest (or at least older) characteristics in coast redwood forests.

The survey was predicated on four main premise "themes" that were designed to serve as the basis for a number of questions asked to each practitioner. The thematic premises where used to help focus the scope of the questions and responses to better compare and identify "key findings" – similarities in approaches.

Results

Results of the survey are presented by premise. Each of the four premises is stated, followed by questions and key findings for each premise.

Premise 1

Statement: Silviculture is widely viewed as the management option that holds the greatest potential to accelerate and restore old forest characteristics. The following questions are intended to identify those practices that best accomplish this objective.

Question 1a

Do you have an existing stand of older trees? How are you managing that stand to maintain old forest characteristics? Do you manage older stands of trees differently than any other?

Key findings

In every case where older trees still exist they are identified by the owner/manager as an important component of the ownership that has High Conservation Values "HCV". In all cases, the owner/manager recognizes the older trees as "old growth" delineated as either whole stands or residual trees left from past harvest activities. In every case, stands that have not yet been harvested will not be entered and are being preserved. Where partial harvesting has occurred and residual trees remain, future harvest activities will focus on the trees within the matrix surrounding the residual old trees. In specific instances (MRC, SDSF, JDSF) policy language has been developed that 1) articulates the qualitative description of the stands/trees to be protected, 2) identifies the stands of old trees to be reserved, and 3) describes the management options available for the conservation of their values.

Question 1b

What are your management goals, monitoring techniques etc. toward redwood regeneration?

Key findings

In every case, the properties visited are being managed for the long-term production of timber and sawlogs. Hence, regeneration is a key component of the property's management objectives. Shared management objectives regarding regeneration included:

1) Stump sprouting is promoted since it is a unique attribute of coastal redwood and it avoids genetic contamination of the stand.
2) Harvest activities focus on the dominant and co-dominant trees within a sprout clump retaining the smaller size cohorts for future harvests. Thus retaining the genetic architecture of the redwood component of the stand. Harvest trees within clumps are selected on the basis of achieving both spacing and regeneration criteria.
3) Timber volume data are collected and monitored using conventional field plots in combination with growth and yield models. Standing timber volumes and recruitment estimates across all size and age classes are commonly collected to ensure that regeneration and future harvest objectives will be assured.

Question 1c

What are the most commonly applied silvicultural treatments on the properties you manage?

Key findings

1) Commercial thinning of redwood is the most widely applied silvicultural practice. In all cases, the RFP is selecting trees "from below" selecting co-dominant trees while retaining dominant individuals. This prescription concurrently achieves spacing, growth and volume objectives while maintaining contiguous canopy cover (*fig. 3*). *Figure 3* illustrates the concept of harvesting while continually increasing volume over time and space. The

incremental volume is achieved through selection of large tree retention as well as recruitment of younger cohorts.

2) To a lesser degree single-tree selection was the second most applied silvicultural practice. This selection approach was utilized to achieve specific site or ownership objectives regarding growth or volume.

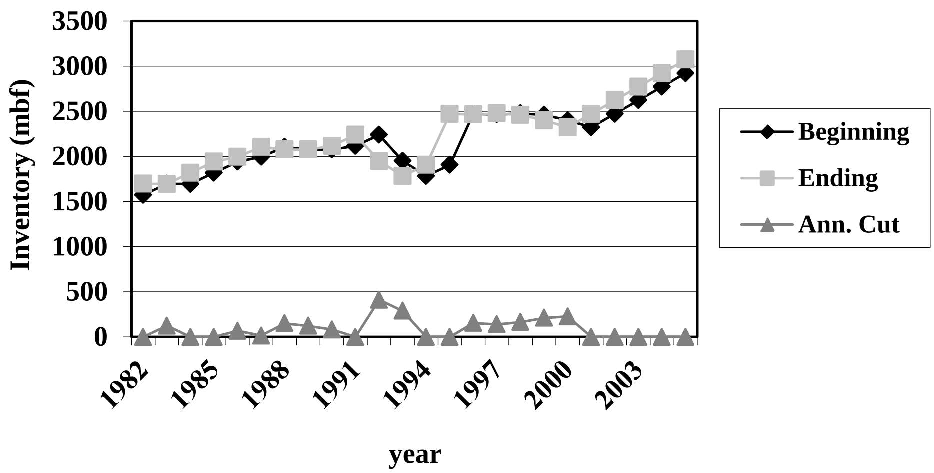

Figure 3—Humboldt Co. commercial thinning 1982- 2000. Site II. 70 acres. Birth of Stand 1915-1925. First commercial thin 1976. (Source: Able Forestry).

Question 1d

What is your philosophy regarding early spacing treatments (pre-commercial thinning) and when is it best applied (if ever)?

Key findings

Pre-commercial thinning (PCT) is virtually ignored as a management approach on most of the sites visited. Those individuals who practice PCT consider the optimum stand age to achieve desirable stand characteristics to be between 8 and 18 years (JDSF & MRC). This is based on their field experience for optimal release of remaining redwoods.

Question1e

What is your philosophy about the relationship between commercial (mid-rotation) thinning and redwood regeneration? When is it best applied?

Key findings

Commercial thinning is the most widely utilized silvicultural practice of the sites visited. In all cases, the approach is similar. The RPF chooses the harvest trees after assessing dominant and co-dominant trees. Generally, dominant trees are retained in the stand to increase stand volume through release. Volume is increased from both increased diameter and height. Additionally, commercial thinning initiate’s gap phase successional processes allowing for natural recruitment of future crop trees from stump sprouting.

The amount of wood volume harvested during each entry is dependent on the forester and the site. In some cases, RPF's concentrate on removing 30 percent of the volume per entry, while other RPFs remove 1/3 of the trees. *Figure 4* demonstrates the intent of this silvicultural approach. Here the "inverse J" curve is adjusted to move trees to the right of the "x" axis (harvestable trees) while addressing recruitment concerns along the "y" axis.

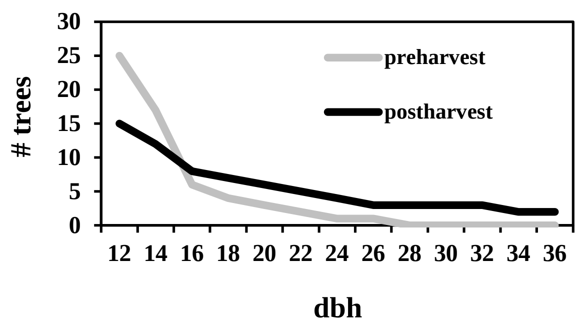

Figure 4—Desired outcome over time from commercial thinning. Source: Blencowe Forestry.

Question 1f

What experiences relative to recruiting old forest characteristics can you share regarding those treatments?

Key findings

1) By design or happenstance the use of mid-rotation selection cutting has resulted in the gradual increase in both average tree girth and height while maintaining dense forest canopy.
2) Mid-rotation thinning further allows the forester to identify and retain distinctive habitat elements that are present in the stand prior to harvest.
3) Furthermore, selection cutting allows for development of multiple age stands that address both old forest characteristic retention and future crop tree recruitment.
4) When recruitment goals are not being met, mid-rotation thinning allows the RPF to select appropriate sites for group selection harvest units. This combination of silvicultural treatments allows for increased regeneration while minimizing canopy disturbance and retaining old forest characteristics in the remainder of the stand.

Question 1g

Does your approach to regeneration require active vegetation control for undesirable species?

Key findings

The majority of interviewees have no need to consider vegetation management scenarios in established stands. They are achieving control of unwanted plant species either through the use of shade and/or the direct impacts of logging which are targeting the removal of undesirable species.

Question 1h

Is pruning a viable option for improving tree vigor, stand characteristics, and economic returns?

Key findings

Pruning is not generally being applied at the sites visited to improve tree vigor. The consensus by those interviewed is that pruning is an activity to promote “clean log production” and the cost of pruning is not justified under the silvicultural methods being used. Furthermore, few saw the utility of pruning in recruiting older forest characteristics.

Question 1i

What has experience taught you about the type of un-even age prescriptions that may have the greatest potential to provide information useful in old forest restoration?

Key findings

This question represents the primary purpose for undertaking this project. The guidance provided by the interviewees can inform future management decisions with regard to stand manipulations intended to achieve old forest characteristics. Key points are:

1) Un-even age prescriptions allow the forester to identify and retain old forest characteristics already present in the stand while designing a system to ensure adequate stand regeneration to meet financial objectives.
2) Managers are exploring and evaluating the harvest of all stems within a stump sprout to achieve both volume and regeneration objectives while creating canopy gaps.
3) Stump sprout harvesting most likely mimic the natural disturbance pattern for individual tree replacement in coast redwood.
4) Silvicultural prescriptions should be viewed as a mechanism that achieves both volume production and initiates a stand response to set the path of growth and development into the future.
5) Un-even age management does not commit the entire stand to a predetermined path of growth and habitat development. It allows the forester to make adjustments at any point in the development of the stand.
6) Through the use of uneven-age management the forester is able to maintain continuous forest canopy thereby positively affecting ambient conditions in the stand i.e. shade, humidly, temperature, forest floor cover, mid-level canopy characteristics.

Premise 2

Statement: Natural disturbances are widely viewed as playing a key role in defining old forest characteristics. Many of those features include peculiar traits exploited by various species of terrestrial wildlife and flora.

Question 2a

What specific habitat elements do you view as being uniquely characteristic to coastal redwood forests?

Key findings

The respondents each identified several habitat characteristics of coastal redwood that they are mindful of during their management decisions. These generally can be categorized into biotic and abiotic characteristics.

Biotic Elements	Abiotic Elements
Trunk hollows (goose pens) the result of multiple burns. Not unique to coast redwood but difficult to recruit under current fire suppression regimes.	Soil types, depth, slope and aspect of management unit and how these conditions affect species composition and tree growth.
Broken tops, deformed trunks, large woody debris caused by lighting, wind, disease and pests (in hardwoods) etc.	Rock outcroppings and shallow soils in the middle and southern portions of the range. Steep slopes and numerous 1st order streams common throughout the region.
Reiterative branching unique of older trees and difficult to recruit using modern silvicultural time frames.	Influences of past fire events (natural and induced) on stand composition and structure.
Multiple layered canopy with redwood and Douglas-fir being dominant.	
Ability to sprout, a unique characteristic for a North American conifer.	
Longevity, compared to other species found in the redwood region.	

Question 2b

What specific management practices (if any) have you applied or come to recognize that best create specific wildlife habitat feature(s)?

Key findings

1) Timber harvesting allows for the creation of canopy gaps of any desired size and shape. This freedom allows for canopy management using timber harvest to achieve regeneration goals as well as meeting wildlife management objectives.
2) White woods are most likely to provide the greatest opportunity for rapid nest cavity recruitment. Recognizing the role of white woods, in addition to hardwoods, for habitat element maintenance and recruitment can assist a manager achieve desired wildlife habitat goals.
3) Timber harvesting provides for a source of down and dead wood of any desired size and soundness.

Question 2c

From your experiences, are there any habitat elements that are easily recruited through silvicultural practices?

Key findings

A number of habitat elements are widely recognized as being relatively easy to recruit. These elements include:

1) Dead and down wood;
2) Large trees can easily be retained;
3) White woods are easily regenerated and retained;
4) Timber harvesting creates soil disturbance which aids in the germination of many botanical species;
5) Slash and litter are easily recruited which provides for nutrient cycling and erosion prevention in addition to cover for small mammals and herps.

Question 2d

From your experiences, are there any particular habitat elements that are particularly difficult to recruit through silvicultural practices?

Key findings

1) All respondents recognize the difficulty in recruiting trunk hollows (goose pens) associated with repetitive fire scarring. Subsequently, they further acknowledged the importance of trunk hollows as a habitat element and expressed a desire to see applied research projects designed to address this question.
2) Each of the interviewees recognized the difficulty in recruiting and maintaining phenotypically inferior trees in the stand. This paradigm is entrenched namely that retaining any inferior tree will be viewed as “high-grading”. Though an important consideration, overly aggressive sanitation policies can inhibit the recruitment of habitat elements often associated with less thrifty trees.
3) The other most notable characteristic not readily recruited through management are those elements directly associated with age i.e. reiterative branches, bryophyte colonization, and rotten logs. Most recognize the importance of allowing these conditions to develop over time.

Premise 3

Statement: Some natural disturbance patterns may not be easily mimicked by silvicultural practices i.e. fire, landslides. (These questions were designed to explore the practitioner’s management approaches in light of the fact that some variables may be out of their control.)

Question 3a

In this time of aggressive fire suppression, how do you manage forest fuels? Can silviculture affect fuel loads? If so, how?

Key findings

There is strong agreement that silviculture should not be viewed as a surrogate practice that can always be substituted for fire. It is further recognized that fire serves as a catalyst in many forest processes such as nutrient cycling and the formation of key habitat elements as well as potentially depletion of forest nutrients and downstream eutrophication. However, there was strong consensus among those interviewed that silvicultural practices can be used to mitigate fuel loading in the absence of fire. Specifically, silviculture and timber harvest activities are often aimed at reducing wildland fire fuels:

1) Lopping of slash is a common practice among RPFs working in areas where aesthetics and fuel suppression are equally important.
2) Tractor crush. As the name implies, the machinery used in timber harvest is used specifically to reduce the height of slash and other residual vegetative materials generated by harvesting.
3) Harvest activities that reduce ladder fuels are widely viewed as a positive activity associated with active silvicultural manipulations.
4) Windthrow. All of those interviewed recognize the positive relationship between silviculture and tree spacing as a means of inhibiting the spread of fire into the crowns. Additionally, several individuals further recognize the importance of incorporating naturally occurring fuel breaks into their silvicultural designs. Windthrow and other naturally occurring breaks in the vegetation across the landscape are generally viewed as opportunities that should be incorporated into management designs to maximize their effectiveness to limit the spread and impact of wildland fires.

Question 3b

How do you discourage the recruitment of less desirable tree species using silvicultural practices in absence of other natural disturbance regimes?

Key findings

Respondents viewed this question more broadly to include all less desirable species, not just trees.

1) Canopy management to maintain shade. Since all of the participants for this project are utilizing un-even aged management they acknowledged the relationship between canopy management and the role of shade on the forest floor to inhibit the recruitment of some less desirable species into the forest canopy i.e. tanoak (*Notholithocarpus (Lithocarpus) densiflorus*). Subsequently, respondents further acknowledged that maintaining high shade component creates a greater chance of recruiting other less desirable species that are more shade tolerant i.e., grand fir (*Abies grandis*).
2) Species management using market conditions. Where applicable, particularly in the management of less desired conifers, positive market conditions often provide an opportunity to mark a stand to reduce the number of undesired trees and thus reduce potential seed sources for future recruitment.
3) Neighbor awareness program. The Arcata Community Forest has initiated a neighbor awareness program aimed at increasing public consciousness to reduce the planting and introduction of exotic and noxious plant species on

properties adjacent to the forest. By involving broader community participation, the managers of the forest hope they have been effective in slowing the recruitment of many less desirable species from the surrounding urbanized area.

Question 3c

Is fire a useful tool? Is it realistic to assume that fire can be used effectively?

Key findings

Yes. Fire was unanimously viewed as a useful tool. Cited as the “missing piece” in modern forestry practices, fire is seen as a key element in shaping the structure and composition of coastal redwood forests by all of the respondents. There was further consensus that serendipitous fire occurrences i.e. lightning strikes, are generally viewed favorably if conditions allow for low-intensity burns.

Premise 4

Statement: There may be differences in social preferences vs. scientific data for desirable characteristics of old forests. It is important to identify obstacles that exist that serve as an impediment to improving active management practices aimed at maintaining or recruiting old forest characteristics in redwood stands.

Question 4a

Are there treatments or other considerations that we have not discussed that may be important in recruiting old forest characteristics and function while meeting landowner expectations?

Key findings

Several pragmatic factors were identified by the respondents that regularly affect how the best intentions of an RPF can be negatively influenced. They include:

1) Logging contractor decisions.
2) Cable yarding considerations.
3) Harvesting techniques and landowner expectations.
4) Roads, landings and yarding corridors.
5) Mill Infrastructure and availability
6) Disincentive to produce large logs.
7) Newly introduced pests and pathogens (SOD).

Question 4b

Have you identified/delineated areas with High Conservation Values or other old forest characteristics? Have you articulated your management treatments in these areas in a management plan or some other document?

Key findings

In every case, interviewees have identified areas or elements of High Conservation Value (HCV). Generally, the larger ownerships/parcels tended to have their HCV areas/elements delineated in landscape plans, while smaller, non-industrial ownerships used the NTMP as the vehicle to identify HCV components. The most commonly identified and/or delineated HCV areas or elements included:

- Old growth trees & stands.
- Riparian areas.
- Wet areas.
- Threatened and endangered (T&E) species sites.
- Special project areas i.e. arbon sequestration sites.
- Scenic corridors.

Discussion

Little is known about how old redwood forests developed over time. Information on the development of redwood stands is important for determining if second-growth forests can develop old forest characteristics. The results of the survey suggest that periodic entries into a stand can place the stand on a pathway toward achieving some of the aspects of an older forest. This is supported by Hunter and Parker (1993) who wrote old forests experience intervals of repeated small-scale disturbance events rather than a single disturbance event. They were able to illustrate the episodic nature of gap formation, the links between canopy tree biology and gap properties, and the relationship between topography and disturbance regime. They found half of the 80 canopy gaps in an old redwood forest to have resulted from more than one disturbance event. It appears that for redwood forests, the gaps created by small-scale disturbances contribute to the overall structural complexity of the forest, but do not affect the long-term composition of the overstory layer (i.e., redwood remains the dominant overstory species) (Sugihara 1996).

Fire is considered a primary factor in shaping the structure and composition of old growth forests. The exclusion of fire is viewed by all interviewed as a principle constraint in our ability to recruit certain components found in older forests.

Older forests are more complex than metrics that are limited to tree size, basal area and volume growth can accurately describe. Subtleties such as bark configurations, limb and branch configurations, epiphytic plant communities, trunk hollows (both conifers and hardwoods), persistent, large down woody materials (redwoods), mid and sub-canopy plant communities, and soil deposition and recruitment create a synergy that is very difficult to mimic in time frames comfortable for human acceptance. The best we can hope to accomplish is to retain and recruit (when possible) any of the remaining older forest characteristics lost during the period of intense timber harvesting of the 19th and 20th century under the constraints of the 21st century regulations.

The properties visited are not uniquely challenged when trying to address restorative actions similar to many other projects on the north coast who are concurrently wrestling with the condition of their lands. Collectively the lands and the practitioners surveyed believe that restoration and economic return on investments is a positive approach to addressing environmental and ecological issues associated with recruitment of old forest characteristics over time.

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Carbon Storage in Young Growth Coast Redwood Stands

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Abstract

Carbon sequestration is an emerging forest management objective within California and around the world. With the passage of the California's Global Warming Solutions Act (AB32) our need to understand the dynamics of carbon sequestration and to accurately measure carbon storage is essential to insure successful implementation of carbon credit projects throughout the State. As the leader in forest carbon credit projects within the U.S., California's forest carbon protocols are being looked to as templates for successful carbon credit accounting. Coast redwood (*Sequoia sempervirens*) stands have the largest measured biomass per acre making the argument for use of the species in long-term carbon sequestration projects self evident. To date no direct measurement of both carbon fraction and wood density of coast redwood has been undertaken. With this study we tested the applicability of the current forest carbon project protocols set out by the Climate Action Reserve for forest carbon credit projects within California. Specifically we tested the applicability of a carbon fraction of 0.5 and a greenwood density of 0.34 g/cm^3 for coast redwood trees. Our main findings were that: 1) a species-average of 0.34 g/cm^3 significantly underestimated the wood density of the measured trees, 2) wood density varied predictably with tree height and wood type, 3) carbon fraction was significantly higher than the default 0.5, and 4) carbon fraction varied predictably in relation to wood type. Our results indicate that a simple approach to estimating carbon storage utilizing a carbon fraction of 0.5 and a species-average wood density underestimates carbon budgets in young growth redwood stands. Given the regional nature of carbon credit projects our results indicate that forest project protocols should be adjusted to allow for calculations that incorporate directly measured wood density and carbon fraction values within the forest carbon project area.

Key words: biomass, *Sequoia sempervirens*, carbon fraction, sequestration projects, wood density

Introduction

Carbon sequestration is emerging as an important goal for forestry. In order to make informed decisions related to managing forests for carbon sequestration it is necessary to have precise and empirically based estimates of carbon budgets (Brown 2002). Forest carbon budgets are typically determined by estimating the volume of a tree using species-specific allometric equations or genus-specific equations (Brown 2002, Jenkins and others 2003) to convert easily measured tree characteristics such as diameter at breast height into wood volume. The bone-dry wood density of the species is then used to convert this overall volume into tree biomass. Tree biomass is multiplied by a carbon fraction to estimate the mass of carbon contained in that tree. Species-specific values of carbon fraction are not widely available and therefore a

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generalized value of 0.50 is typically used (Brown 2002, Goodale and others 2002, Harmon and others 1990, Pacala and others 2001, Van Deusen and Roesch 2011).

Using a carbon fraction of 0.50 is problematic because carbon values that have been directly measured range from about 0.45 to 0.55 (Lamlom and Savidge 2003). The associated error with using 0.50 can therefore be as high as ten percent (Lamlom and Savidge 2003).

Another source of error in carbon budget estimates arises from variation in species specific wood density values from region to region. The default value for young growth redwood whole-tree basic wood density (called wood density hereafter) is 0.34 g/cm^3 (CAR protocol methodology - climateactionreserve.org), however regional estimates range from 0.32 g/cm^3 to 0.38 g/cm^3 (Luxford and Markwardt 1932) equating to deviations from the species average of 6 and 12 percent respectively.

Due to the longevity of the species and it's propensity to accumulate high levels of biomass (Busing and Fujimori 2005), young growth coast redwood forests have the potential to become significant carbon sinks for hundreds of years. Redwood stand volume production has been shown to be affected by stand density and structure implying that there is the potential to effect stand volume productivity with carefully planned silvicultural treatments (Berrill and O'Hara 2009). In addition to the high wood volume productivity within a stand of coast redwood, the wood products harvested from those stands has been shown to have long lifecycles (Highley and others 1995) allowing for long term carbon storage in wood products.

This study focuses on second growth coast redwood trees to pursue the following study objectives: 1) determine whether any differences in carbon fraction exist within three wood types of the tree bole at breast height; 2) determine whether any differences in density exist between the three wood types; 3) determine whether any carbon is lost due to oven drying; 4) develop methodology to precisely determine the carbon fraction of tree boles in young growth redwood; and 5) compare the directly measured values in the study area to the default values specified in standard methodologies (carbon fraction of 0.50 and wood density of 0.34 g/cm^3).

Methods

Study area and description

Tree cores were collected from coast redwood stands located at Railroad Gulch on Jackson Demonstration State Forest near Ft. Bragg, California. Redwood stands at this location regenerated naturally from stump sprouts and from seedlings planted after removal of the old growth stands around 1920 (Jackson 1991). Sample trees were randomly selected from a database created in 1982 when Railroad Gulch was divided into 14 contiguous blocks. A total of 42 trees were selected ranging in breast height age from 23 to 87, ranging in diameter from 13 to 86 cm and ranging in height from 5 to 45 m.

Core extraction and processing

Two cores were taken perpendicularly to the bole axis at a 45° angle to slope aspect on opposite sides of sample trees. The increment corer was cleaned between trees to avoid contamination. A subset of four trees were selected for additional core extractions taken from halfway between base of live crown, at live crown and halfway between tree top and live crown. These cores were used for analysis of vertical variation in wood density. Cores were then placed in straws, sealed and labeled.

Ten cores for paired carbon and density analysis were cut in half lengthwise and divided up into three categories: 1) juvenile heartwood (first seven growth rings from pith); 2) mature heartwood (three growth rings away from sapwood heartwood boundary and showing no curvature of the growth rings); and 3) sapwood (determined by color). This resulted in 30 samples analyzed for density (10 of each wood type) with corresponding halves of segments analyzed for carbon. Ninety-six additional core segments were selected strictly for carbon analysis, however, these samples were not cut in half.

Core segment samples were ground into a fine powder which was placed into a sample tube with a two-stage cap to either permit or exclude airflow. To determine if differences in air-drying and oven-drying affected carbon fraction the tubes and wood powder were first air-dried in a vacuum chamber with desiccant then analyzed for carbon fraction. The same sample tubes were later oven dried at 65 ºC and then again at 105 ºC and carbon analysis was repeated each time to determine any potential loss of carbon due to volatilization of organic compounds.

Carbon fraction

Carbon fraction was determined by taking a 5 mg sample of wood powder from each of the 126 (42 of each wood type) sample tubes for the analysis of air-dried samples (Cai) then another 5 mg sample was taken from the same tubes after oven drying for the oven-dried carbon (Cov) samples at 65 ºC then again at 105 ºC. Mass lost due to oven drying was recorded for each sample at each stage. Each sample of wood powder was placed into a clean dry tin container (Costech Analytical Technologies, Inc.) that was first zeroed on a Metler Toledo microbalance. A CE Instruments NC 2100 Elemental Analyzer (Rodano, Milano, Italy) was used to quantify total carbon of each sample on a mass per mass basis using a calibration curve with an $r^2 > 0.9999$. To ensure correct carbon values the air dried carbon fractions were adjusted by dividing the initial Cai value by one minus the result of subtracting the volatile carbon fraction (Cv) from the total mass lost fraction. Effectively this adjusted the initial carbon values to reflect the amount of residual moisture left in the wood powder.

Wood density

The volume of the samples (Vs) used to determine the density of wood was determined by zeroing a graduated cylinder of deionized water with a submerged capsule used to hold samples on an OHAUS Analytical Plus scale. The wood samples were then placed into the capsule and submerged into the water making sure to lower the capsule to the same level as it was zeroed. The reading on the OHAUS scale was recorded as the volume of the sample after determining the proper density of the water at room temperature (20 ºC), which was done by treating the scale reading as unitless and multiplying that reading by the density of water at 20 ºC (0.998199 g/cm^3) (Aleksandrov et al. 2004). Density samples were oven-dried at 105 ºC until a constant mass was achieved. Once the samples were stable their masses

(Ms) were recorded and the density determined by calculating (Ms+Cv*Ms)/Vs. This density value adjusts for the mass of volatile carbon lost during oven drying.

Statistical methods

The mean and standard error of carbon fraction for 42 samples for each of the three wood types were determined. Variation in carbon fraction was tested using a one-way analysis of variance ($\alpha = 0.01$) and significance of differences between the means of each wood type was determined using a Tukey-Kramer t-test ($\alpha=0.01$). The mean and standard deviation of 10 replicates for each wood type were calculated and a one-way analysis of variance ($\alpha = 0.05$) was used to test for differences in density among wood types. A z-test was used to determine the significance of differences in the overall mean from 0.34 g/cm^3. The mean for Cai and the mean of Cov were compared using a paired t-test. Linear regression was used to test relationships between carbon fraction of each wood type and growth rate, diameter at breast height, height, height to live crown and length of live crown. One standard error is represented by values in parenthesis within the text and tables.

Results

Variation in carbon fraction between wood types

Analysis of carbon fraction for the three wood types showed that heartwood was consistently higher in carbon than sapwood and that mature heartwood was higher in carbon than juvenile heartwood samples (*table 1*). Means were significantly different between the three wood types using a Tukey-Kramer t-test ($p<0.0001$) and all were significantly different from a mean of 0.50 ($p<0.0001$).

Table 1—*Carbon fractions and associated standard errors for each wood type - juvenile heartwood (HWj), mature heartwood (HWm) and sapwood (Sw) - for two drying methods, air dried and oven dried to 105 ºC. Forty two samples were used for each wood type for each drying method.*

Drying	HWj	HWm	Sw
Oven dried	0.5179 (0.00096)	0.5235 (0.00098)	0.5047 (0.00096)
Air Dried	0.5371 (0.00107)	0.5443 (0.00104)	0.5305 (0.00105)

Linear regression of Cai on the tested variables showed a significant negative relationship between carbon fraction of mature heartwood and average growth rate ($p =0.026$, $r^2 = 0.134$)). The relationship is described by the following equation: Cai (HWm) = 55.051 – 1.503*AveGrowth. AveGrowth is the average growth of the sample in cm/growth ring. No significant relationships existed between carbon fraction of juvenile heartwood or sapwood and any of the variables tested. Linear regression of Cov on the above tree variables showed no significant relationships for any of the tested variables. An ANOVA on wood types and carbon fraction resulted in an adjusted r^2 of 0.41 for Cai and 0.64 for Cov ($p<0.0001$).

Oven-dried versus air-dried methods

A paired t-test showed significant differences ($p<0.0001$) in carbon fractions between air-dried wood powder samples and oven-dried wood powder samples (*table*

1). Mean differences for carbon fractions between air and oven dried samples were: 0.0192 for HWj, 0.0208 for HWm and 0.0258. The volatile fraction between air dried and oven dried carbon samples (to 65 ºC) was not significantly different from the volatile carbon fraction of samples dried at 105 ºC therefore the 105 ºC values are reported for incorporation with wood density dried to the same temperature.

Density of wood types and overall wood density

Wood density was significantly different for sapwood found above the live crown (ALC) versus below live crown (BLC) but not significantly different for heartwood by location in tree or between juvenile and mature heartwood. The mean wood densities of sapwood by location in tree (ALC or BLC) were: 0.300 g/cm^3 (0.017) ALC, and 0.372 g/cm^3 (0.017) BLC. The mean wood density for heartwood was 0.382 (0.010). The mean for heartwood and below crown sapwood were significantly (p=0.010, α=0.05) higher than the 0.34 g/cm^3 value for bone-dry density of redwood. The average of sapwood density sample values resulted in a mean of 0.351 g/cm^3 (0.014).

Carbon density comparison

The average values for the combination of carbon fraction and wood density (carbon density) are given below (*table 2)*. Significant differences existed between directly measured values of carbon density and the default values for carbon density obtained by multiplying a wood density for redwood of 0.34 g/cm^3 by the default carbon fraction of 0.50 (p<0.05). Only the ALC sapwood carbon density was close to the resulting value (default value).

Table 2—*Comparison between default and measured values for carbon densities and standard errors for two wood types found within the boles of young growth redwood trees measured above live crown (ALC) and below live crown (BLC).*

Wood type	Location	Default value (g C/cm^3)	Measured values (g C/cm^3)	Difference (%)
Heartwood	ALC	0.17	0.2052 (0.0037)	20.70
Sapwood	ALC	0.17	0.1591 (0.0061)	-6.41
Heartwood	BLC	0.17	0.2076 (0.0037)	22.12
Sapwood	BLC	0.17	0.1973 (0.0074)	16.06

Discussion

Carbon fraction variation

The significant variation in carbon fraction between the three wood types indicated that using an overall average for carbon fraction, even a species specific one, may lead to underestimates of the carbon fraction of young growth redwood. There are few studies that have directly measured carbon fraction in trees (Chow and Rolfe 1989, Elias and Potvin 2003, Lamlom and Savidge 2003, Lamlom and Savidge 2006, Thomas and Malczewski 2007), potentially explaining the reliance on a general carbon fraction of 0.50 to convert dry biomass to carbon mass despite the inherent inaccuracy of utilizing such an average to predict carbon fractions of unmeasured species. The overall measured mean of carbon fraction in young growth redwood is significantly greater than the 0.50 value that is often used for trees (Brown 2002, Goodale et al. 2002, Harmon et al. 1990, Pacala et al. 2001, Van Deusen and Roesch 2011).

The differences in wood characteristics such as density, shearing strength and extractive content between old-growth redwood and young growth redwood (Resch and Arganbright 1968), make extrapolation of the values from this study to old-growth redwood inadvisable. The higher extractive content in old-growth redwood would most likely result in higher carbon fraction of the stem compared to young growth as extractives can be made up of as much as 66 percent carbon (Resch and Arganbright 1968). Wood type explained a good deal of the variation found in the tree boles with an adjusted r^2 of 0.41 for Cai predicted by wood type and an adjusted r^2 of 0.64 for Cov predicted by wood type. The difference in r^2 between the two drying methods indicated volatile compounds may vary independently from structural carbon components such as lignin and cellulose. Though the causes of carbon fraction variation within tree boles are not well understood (Elias and Potvin 2002, Lamlom and Savidge 2003), it is clear from the data that volatile organic compounds are a significant portion of the overall carbon fraction.

Densities

Differences in wood density between wood types in redwood have been reported (Luxford and Markwardt 1932, Resch and Arganbright, 1968) but were not detected with our small sample size. Significant differences in wood densities from 0.34 g/cm^3 ($p<0.05$) imply that utilization of an average density for young growth redwood may not be appropriate for all regions or for all stand structures. More studies into the factors controlling wood density in young growth redwood are needed in order to create accurate wood density prediction models to aid in predicting overall carbon stocks. The correlation of lower wood density to position in live crown of young growth redwood was not unexpected. Resch and Arganbright (1968) and Sillett and others (2010) found similar trends in old growth redwood implying that this trend is a species trait. The wood densities found in the study area are close to the 0.38 g/cm^3 value reported by Luxford and Markwardt (1932) for Mendocino implying that despite the relatively small sample size the wood density numbers found in this study may be representative for the region.

Loss of carbon during oven drying

The loss of carbon during oven drying at 65 ºC and 105 ºC is a significant finding. The methodology for carbon measurement in woody tissues commonly calls for oven drying (Lamlom and Savidge 2003). Studies have found that oven drying can lead to the loss of carbon, however, they have either used very high temperatures (Lamlom and Savidge 2003, Thomas and Malczewski 2007) or have not directly measured differences in total carbon before and after oven drying (Beakler et al. 2007). Oven drying wood at temperatures as low as 65 ºC and potentially lower (Beakler and others 2007) can lead to the loss of carbon. This suggests that oven drying should not be used in determining carbon fraction. Instead utilizing vacuum desiccation as outlined by Lamlom and Savidge (2003) or freeze drying as outlined by Thomas and Malczewski (2007) would be most appropriate. Kiln dried wood is typically processed at temperatures near 65 ºC (Beakler and others 2007) so the oven-dried carbon fractions from this study is appropriate for estimating carbon budgets for young growth redwood lumber. The oven dried values from this study are close to the value of 0.512 used by Wilson et al. (2010) indicating some consistency in carbon fraction of young growth kiln dried redwood. Timber production has been proposed as a potential method for increasing the amount of carbon stored by a given area of forest land (Skog 2008), however, it is important to understand the differences between carbon fractions of kiln dried lumber and that of live trees for accurate carbon accounting.

Improved methodology

The methodology presented in this study resulted in more precise and accurate carbon fraction values then the standard methodology. Measuring both carbon fraction and wood density is necessary in determining accurate values for a given species or location as these characteristics can be highly variable across species and regions (Chave et al. 2009, Lamlom and Savidge 2006). Lamlom and Savidge's (2003) methodology works well in a laboratory setting but the destructive nature of the methodology does not allow for comparison of carbon fractions over time. The ability to measure stand conditions over time along with carbon fraction and wood density, could be very helpful in explaining some of the environmental factors impacting those two variables. It is important that future studies into carbon fractions of tree species utilize either the vacuum desiccation methods suggested by Lamlom and Savidge (2003) or the freeze drying method described by Thomas and Malczewski (2007). Additionally the loss of carbon due to oven drying must be accounted for both in total carbon fractions and in the wood density values in order to ensure accurate carbon stock estimates.

Comparison of measured values to default values

The difference between the values suggested by the CAR protocols (0.17 g C/cm^3) and the values measured directly in this study were as high as 22 percent of the default estimate (*table 2*). The debate over improving carbon budget estimation seems to focus almost entirely on improvements in volume estimation (Jenkins et al 2003, Van Deusen and Roesch 2011). This study, however, clearly indicates the magnitude of the error that can be introduced into carbon budget estimates if variation in carbon and wood density are not addressed. The current methodology (CAR) to estimate forest carbon budgets in redwood calls for the use of values that underestimate the amount of carbon stored within redwood trees. It is likely that the default values for other species could result in overestimates of their respective carbon budgets. For example it is known that hardwoods have carbon fractions that are typically lower than 0.50 (Lamlom and Savidge 2003) and therefore utilizing a default value of 0.50 for estimates of hardwood tree carbon will systematically overestimate the amount of carbon contained within many hardwood species. Additionally there is the likelihood that tree species within some regions will have lower wood densities than the species average and therefore local estimates of carbon sequestration rates will be biased when the species average us used. To alleviate these potential biases it will be necessary to measure carbon fractions of tree species found in California and to accurately estimate the regional and stand level variations in wood density that are known to occur.

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Variable-Density Thinning in Coast Redwood: a Comparison of Marking Strategies to Attain Stand Variability

Kevin L. O'Hara[1], Lathrop P. Leonard[2], and Christopher R. Keyes[3]

Abstract

Variable-density thinning (VDT) is an emerging thinning method that attempts to enhance stand structural heterogeneity by deliberately thinning at different intensities throughout a stand. VDT may create stands with dense areas, open areas, and other areas that may be intermediate in density. Subsequent stand development forms a more varied structure than is common in many even-aged forest stands. VDT is becoming a treatment of choice in many restoration efforts where even-aged stands are being directed on trajectories towards old forest stand structure conditions because: 1) VDT enhances structural heterogeneity, and 2) a reduction in tree density provides for an accelerated development towards old forest conditions. A primary difficulty in marking VDT treatments is systematically attaining the variability necessary in a VDT prescription. Instead, markers/thinners are seemingly trained to implement a prescription by applying a uniform treatment across a stand that enhances structural homogeneity.

In coast redwood (*Sequoia sempervirens*), VDT has become the primary restoration treatment for young stands within the national and state parks of Humboldt and Del Norte Counties. These stands are young and even-aged developing following clearcutting by previous industrial landowners. VDT is being used to increase structural heterogeneity, increase the proportion of redwood or other conifers, or accelerate development towards old forest structures. Six marking/thinning prescriptions are described that have been used to date to achieve the VDT objectives in coast redwood:

1) "Randomized grid – moderate density": 371 tph (150 tpa). Required marker to visualize grid area and randomly assign 0 to 3 trees for each grid area (more fully described in O'Hara et al. 2010). Used in research study to achieve target densities. Post-treatment densities ranged from 346 to 558 tph in three treatment areas (blocks). Labor intensive to use on operational basis.
2) "Randomized grid – low density": 185 tph (75 tpa). Same as number 1, but lower target density. Attempted to achieve target restoration density in one treatment. Lower density achieved by marker visualizing larger grid but still randomly assigning 0 to 3 trees to each grid area. Post-treatment densities ranged from 192 to 331 tph (more fully described in O'Hara et al. 2010).
3) "Dx rule" – in predominantly coast Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) stands. Largest Douglas-fir greater than 12.7 cm is identified, diameter in inches is multiplied by 2, and all trees between 12.7 and 24.5 cm are cut within that radius (feet) but not more than 6.1 m (20 ft). Performed on two contrasting sites where resultant densities were 238 and 576 tph. Feasible for thinners to implement without marking and allows heavier cutting of broadleaved trees (more fully described by Keyes et al. 2010).
4) "16 x 16 ft" or 4.9 x 4.9 m: Thinning attempted to leave most desirable trees on a 4.9 x 4.9 m ($\pm$ 1.3 m). Did not cut any redwood stump sprouts and counted a redwood sprout

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clump as one tree. Easy for thinners familiar with standard precommercial thinning methods. Easy to achieve target density but relatively low variability in spacing (*table 1*).

5) "20 x 20 ft" or 6.1 x 6.1 m: Thinning attempted to leave most desirable trees on a 6.1 x 6.1 ($\pm$ 1.3 m). Similar attributes as "16 x 16 ft" treatment but achieved lower density and greater variability (*table 1*).
6) "Localized release": Thinners cut 7.6 m (25 ft) circles leaving three best retention trees in any position in circle. Spaces between two circles thinned to 3.7 x 3.7 m spacing. Spaces between three or four circles left unthinned. Did not cut any redwood stump sprouts and counted a redwood sprout clump as one tree. Achieved highly variable spacing (*table 1*) and approximately 112 tph with full exposure. More difficult to implement for thinners.

The coefficient of variation is used to provide a measure of variability for trees per ha between sample plots (*table 1*). Controls represent either unthinned controls for all six methods except Panther Creek where the controls were the pretreatment densities. All six methods resulted in an increase in the coefficient of variation indicating an increase in stand level spatial heterogeneity. O'Hara et al. (2010) also showed increased tree size variability with both the medium and low density treatments with the "randomized grid" method.

Key words: *Sequoia sempervirens*, variable-density thinning, precommercial thinning, intermediate operations, timber stand improvement

Table 1—*Coefficient of variation (CV) for the six prescriptions/marking tools. CV was calculated as the average of block CVs and the number of blocks is shown as the sample size (n). CVs for untreated stands are shown except for Panther Creek which is a pretreatment CV. Other prescriptions share the same untreated stand data.*

Prescription	Site	CV treated stands	n	CV untreated stands or before treatment	n
"Randomized grid moderate density"	Mill Creek	0.390	5	0.245	5
"Randomized grid low density"	Mill Creek	0.432	9		
"Dx Rule"	Panther Creek	0.780	2	0.488	2
"16 x 16 ft"	Mill Creek	0.179	5		
"20 x 20 ft"	Mill Creek	0.208	5	0.166	5
"Localized release"	Mill Creek	0.334	5		

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The Scotia Plantation: Implications for Multiaged and Even-aged Silviculture

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Abstract

The Scotia Plantation was established in 1982 on the large alluvial flat south of Scotia and adjacent to the Eel River. Seedlings, from local "woods run" seed sources, were planted on a 3.1 x 3.1 m (10 x 10 ft) grid. Site quality was very high, with site index averaging greater than 45 m (50 yr base). In 1997, the area was divided into blocks and a series of thinning treatments were randomly assigned to these blocks. Thinning treatments included alternate row, diagonal row, every-third row, a double-alternate row, and a low density treatment. Plots were established and measured in the winter of 2002 to20003 and remeasured in 2007 after 5 growing seasons.

Stump sprout development following thinning treatments was very sensitive to available light resources. In the most heavily thinned treatment, the largest understory sprouts were up to 12.8 m (42 ft) in height and 12.7 cm (5 in) at dbh in only 10 years. However, at low light intensities many sprout clumps were completely dead. For example, in 2007 after 5 years, clump survival was only 9 percent in the third row treatment, but 100 percent in the low density treatment (O'Hara and Berrill 2010). Implications for multiaged silvicultural strategies suggest that treatments that create stump sprouts may not be sufficient by themselves for regenerating a new age class; additionally, light levels must also be sufficient to allow these sprouts to grow and survive. Our results suggest a light level of 40 percent of full sunlight is needed to sustain sprout height growth approaching 1 m/yr and light levels of 10 percent of full sunlight is needed for sprout survival. These sprout development results are published in more detail in O'Hara et al. (2007) and mortality patterns in O'Hara and Berrill (2010).

Volume increment for the Scotia Plantation was comparable to the fastest growing conifer plantations in the world. Periodic annual increment (PAI) ranged from 42.9 to72.6 m^3/ha/yr (613 to 1082 ft^3/ac/yr) depending on treatment (*table 1*). Mean annual increment (MAI) ranged from 15.4 to 40.0 m^3/ha/yr (220 to 570 ft^3/ac/yr); MAI was considerably lower than PAI suggesting MAI was far from culmination and that these values underestimate potential productivity for the Scotia Plantation. These results are comparable to the greatest production from other conifer and broadleaf plantations around the world and demonstrate the enormous production potential of coast redwood.

Key words: *Sequoia sempervirens*, thinning, plantation management, forest production, coppice, sprout development

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Table 1*—Volume production of four thinning treatments and one control from the Scotia Plantation through age 21 and 25 years. Values are means from 2 to 3 plots per treatment.*

Treatment	MAI (21 yrs) (m^3/ha/yr)	MAI (25 yrs) (m^3/ha/yr)	PAI (2003-06) (m^3/ha/yr)
Low density	10.3	15.4	42.9
Diagonal row	21.6	28.6	65.2
Alternate row	18.3	25.1	60.8
3rd row	21.2	23.8	65.7
Control	33.2	40.0	75.6

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Coast Redwood Responses to Pruning

Kevin L. O'Hara[1]

Abstract

A large-scale pruning study was established in the winter of 1999 to2000 at seven different sites on Green Diamond Resource Company forestlands in Humboldt County. The objective of this study was to determine the effects of pruning on increment, epicormic sprouting, stem taper, heartwood formation, and bear damage on these young trees. Pruning treatments varied pruning severity and were usually applied in conjunction with thinning treatments. Trees were assessed six years after pruning. Basal area increment decreased with increasing pruning severity but results were inconsistent from one study site to another. Height increment was unaffected by pruning. Six-year volume increment results resembled those for basal area increment: heavier pruning sometimes resulted in lower increment. Additionally, the negative effects of pruning on tree increment were probably short-lived in redwood because of the fast growth rates of this species. The number of epicormic sprouts was generally unaffected by pruning severity with the notable exception of the most severe pruning treatments. By year six, the number of sprouts was no different in the unpruned treatments than in most pruned treatments. The exception was the severe crown removal that left only approximately 15 percent residual live crown length. Epicormic sprouting does not appear to be a deterrent to pruning in redwood. Tree stem taper was also unaffected by pruning severity. Heartwood formation was expected to increase with pruning severity. However, no effects were evident in these data. Apparently, the greater heartwood expected in more severely pruned treatments was obscured by the same factors that minimized treatment effects on increment: in the six years following treatment, the pruned trees had rebuilt crown foliage and required similar sapwood for water transport as unpruned trees. Bear damage was observed at four of the seven study sites and was severe in several plots. However, no trends were evident relative to pruning treatment. In summary, pruning that leaves residual crown lengths of 40 to 60 percent should result in minimal levels of epicormic sprouting and no effects on tree increment.

Key words: *Sequoia sempervirens*, forest pruning, timber stand improvement, silviculture, wood quality

Introduction

Forest pruning is a common means to enhance wood quality in young forest stands. By removing lower branches, wood is formed over the branch wound that is clear and usually straight-grained. This clearwood is generally expected to produce higher grade wood products and earn a premium. Pruning may also improve stem taper from pruned logs and enhance heartwood development. In some species, such as coast redwood (*Sequoia sempervirens*), heartwood has desirable appearance and decay resistance properties which make it more valuable than sapwood.

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Although the effects of pruning forest trees are generally positive, it can lead to excessive epicormic branch production in some species, including coast redwood. Epicormic sprouts respond to increased light or temperature on the pruned stem resulting in a profusion of branches. These can negate the value of the pruning treatment by producing wood quality defects in the clear wood produced in the outer tree boles.

No previous study has provided an analysis of the effects of pruning on coast redwood. This paper describes a six-year study to look at the effect of pruning on tree increment, stem taper, epicormic sprouting, heartwood development, and bear damage.

Project description

Pruning plots were established during the winter of 1999 to 2000 in seven stands (study sites) that received pruning treatments in the fall of 1999. All seven study sites were located in Humboldt County, California on Green Diamond land. These plots were designed to measure the effects of pruning on individual trees. Hence plots were assemblages of approximately 30 trees per treatment rather than being of fixed area. All study sites included control plots and some included more than one pruning treatment. Of the seven study sites, all but two were designed to represent stand-level, operational pruning treatments. The other two study sites – *299 Cutoff* and *M-line* – provided variable pruning treatments over small areas, but not on an operational scale. At time of initial measurement, trees were tagged, and marked at dbh. Measurements taken at time of plot establishment included dbh, height, height to live crown after pruning, height to base of live crown on unpruned trees, counts of epicormic sprouts, and some upper stem diameter measurements (either at 2.7 or 5.5 m height). Additionally, trees of common stump origin were noted. Tree characteristics at first measurement are presented *in table 1*.

The plot remeasurements and data analysis in 2001/02 and 2006 included studies of epicormic sprouting, increment, heartwood formation and stem taper. Analysis of variance was used to compare treatment means within study sites. When ANOVA indicated significant differences between treatment means, Tukey's HSD multiple range tests were used to identify significantly different means.

Growth response studies

Trees were remeasured in 2001 and 2002 and spring 2006 for height and dbh. Nearly all trees were also measured for diameter at tree height of 2.7 m (9 ft). The original plot measurements in 1999 to 2000 included height and dbh measurements, but not all trees were measured for the 2.7 m upper stem diameter measurement. Cubic volumes were estimated assuming the tree base to bh was a cylinder, bh to 2.7 m a cone frustum, and above 2.7 m as a cone for smaller trees. Equations from Krumland et al. (1977) were used for larger trees. Relative growth was calculated as the 6-year volume increment divided by volume in 1999 to 2000.

Results resemble those from pruning studies with other species except for an apparent lack of sensitivity to more severe pruning treatments in redwood than has been observed in other pruning studies (*table 2*). In other species, a strong decrease in increment with increasing pruning severity is usually found (O'Hara et al. 1995). In terms of plot comparisons, there were abnormalities in both the *299 Cutoff* and *M-Line* study sites. At *299 Cutoff*, the control plot had much greater increment than expected. At M-Line, the 15 percent residual treatment had greater increment than expected.

Table 1—*Average pruning heights, initial tree heights and diameters by plot and study site (pruning heights for control treatments are live crown heights at beginning of study) Average residual crown lengths show actual average crown lengths following pruning.*

Study Site	Treatment (residual crown length in percent of total height)	Average tree height in 1999-00 (m)	Average residual crown length (%)	Avg dbh in 1999-00 (cm)
299 Cutoff	Control (E)	9.6	71	14.2
	60% (D)	9.4	56	12.3
	45% (C)	10.4	46	13.9
	5.5m or 35% (F)	9.6	47	14.2
	30% (B)	9.0	35	12.0
	15% (A)	9.7	24	14.5
M-line	Control (D)	7.4	80	9.9
	60% (E)	6.0	57	7.6
	45% (C)	7.6	42	11.3
	30% (A)	8.3	39	11.8
	15% (B)	6.8	22	9.7
Maple Creek	Control	9.7	78	13.2
	4.9-5.5m	10.8	53	17.2
Fortuna	Control	12.5	-	21.1
	3-5.5m, 40% (A)	11.9	54	24.6
Carlotta	Control (E)	10.7	67	14.5
	2.4m (D)	10.4	65	16.0
	3-5.5m, 40% (B)	10.2	49	14.9
	3-5.5m, 40% (C)	11.6	46	18.6
M-150	Control (B)	10.2	80	13.4
	Control (C)	10.1	78	12.1
	2.4m (B)	10.4	72	13.8
	3-5.5m, 40% (A)	8.9	43	12.3
Mitsui	Control (C)	7.6	81	11.2
	2.4m (A)	7.2	67	10.2
	3-5.5m, 40% (B)	7.4	49	13.2

Basal area increment/tree was largely unaffected by pruning intensity. Many study sites experienced increased basal area increment following pruning as compared to controls (*Maple Creek*, *Fortuna*, and to a lesser extent *M-150*). However, at *299 Cutoff*, the control basal area increment/tree was considerably greater than the pruned treatments. At *M-line*, no trend was evident and the 15 percent residual crown treatment had virtually the same basal area increment as the control. This suggests a combination of site irregularities across plots and the possible influence of more productive clones on some plots. For example, it is possible that one plot could be dominated by the sprouts of one stump that is very productive, whereas the next plot is dominated by a less productive sprout clump. An additional factor was pretreatment tree size variability and its affect on growth. Growth was therefore also expressed as "relative growth" by dividing 6-year basal area increment per tree by pretreatment basal area. This analysis produced substantially different results than the absolute basal area increment/tree analysis. For the *299 Cutoff* study site, the highest relative growth was in the 30 percent residual treatment whereas at *M-line* it was the 60 percent residual.

There were few significant differences in height increment among study plots. This result was expected as only very severe pruning treatments generally affect height increment. Significant differences were found at other study sites between control and pruned height increment, but these may have been the result of bear damage preferentially affecting taller trees thereby reducing average increment.

Cubic volume increment followed similar trends as basal area and height increment (*fig. 1, table 2*). Statistical significance was only found at Mitsui and these results were influenced by bear damage. Relative volume increment (volume increment per unit of pretreatment volume) was statistically significant for some treatments/study sites. For example, at *299 Cutoff*, the 30 percent residual treatment was significantly greater than the 15 percent and 35 percent treatment. This suggests an affect related to clonal patterns within measurement plots rather than a positive effect of that particular pruning treatment.

In summary, pruning apparently has relatively minor effects on increment in coast redwood as represented by basal area increment, height increment, and volume increment. There is some evidence of reduced increment with the most severe pruning treatments, but for the operational range in pruning severity (> 35 percent residual live crown) there is no effect 6 years after pruning.

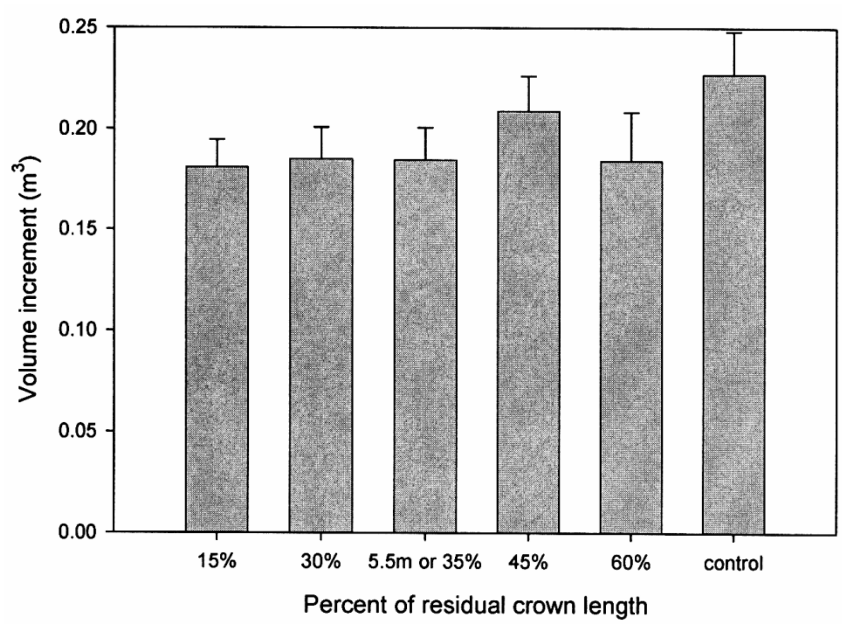

Figure 1—Cubic volume response after six years at *299 Cutoff.*

Table 2—*Six-year cubic volume increment and relative growth rates. Relative growth rates were volume increment divided by volume at time of pruning. Means followed by common letters within a study site were not significantly different (α=.05). Standard errors (SE) are given in parentheses following means.*

Study Site	Treatment (residual crown length in percent of total height)	Avg. volume growth (m^3)	Relative volume increment	Sample size
299 Cutoff	Control (E)	0.226a	3.8ab	31
	60% (D)	0.184a	3.5ab	31
	45% (C)	0.209a	3.1ab	30
	5.5m or 35% (F)	0.184a	3.0a	30
	30% (B)	0.184a	4.4b	31
	15% (A)	0.181	2.6a	31
M-line	Control (D)	0.088a	3.5a	25
	60% (E)	0.062a	3.8a	25
	45% (C)	0.065a	3.6a	33
	30% (A)	0.065a	2.1a	28
	15% (B)	0.076a	3.4a	30
Maple Creek	Control	0.042a	1.6a	58
	4.9-5.5m	0.051a	1.2b	61
M-150	Control (B)	0.105a	2.0a	13
	Control (C)	0.082a	1.6a	21
	2.4m (B)	0.133a	2.3a	24
	3-5.5m, 40% (A)	0.074a	1.8a	21
Mitsui	Control (C)	0.082a	2.7a	22
	2.4m (A)	0.074ab	2.8a	40
	3-5.5m, 40% (B)	0.031b	1.4b	29

Epicormic sprout studies

Approximately 10 trees per plot were randomly selected in 2002 for epicormic sprout sampling. For pruned trees, the pruned section of the bole was divided into six sections and one section was randomly selected for sampling and marked with paint. For example, a tree pruned to 3 m would have six 0.5 m sections and one of these sections would be randomly selected for measurement. In these sections, all sprouts were counted and the highest and lowest sprouts in the section were measured for length and caliper (diameter at base). The estimated number of sprouts per tree was the section count multiplied by six. Unpruned trees were sampled in a similar way: the section of bole, equal to 5.5 m or the highest level pruned in adjacent pruned plots, was divided into six sections and one section was randomly selected for measurement. On control plot trees, only sprouts that originated at the time of the pruning or after the pruning were counted. This may be a potential source of error as sprout age was difficult to determine. Caliper measurements on small sprouts may also be biased because the presence of scale-like leaves at the base of these sprouts may have exaggerated their diameter. The presence of leaves on larger sprouts had little effect on caliper measurements. This sampling procedure was followed in both 2002 and 2006. The same stem section was therefore sampled at each measurement. Because of sprout mortality between 2002 and 2006, however, the same sprouts may not have been measured in both years.

The estimated number of sprouts per tree was unaffected by pruning severity except at most severe pruning (*table 2*). Numbers of sprouts declined from 2002 to 2006. *Figure 2* shows the relationship between estimated number of sprouts and residual crown length for 2006 at *299 Cutoff*. A flat trend in number of sprouts is apparent until a sharp increase with the 15 percent pruning. After 6 years, there were no significant differences in numbers of sprouts in any of the study sites except the most severe treatments at the *299 Cutoff* and *M-Line*. However, variance was large in these samples because some trees appeared to be much more prone to sprouting than other trees in the same treatment. There may also have been a clonal component to these results where one clone dominated a plot and had a disparate influence on the plot average. In a separate analysis of these data, O'Hara and Berrill (2009) showed a clonal effect to sprouting where some clones were more likely to produce epicormic sprouts than others.

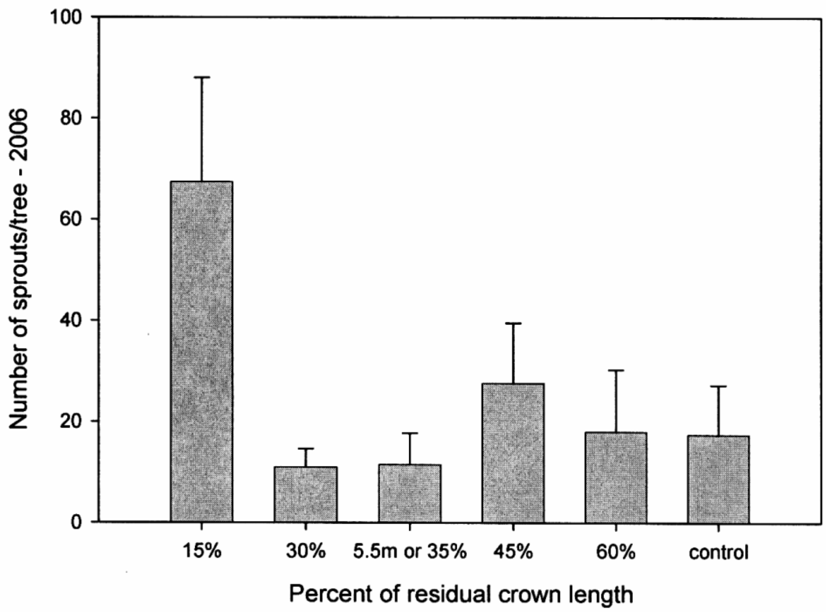

Figure 2—Estimated number of sprouts per tree in 2006 for treatments at the *299 Cutoff* study site. The "5.5 m or 35 percent" residual crown length treatment was pruned to 5.5 m (18 ft) lift or no less than 35 percent residual live crown.

Larger sprout length and diameter is indicative of the size of the resulting wood quality defect, and probably indicative of sprout persistence over time. Larger sprouts were found on the most severely pruned trees which suggested these sprouts are a more important component of photosynthesis needs and are therefore likely to persist. This result is consistent with other species including giant sequoia (O'Hara et al. 2008). In this study, more severe pruning treatments resulted in a large increase in average sprout size in most study sites, and particularly in the *299 Cutoff* and *M-Line* study sites that included severely pruned trees. For example, at *299 Cutoff*, the average sprout caliper in the 15 percent residual treatment was five times the average in the 60 percent residual treatment. Because of the decline in number of sprouts from 2002 to 2006 (*table 3*), the sample sizes for sprout measurements also declined during this period. This contributed to the high variation in sprout sizes, the lack of consistent trends in sprout size with increasing pruning severity, and the lack of statistical significance when means were quite different.

It should be noted that trees in unpruned plots also produced epicormic sprouts. Some of these plots received precommercial thinning treatments at the time of pruning while others were thinned up to three years before pruning. This indicates that some background level of sprouting is present on young redwood trees. In

Table 3—*Epicormic sprout data for pruned and unpruned plots in 2002 and 2006. The Carlotta plot D was not measured for epicormic sprouts because of extensive bear damage. Common lower case letters within study sites denote means that were not significantly different. Standard errors (SE) are given in parentheses following means.*

Study Site	Treatment (residual crown length in percent of total height)	Avg number of sprouts per tree 2002	Sample size 2002	Avg number of sprouts per tree 2006	Sample size 2006
299 Cutoff	Control (E)	27.0 (11.9)a	10	17.4 (9.8)a	10
	60% (D)	28.6 (23.6)a	9	18.0 (12.3)a	9
	45% (C)	73.7 (33.8)ab	10	27.6 (12.0)a	10
	5.5m or 35% (F)	30.0 (13.5)a	10	11.5 (6.2)a	9
	30% (B)	38.5 (9.0)ab	10	11.0 (3.7)a	10
	15% (A)	154.3 (37.2)b	17	67.3 (20.6)a	17
M-line	Control (D)	6.0 (4.8)a	10	3.6 (2.6)a	10
	60% (E)	18.0 (10.2)a	10	3.6 (3.6)a	10
	45% (C)	44.0 (18.5)a	10	16.2 (8.3)a	10
	30% (A)	76.7 (19.2)a	8	13.7 (5.4)a	7
	15% (B)	342.7 (128.4)b	10	169.0 (75.7)b	10
Maple Creek	Control	21.0 (6.8)a	12	6.0 (2.3)a	12
	4.9-5.5m	38.8 (15.6)a	11	12.3 (5.8)a	11
Fortuna	Control	6.0	1	0	1
	3.0-5.5m, 40% (A)	4.5 (1.9)	8	7.3 (5.3)	8
Carlotta	Control (E)	34.7 (12.5)ab	10		
	2.4m (D)				
	3.0-5.5m, 40% (B)	24.9 (3.9)a	10	9.4 (3.6)a	10
	3.0-5.5m, 40% (C)	86.1 (24.6)b	10	21.4 (10.0)a	7
M-150	Control (B)	2.7 (2.7)a	9	1.5 (1.5)a	8
	Control (C)	2.4 (1.6)a	10	2.0 (1.3)a	6
	2.4m (B)	3.8 (2.3)a	11	2.7 (1.2)a	11
	3.0-5.5m, 40% (A)	74.9 (25.9)b	10	27.1 (20.8)a	10
Mitsui	Control (C)	19.2 (9.2)a	10	11.4 (8.4)	10
	2.4m (A)	3.0 (1.6)a	10		
	3.0-5.5m, 40% (B)	19.2 (8.0)a	10		

unpruned trees, this sprouting is not a major consideration since clearwood is not an expectation from these plantations. In pruned trees, development and persistence of sprouts, whether background level or not, is likely to cause wood quality defects and reduce potential returns from a pruning investment. However, the results in 2006 indicate no difference in level of sprouting or persistence of sprouts between the unpruned and all but the most severely pruned trees

Stem taper

A measure of stem taper was calculated by dividing tree diameter at 2.7 m (9 ft) by tree diameter at bh (1.37 m). Note that some plots were not measured at 2.7 m, or experienced so much bear damage by 2006 that samples were too small for analysis. Pruning generally reduces stem taper through reduced radial increment at the base of tree and only a minor effect further up the stem (O'Hara 1991). Results from this study showed virtually no affect of pruning on stem taper. *299 Cutoff* showed increased taper in the control but *M-Line* had the greatest taper in the most severe pruning treatment. Only the results at *299 Cutoff* were statistically significant.

These results suggest that pruning has no effect on stem form as represented by the measure of stem taper used here. An upper stem diameter measurement higher on the stem may have yielded a stem form measure more sensitive to pruning. These results were also affected by the rapid growth rates and recovery of trees following pruning that may have obscured affects of pruning on stem taper. Additionally, all plots were thinned at about the same time as the pruning treatments. Since thinning generally increases stem taper, these two opposite effects were both affecting stem form of study trees.

Heartwood formation

Sample trees were cored at time of establishment of this study, and again in 2006. These trees were randomly selected. Cores were removed at bh and were immediately measured for sapwood thickness based on the visual translucence of the sapwood. Sampling was limited to the same 10 trees/plot sampled for epicormic sprouts.

Reducing the size of the crown reduces the transpiration requirements of the tree and also the need for sapwood conducting tissues. However, contrary to expectations, pruning had no effect on heartwood formation in this study.

Possible explanations for the lack of a positive heartwood response in these data include the small sample sizes. The only post-pruning significant response was at *Maple Creek* where sample sizes were also highest. However, when ignoring statistical significance, these results do not reveal the expected trends of increasing percent heartwood with increasing pruning severity. At both *299 Cutoff* and *M-Line* the control treatments were among the highest in percent heartwood and heartwood expansion. A more likely explanation is the rapid growth of these trees over the 6 year period has allowed them to rebuild crowns (through both height growth and epicormic sprouting) therefore requiring comparable sapwood conducting tissues regardless of treatment. Affecting heartwood at final harvest would apparently require repeated pruning or more severe pruning to significantly reduce crown size at end-of-rotation.

Bear damage

Bear damage was noted in plots at both the 2002 and 2006 remeasurements. The circumference of the damage around the base the tree was estimated to the nearest 10 percent. Bear damage was found in four of the seven study sites: M-Line, Carlotta, M-150, and Mitsui. The worst damage was at Carlotta and Mitsui, but concentrated in several plots. In the case of the Mitsui study site, the plots were considerable distance from each other. There were also no patterns evident with regard to pruning treatment. At Carlotta, two of the pruning treatments experienced damage to greater than 50 percent of the trees whereas the control and another pruning treatment experienced relatively light damage. At M-150, the percent of damaged trees was constant across all treatments. The percent of bole circumference damaged is of limited value because of healing that might have taken place since the damage occurred that reduced the visible damage and the percent values were just visual estimates. Future studies to ascertain the effects of silvicultural treatments on bear damage should involve much larger treatment areas and larger measurement plots than those used in this study.

Summary and recommendations

The response of coast redwood to pruning varies from typical responses of conifers. The typical decreasing increment with increasing pruning severity pattern was not observed in these study plots despite the inclusion of very severe pruning treatments. Instead, redwood growth is less sensitive to pruning and reductions in crown size than other conifers.

Epicormic sprouting occurred following pruning and resembled patterns for other species. These included an initial sprouting response after pruning but a decline in sprouting over time. Six years after pruning, the number of sprouts in the moderate pruning treatments had declined to levels that were comparable to the unpruned controls. Pruning at moderate levels (leaving 40 to 60 percent live crown) had no effect on epicormic sprouting.

Pruning did not affect stem taper based on the ratio of diameter at 2.7 m (9 ft) and diameter at bh. A taper measure based on a higher stem diameter measure may have produced results that were consistent with other pruned conifers. Additionally, all plots were also thinned: pruned trees were therefore also reacting to the effects of thinning which generally increases taper. Similarly, there was no effect of pruning on heartwood development. The expectation was an expansion of heartwood because of reduced sapwood requirements for smaller crowns in pruned trees. The rapid recovery of foliage after pruning that contributed to the rapid growth recovery apparently required sufficient sapwood conducting tissues and negated the effect of crown reduction on heartwood development.

Several pruning plots have experienced major damage due to bear girdling. There was no apparent pattern to this damage with regard to pruning treatment. However, these small plots were too small to assess bear preferences for pruning treatments.

Without the threat of excessive bear damage to pruned trees, pruning would appear to be a viable treatment to enhance wood quality in coast redwood. The long-term effect of pruning on epicormic sprouting is minimal with moderate pruning regimes. Pruning has little effect on redwood volume or height increment making this species one of the least sensitive conifer species to pruning.

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Using FORSEE and Continuous Forest Inventory Information to Evaluate Implementation of Uneven-aged Management in Santa Cruz County Coast Redwood Forests[1]

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Abstract

Swanton Pacific Ranch in northern Santa Cruz County has been owned and managed by California Polytechnic State University (Cal Poly) Foundation since 1987. The California Forest Practice Rules specific to Santa Cruz County limit harvest rate and opening size. Cal Poly forest managers are implementing uneven-aged forest management on 1,182 acres of 80 to 110 year old, second-growth coast redwood forests using a modified BDq approach. The Lockheed Fire spread into most of the managed forest area during the summer of 2009 causing significant mortality in lower diameter classes.

Little information is available on implementation of uneven-aged forest management for coast redwood stands in Santa Cruz County especially regarding the influence of fire. This McIntire Stennis funded, observational research study used the Forest and Stand Evaluation Environment (FORSEE) program and a 22 year Continuous Forest Inventory (CFI) record to evaluate changes in past and current stand structure using trees per acre, basal area per acre, and volume per acre. Stand structural changes associated with uneven-aged management and disturbance were detected using FORSEE analysis of the CFI records. FORSEE is a useful inventory, growth and yield model that will become better with local calibration.

Key words: *Sequoia sempervirens*, coast redwood, multi-aged, BDq, FORSEE, growth and yield model, fire effects, monitoring, Santa Cruz County

Introduction

Forest management at California Polytechnic State University's Swanton Pacific Ranch and School Forest began in 1986 when owner Mr. Al Smith, requested Cal Poly assistance with management of his agricultural and forested properties which he bequeathed to Cal Poly in 1993. Mr. Smith's long-term vision focused on Swanton

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Pacific Ranch being sustainably managed as a working ranch and forest with many interdisciplinary learn-by-doing activities and projects involving students, staff and faculty.

Swanton Pacific Ranch is located in the southern sub-district of the Coast District (Cal Fire 2011). Very strict forest practice rules have been 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 1907 earthquake. These current, sub-district California Forest Practice Rules specify tree removal limits by diameter class, maximum permitted opening size, and Water Lake Protection Zone requirements.

In consultation with Big Creek Lumber Company and other forest managers, sustainable management of the working forest areas was approached through implementation of uneven-aged, modified BDq forest management practices as discussed in the 1991 forest management plan (Alexander and Edminster 1977, Baker et al. 1996, Big Creek and Cal Poly 1991, de Liocourt 1898, Guldin 1991, Leak and Gottsacker 1985, Marquis et al. 1992, Meyer 1943, Meyer 1952, O'Hara and Gersonde 2004). Two presentations at earlier Redwood Symposiums focused on initiation of forest management practices at Swanton Pacific Ranch (Piirto et al. 1996b, Piirto et al. 2007). Further, a paper on approaches to managing tanoak at the Cal Poly's Swanton Pacific Ranch was presented at the 1996 California Oak Symposium (Piirto et al. 1996a). Current forest management at Swanton Pacific Ranch is guided by the 2008 Non-Industrial Timber Management Plan (Big Creek and Cal Poly 2008).

Forest managers must consider human and ecological legacy as they develop management plans for a specific forested area. Fire, windstorms, landslides, and flooding are several examples of ecological disturbance processes that affect coast redwood forests. Fire has been reported as being a low to moderate disturbance factor that can ultimately lead to multi-aged coast redwood stands (Lorimer et al. 2009). The ecological role of fire in Santa Cruz Mountain coast redwood forests has been and continues to be the subject of a number of recent studies (Brown and Swetnam 1994, Greenlee and Langeheim 1990, Lorimer et al. 2009, Ramage et al. 2010, Stephens and Fry 2005, Stephens et al. 2004). A study by Hyytiainen and Haight (2010) reported that fire had differing effects in even-age stands (decreased optimal rotation lengths and planting densities) as contrasted to uneven-aged managed stands (reduced optimal diameter limits). The occurrence of the Lockheed Fire at Swanton Pacific Ranch in 2009 provided an opportunity to study the impact of fire on a forest under uneven-aged forest management.

Measuring key ecosystem attributes is fundamental to successful forest management. The application of ecosystem management principles requires understanding of past and present conditions (i.e., reference conditions) as desired future conditions are defined and adaptive forest management occurs (Manley et al. 1995, Piirto and Rogers 2002). Given these considerations, are we succeeding with implementation of uneven-aged forest management at Swanton Pacific Ranch? The objectives of this observational study were to:

1. Consolidate and standardize the 1989 initial forest inventory, the 1997, 2008, and 2010 (post-fire) Continuous Forest Inventory (CFI) data (*fig. 1*) into one database for use in FORSEE.
2. Utilize the inventory data and the FORSEE model to test two hypotheses:

a. Stand structural changes can be detected using a CFI System and a FORSEE Analysis Tool; and
b. Major disturbance events quantifiably influence establishment and survival of young age-classes in uneven-aged managed stands.

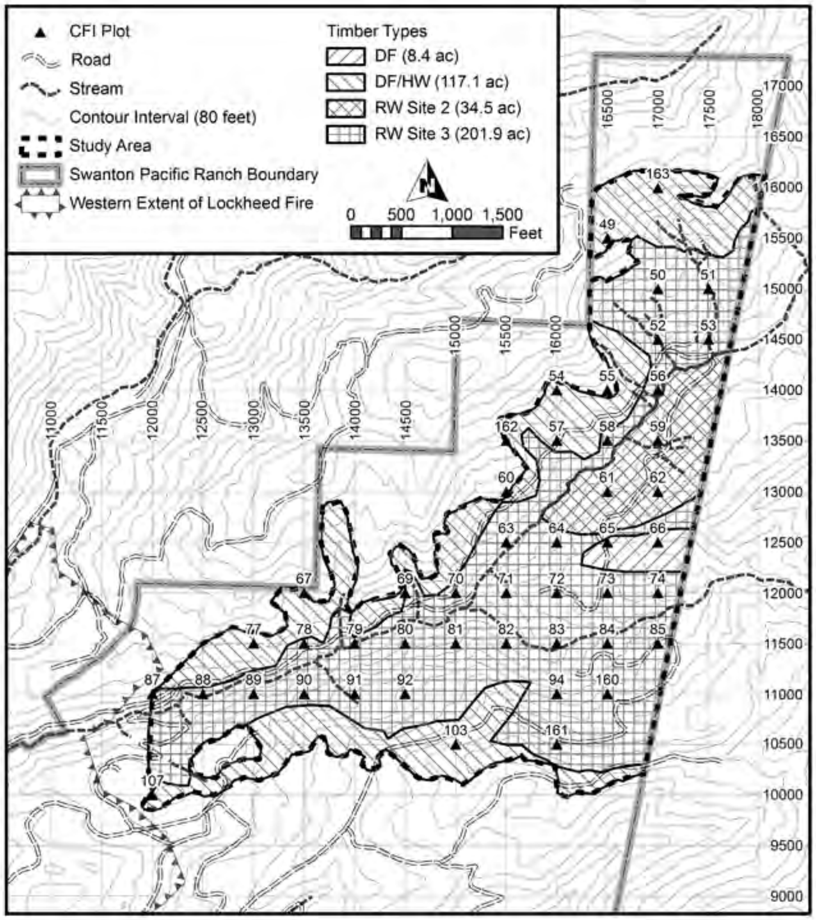

Figure 1—Swanton Pacific Ranch CFI plots and vegetation types in study area.

Methods

The process used to implement forest management, delineate the project area and map development, 1989 forest inventory methods, CFI methods (1997, 2008, 2010), data compilation, data auditing, data analysis, FORSEE calibration, and assumptions are described below.

Implementation of forest management

A modified, specified residual basal area in square feet (B), maximum retained diameter class (D), and negative exponential constant between diameter classes (q) (de Liocourt 1898, Guldin 1991, Leak and Gottsacker 1985, Meyer 1952) approach was employed to implement uneven-aged forest management at Swanton Pacific Ranch consistent with current California Forest Practice laws and associated regulations (Cal Fire 2011). Detailed prescription reports were developed for each of the stands in the Little Creek area of Swanton Pacific Ranch (Haupt and Piirto 2006, Piirto and Cross 1996 [revised 1999], Piirto and Piper 1990). Tree removal was guided by the tree marking rules with BDq adjustments made for specific stands.

Residual stand density basal area (B) target was set at 180 to 220 square feet per acre. This residual largely conifer stand density target was initially established as a percentage reduction of normal stocking as shown in Lindquist and Palley (1963) and Piirto et al. (1996b) consistent with expected growth for the next 10-year growing cycle. This basal area target range thus results in retaining 42 to 51 percent of full stocking if we assume that: 1) a Lindquist and Palley (1963) normal stand with a site index of 130 (site class III, base age 100 years) has 431 square feet and 2) that this Lindquist and Palley (1963) normal basal area value is a reasonable estimate of complete site occupancy for a 100 year old coast redwood stand. Recent work by Berrill and O'Hara (2009) and Oliver et al. (1992) discuss stocking control in multi-aged coast redwood stands.

The largest tree retained in the stand varied between 30 and 38 inches DBH in relation to individual stand objectives. Stand F (The Tranquility Stand), for example, was a favorite spot for Al Smith. He requested that we work to retain some of the larger second-growth trees in Stand F as part of the uneven-aged forest management objectives. As such, the maximum tree size in Stand F was set at 38 inches with retention of a few larger trees consistent with Mr. Smith's direction. A maximum DBH tree size of 34 inches for uneven-aged managed stands provides good yield and a sizeable individual tree heartwood core which is important to lumber grades and associated value (Piirto and others 1996b).

The negative exponential constant (i.e., in 2 inch DBH classes) was set between 1.2 and 1.3 consistent with advice received from managers at Jackson Demonstration State Forest, Guldin (1991), and as described in Long and Daniels (1990). Further research by Berrill and O'Hara (2009) utilized growth and yield (CRYPTOS) and stocking assessment (redwood MASAM) models to simulate the effects of multi-aged coast redwood management regimes and harvesting scenarios. The timber harvest projects administered by Cal Poly and Big Creek Lumber Company since 1990 are shown in *table 1* (harvest/stand area maps are available upon request).

Study area delineation

The area delineated for this study was determined using the following criteria: 1) inventoried at least twice and preferably three separate times; 2) affected by at least one timber harvest activity since 1990; and 3) affected by the 2009 Lockheed Fire. The total area within the defined study area was 361 acres. Four forest vegetation types occur within the project area: Redwood Site Class III–202 acres; Redwood Site Class II–34 acres; Douglas-fir–8 acres; and Douglas-fir-Hardwood–117 acres. The project area, vegetation types, 1989 initial inventory plots, and 1997/2008/2010 CFI plots are illustrated in *fig. 1*.

Table 1—*Swanton Pacific Ranch Little Creek timber harvest projects (1990 to 2011).*

THP No. and Date	Location	Acreage	Logging system	Gross volume			Net volume			Defect (%)	
				Redwood	Douglas-fir	Total	Redwood	Douglas-fir	Total	Redwood	Douglas-fir
THP 1-89-539 SCR											
1990	South Fork	59	T	285,810	391,260	677,070	215,530	281,930	497,460	24	28
THP 1-94-071 SCR	North Fork	120									
1994			T, C	615,790	326,830	942,620	547,790	253,000	800,790	11	22
1995			T, C	578,810	274,490	853,300	500,240	207,330	707,570	14	24
Emergency Salvage											
1998	Little Creek	10	C	4,760	7,740	12,500	4,290	6,490	10,780	10	16
THP 1-04-053 SCR	Lower Little										
2004	Creek	102	T	588,350	82,280	670,630	495,290	57,320	552,610	16	30
1-07 NTMP-020 SCR											
2008 (NTO#1)	North Fork	143	T, C	847,480	22,620	870,100	742,990	13,950	756,940	12	38
2011 (NTO#2)	South Fork	65	H	652,110	46,510	698,620	548,419	32,093	580,512	16	38
Emergency Salvage											
1-10EM-006	Lockheed										
2010	Fire Area	148	H	838,440	20	838,460	700,010	20	700,030	17	0
			Totals	4,411,550	1,151,750	5,563,300	3,754,559	852,133	4,606,692		

Logging system: T = Tractor; C = Cable; H = Helicopter

1989 forest inventory

The forest resources of Cal Poly's Swanton Pacific Ranch were inventoried for the first time in 1989 to support development of a Forest Management Plan (Big Creek and Cal Poly 1991, Piper and others 1989). A 500 foot grid system was established in relation to a baseline and benchmark on Swanton Road at the Little Creek Bridge. Fifty-two circular 1/5th acre plots (radius 52.7 feet) were installed using a systematic approach at the intersecting points of the grid (Piper 1989, Todd 1988). Thirty-one of these plots in the upper Little Creek drainage affected both by timber harvesting and the Lockheed Fire were used for this study. This initial 1989 forest inventory was never intended to be a permanent CFI system even though it was set up on the 500 foot grid system.

1997, 2008, 2010 continuous forest inventories

A CFI system with permanently located plots on a 500 foot grid system was developed under the direction of Cal Poly faculty and staff in 1997. One-fifth acre circular plots have been established. Witness trees were selected and where possible GPS coordinates were established to assure proper follow-up measurement of all CFI plots. A report describing the specific protocol was developed by Sarah Cross (1997). The CFI system was installed in 1997 after the first timber harvest project in Little Creek. The results of the 1997 CFI plot measurements are discussed in a separate report (Bonner 1998). The CFI plots were re-measured in 2008 just after the second timber harvest project for the North Fork area had occurred. The CFI plots were subsequently evaluated 2 years later in 2010 after the 2009 Lockheed Fire to determine the extent of mortality that occurred. The 2008 CFI measurements are discussed in the 2008 Swanton Pacific Ranch Non-Industrial Timber Management Plan (Big Creek and Cal Poly 2008). The South Fork area was harvested for a second time in 2011.

Data compilation, data auditing, and data analysis

All inventory entries were converted into one spreadsheet template with consistent species designations, fields, notation and formats. Data were sorted by each field and verified with the original field notes where possible (Ali and Cody

2011). All edits and assumptions were annotated and are nested within the master spreadsheet. Tree, stand, and site data went through an integrated auditing process as it was imported into Access and uploaded to FORSEE.

FORSEE processing

The inventory data were compiled and analyzed using the FORest and Stand Evaluation Environment (FORSEE) computer program (CAGYM 2011). The CRYPTOS growth model option within FORSEE was used to model forest growth, as this model is appropriate for the coast redwood forest type (Wensel et al. 1987). All suitable site trees within the project area (dominant/co-dominant trees with no observable abnormalities and greater than 30 percent crown ratio) for both the 1997 and 2008 measurements were utilized to establish the initial site index value for the delineated project area (Cross 1997). The FORSEE calculated coast redwood site index (base age 50) came out to be 100 for this study (Krumland and Eng 2005) with a standard deviation of 10 which is considered to be an acceptable estimate. Estimates of site index (base age 100) based on Lindquist and Palley (1961, 1963) have ranged as low as 80 to a high of 200 with an average around 130 (Big Creek and Cal Poly 2008). Both of these average site index estimates fall into a Redwood Site Class III and are consistent with documented values (Big Creek and Cal Poly 2008).

The 1989 initial inventory, 1997, 2008, and 2009 post fire CFI inventories were processed and analyzed to evaluate stand attributes for: 1) all vegetation types (ALLTREES) over the entire 361 acres; 2) redwood vegetation types on site II (RW2) and site III (RW3) on 236.4 acres; and 3) Douglas-fir and hardwood vegetation types (DFHW). The Douglas-fir/hardwood timber type did not receive full evaluation due to high variability, significantly diminished sample size, and tendency for most forest management activities to occur in more productive vegetation types.

Results

The summarized information for the 1989 inventory and 1997, 2008, and 2010 CFI inventories are shown in *table 2*, and *figs. 2* and *3*. For the ALLTREES scenario, there was an overall reduction in quadratic mean diameter (QMD) from 16.09 inches in 1989 to 14.87 inches in 2008 with a subsequent increase following the 2009 Lockheed Fire to 18.91 inches. Trees per acre (TPA) increased from 154.84 in 1989 to 246.52 in 2008. Following the Lockheed Fire, TPA decreased by approximately 45 percent down to 136.83 with significant mortality occurring in trees 12 inches in diameter and smaller. Volume and basal area averages increased from 1989 to 2008. The percent standard errors for these per acre estimates are also shown in *table 2*.

Table 2—*Swanton Pacific Ranch FORSEE current status by inventory period for all vegetation types combined (361 acres).*

Inventory	Acres	Species Group	Plot Number	TPA	BA	QMD	BF/AC	CF/AC	% SE TPA	% SE BA	% SE BF/AC	% SE CF/AC
1989	361	Conifers	31	73.87	161.68	20.03	31,295	5,262	11	12	16	14
1989	361	Hardwoods	31	80.97	56.03	11.26	7,404	1,597	16	17	44	31
1989	361	Totals	31	154.84	217.72	16.06	38,699	6,859	9	9	14	12
1997	361	Conifers	33	122.12	191.63	16.96	42,930	6,927	9	8	12	11
1997	361	Hardwoods	33	106.82	67.90	10.80	10,387	2,268	22	16	26	21
1997	361	Totals	33	228.94	259.53	14.42	53,316	9,195	11	6	10	8
2008	361	Conifers	33	139.85	231.15	17.41	54,907	8,600	9	7	11	10
2008	361	Hardwoods	33	106.67	66.22	10.67	8,777	2,020	20	16	22	19
2008	361	Totals	33	246.52	297.37	14.87	63,683	10,620	10	6	9	7
2010	361	Conifers	30	81.50	209.60	21.71	51,789	8,044	12	9	13	11
2010	361	Hardwoods	30	55.33	57.32	13.78	8,545	1,930	16	16	22	20
2010	361	Totals	30	136.83	266.92	18.91	60,333	9,974	9	7	10	9

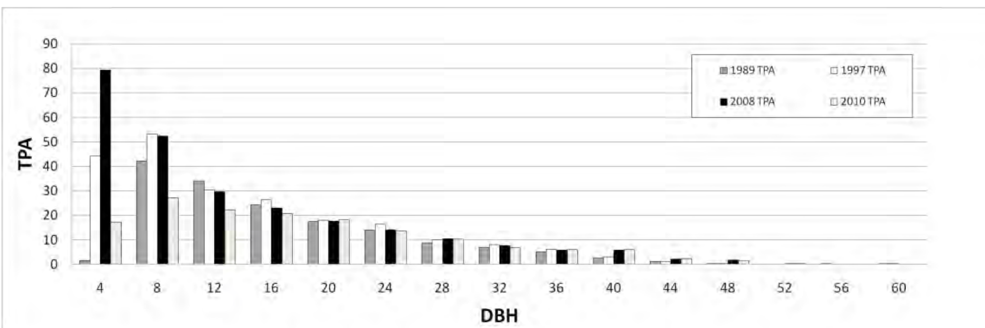

Figure 2—Stand structure for combined redwood site II and III vegetation types (includes hardwoods and conifers on 236.36 acres).

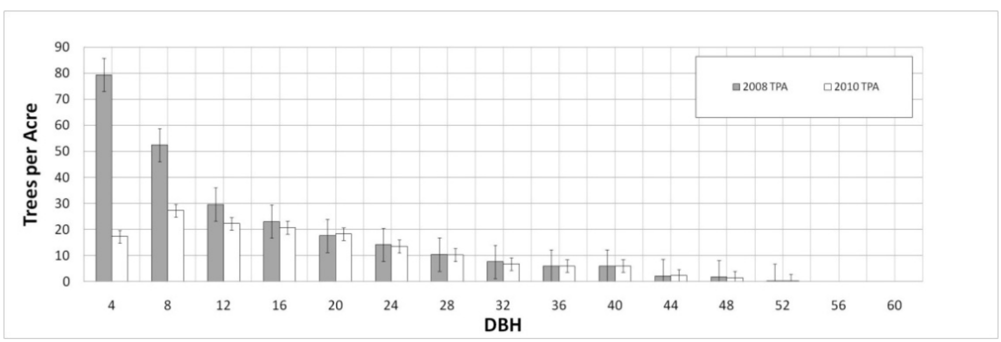

Figure 3—Stand structure for combined redwood site II and III vegetation types (236.36 acres). One standard error is illustrated.

The FORSEE current status analysis for the combined RW2 and RW3 (236.4 acres within the 361 acre project area) areas indicate a similar reduction in QMD from 17.35 to 15.28 inches from 1989 to 2008, and a subsequent increase to 19.05 inches following the Lockheed Fire (*table 3*). Trees per acre (TPA) for the combined

Table 3—*Swanton Pacific Ranch FORSEE current status for redwood vegetation types.*

Inventory	Acres	Species Group	Plot Number	TPA	BA	QMD	BF/AC	CF/AC	% SE TPA	% SE BA	% SE BF/AC	% SE CF/AC
1989	236.36	Conifers	20	87.50	213.05	21.13	45,301	7,467	11	9	13	12
1989	236.36	Hardwoods	20	71.25	47.71	11.08	4,158	1,105	19	23	32	31
1989	236.36	Totals	20	158.75	260.76	17.35	49,459	8,571	10	8	11	10
1997	236.36	Conifers	23	128.26	222.25	17.82	52,091	8,379	10	6	11	10
1997	236.36	Hardwoods	23	88.70	52.00	10.37	6,137	1,502	30	22	27	26
1997	236.36	Totals	23	216.96	274.25	15.22	58,228	9,880	13	5	9	8
2008	236.36	Conifers	23	154.57	263.02	17.66	65,587	10,215	9	6	9	8
2008	236.36	Hardwoods	23	95.43	55.28	10.31	6,945	1,641	26	23	30	28
2008	236.36	Totals	23	250.00	318.30	15.28	72,532	11,856	11	5	8	7
2010	236.36	Conifers	20	105.50	253.27	20.98	64,446	10,019	11	6	10	9
2010	236.36	Hardwoods	20	46.75	48.19	13.75	6,775	1,573	21	24	28	27
2010	236.36	Totals	20	152.25	301.46	19.05	71,221	11,592	9	5	8	7

RW2 and RW3 vegetation type showed a similar increase as the ALLTREES scenario going from 158.75 in 1989 to 250 trees in 2008 with a subsequent decrease in 2010 following the Lockheed Fire to 152.25. Increases in basal area and volume also occurred similar to the ALLTREES scenario from 1989 to 2008 with some reduction occurring after the Lockheed Fire. The percent standard errors for these per acre estimates are also shown in *table 3*.

The 1989 even-age stand structure prior to initiation of Cal Poly forest management is shown in *figs. 2* and *3*. Significant development of understory trees in the 2, 4, 6, 8, and 10 inch diameter classes leading to a reverse J shaped curve representation occurred after implementation of uneven-management which began in 1991. A subsequent reduction in the smaller diameter classes occurred after the Lockheed Fire *(tables 2* and *3*; *figs. 2* and *3*).

Discussion

A number of good lessons were learned as a result of this 25 year effort to initiate uneven-aged forest management at Cal Poly's Swanton Pacific Ranch in the truest sense of our motto of learn-by-doing. First, we did not have 1989 inventory and 1997, 2008, and 2010 CFI in one database. The four Swanton Pacific inventories were completed by different individuals and analyzed per procedures established for that inventory with a subsequent written or electronic report for each. Using FORSEE to evaluate and to predict how our stands were likely to change over time required that we: 1) put all data into one database in a consistent manner; 2) fully consider the appropriate volume equations for standing trees; 3) establish and adhere strictly to plot measurement and site tree selection protocols; and 4) delineate a project area in relation to the objectives of the study. All previously collected inventory information is now in one database for future studies.

Second, we have been collecting separate stand exam information to support individual stand prescriptions and associated marking rules. Whereas this stand specific data was useful for the prescriptions, it could not be used for monitoring purposes given that the plots were not permanently located. Thus, we decided to utilize 1997, 2008, and 2010 CFI inventory data to evaluate if reliable trends in stand structural changes could be detected from timber harvest and disturbance events such as the Lockheed Fire. We have shown here that our CFI system, when delineated to a defined project area, does provide reliable trend information. Both the stratified, combined (RW2 and RW3) and no vegetation type classifications (ALLTREES) show similar abilities to monitor changes in stand structure when there is a large sample of plots, though this threshold is far from being determined. Smaller stratification layers, such as the DFHW type, tend to exhibit larger variance due to smaller sample sizes. In essence, vegetation stratification needs to ensure statistically viable sample sizes in order to meet variance standards commonly used by the professional forestry community, but this requires a much broader study.

Third, we have defined in this study the significant changes in stand structure that resulted with implementation of uneven-aged forest management. Stand structure (refer *to tables 2* and *3*; *figs. 2* and *3*) in 2008 was approaching a regulated state in relation to the BDq targets discussed in Piirto et al. (1996b). The Lockheed Fire caused a substantial reduction in the recruited trees less than 12 inches. The extent of that mortality is illustrated in *figures 2* and *3*. It is clear that a void in the understory diameter classes less than 12 inches has resulted from the Lockheed Fire which will affect forest regulation. These results point out the difficulties and challenges foresters face as they implement, monitor, and adjust their uneven-aged forest management procedures.

Fourth, FORSEE requires the user to thoroughly and systematically audit potential errors and inconsistencies. Once data are prepared and successfully imported, the user can generate custom inventory reports and forecast growth and yield under different harvest scenarios. There is concern among some Swanton Pacific Ranch field foresters that results generated from FORSEE are overestimating current volume per acre from what is actually observed (Big Creek and Cal Poly 2008). This could be related to CFI sample size, plot location, the need for local calibration of volume equations or other factors. FORSEE must be validated for southern coast redwood stands. Additional fall-and-buck studies would validate or provide local coefficients for volume equations. CFI data could be used to validate growth model projections once more repeat measurements are collected. Further research is required.

Conclusions and management implications

Success of uneven-aged forest management in coast redwood stands is dependent on a number of factors in addition to land owner objectives and forest practice regulations. Uneven-aged forest management is as much an art as it is a science in determining the correct residual basal area (B), maximum retained diameter class (D), negative exponential constant between diameter classes (q), and cutting cycle length. Implementing stand management prescriptions via marking rules is complicated by coast redwood’s clumpy spatial distribution which results from coppice sprouting. In that context, we offer the following suggestions based on our 25 years of working and monitoring selection forestry.

Frequency and/or intensity of harvesting can profoundly affect success of coppice development (O’Hara and Gersonde 2004). Maintaining a continuous forest cover in terms of residual basal area stocking therefore must be tempered with site conditions, growing space considerations, harvesting impacts, risk of catastrophic fire, and landowner needs for recurring income. The highly varied nature of Santa Cruz Mountain redwood stands suggests a silvicultural prescription and associated marking rules that adaptively responds to these conditions. Knowing that deficits exist in lower DBH size classes should be considered in relation to leaving a few larger trees to cover that deficit on subsequent entries (modified BDq approach). Long-term success of uneven-aged forest management in coast redwood stands requires development of a documented prescription that describes 1) implementation of individual tree and/or group selection regeneration methods; 2) tree marking rules; 3) follow-up reforestation; 4) timber stand improvement treatments (for example, fuels treatment, site preparation, tanoak control); and 5) resource concerns.

There are a large number of complexities associated with implementing uneven-aged management in coast redwood stands. Much remains to be determined as to the "correct" approach for Santa Cruz County coast redwood stands. We suggest more research to: 1) regionalize FORSEE particularly with regard to southern coast redwood forests; 2) include biomass and carbon quantification as a component of FORSEE; 3) examine how uneven-aged managed coast redwood stands respond after a damaging fire to filling deficit diameter classes; and 4) examine how multi-aged coast redwood stands respond to various stand manipulation approaches.

Acknowledgments

We thank here 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 Swanton Pacific Ranch.

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Using Wood Quality Measures to Evaluate Second-Growth Redwood

Stephen L. Quarles[1] and Yana Valachovic[2]

Abstract

Redwood (*Sequoia sempervirens*) a valued species for use in appearance grade applications, such as decking, exterior siding and interior paneling, because of its dimensional stability. It is also valued for certain exterior-use applications because of its natural decay resistance. Studies have found that young-growth redwood is less resistant to decay fungi, and also exhibits greater shrinkage and swelling than old-growth redwood. Differences in natural decay resistance and dimensional stability between old-growth and young-growth could be accounted for by changes in the extractives found in the heartwood and changes in themicrofibril angle(MFA) in the S2 layer of the cell wall, respectively. Can silvicultural practices that influence growth rate be used to enhance the natural decay resistance and dimensionally stability of second-growth redwood? The objectives of this preliminary study were to 1) review anatomical characteristics of redwood that influence in-service performance and 2) present results of measurements made from second-growth redwood stumps that showed different growth rates. Decay testing data is not available.

Key words: dimensional stability, durability, microfibril angle, redwood, wood quality

Introduction

Redwood has long been a valued species because of the natural decay resistance of its heartwood and its dimensional stability. Both of these attributes contributed to redwood's long-term in-service performance, and therefore its value and reputation as a durable species. In this respect, "durability" encompasses both biological and non-biological performance characteristics – redwood that is performing poorly can be removed from service because it is rotting (a biological process) or because it is badly warped (a non-biological process). Common uses for redwood have included decking, siding and fencing (both as boards and posts). Decay resistance and dimensional stability are desirable features for products used for these applications.

The decay resistance of the sapwood (the part of the tree closest to the bark) of any species is considered nonresistant. The decay resistance of the heartwood of any species is related to the amount and type of extractable (i.e., nonstructural) materials that are deposited when sapwood undergoes the transition to heartwood. The decay resistance of heartwood (untreated) is typically classified, from least to most resistant, as "slightly or nonresistant", "moderately resistant" or "resistant or very resistant" (Wood Handbook 1999).

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The ratio of dimensional change in the tangential (the direction tangent to the growth rings) and radial (the direction extending from pith to bark) directions is sometimes considered a measure of dimensional stability. The tangential to radial (or T:R) ratio typically ranges from 1.5 to 2.5 for all wood species, with a lower number indicating greater dimensional stability. Whereas longitudinal shrinkage and swelling is normally considered negligible, it can be relatively large in products containing wood from near the center of the tree, often referred to as the pith. Dimensional change in the longitudinal (and tangential) direction is a function of the microfibril angle (MFA) in the S2 layer of the cell wall. When this angle is small, dimensional change in the longitudinal direction is also small. As this angle increases, so does longitudinal shrinkage and swelling. Meylan (1968) reported only negligible longitudinal shrinkage for Jeffery pine (*Pinus jeffreyi*) when the MFA was less than 25° but a rapid increase in longitudinal shrinkage when MFAs were greater than 25 to 30°. This change results in a corresponding reduction in dimensional change in the tangential direction. Because the S2 layer constitutes the largest portion of an individual wood fiber, it dominates the respective physical properties (e.g., dimensional change and some mechanical properties).

Differences in decay resistance and dimensional stability between “old growth” and “young growth” redwood have been summarized in the *Wood Handbook* (Forest Products Laboratory 1999). In the *Wood Handbook*, the natural decay resistance of old growth redwood is listed as “resistant or very resistant” and that of young growth as “moderately resistant”. This difference in decay resistance is based on published research by Clark and Scheffer (1983). The decay resistance of redwood has been attributed to its water soluble extractives (Anderson 1961) and, more recently, its non-water soluble extractives (Wilcox and Piirto 1976). Regardless of the particular type of extractives that impart decay resistance to redwood, it is clear that the resistance of young growth redwood is lower and more variable than that of old growth. This is particularly true in the inner heartwood area (Clark and Scheffer 1983).

The *Wood Handbook* reported T:R ratios for old growth and young growth redwood were 1.7 and 2.2 respectively (Forest Products Laboratory 1999). This increase in T:R ratio increases the likelihood of “cup” in a deck board. Another component of dimensional stability is movement in the longitudinal direction (i.e., in the length direction of a board or post).If longitudinal shrinkage was uniform throughout a given member (e.g., a board, post or plank), the expected change in dimension could be accounted for during the manufacturing process (i.e., the member could be cut longer). However, as discussed by Nault (1989), longitudinal shrinkage provides an indication of warping through *differential* (non-uniform) shrinkage. The non-uniform shrinkage in a member (i.e., having higher and lower levels of longitudinal shrinkage) would result in twist or crook in service.

The wood formed in the central zone of the tree near the pith has been referred to as “juvenile wood”, “core wood”, and “crown” or “crown formed” wood (Amarasekara and Denne 2002) and has important influences on dimensional stability of lumber. Wood further from the pith has been referred to as “mature wood” and “outer wood”. Characteristics of wood vary with distance away from the pith, and become more stable with distance (Amarasekara and Denne2002). The transition to dimensional stability is not abrupt, which makes it difficult to discuss the “juvenile wood zone” (the term used in this publication) in absolute terms, however, juvenile wood is negatively influenced by larger MFAs. Much of the research on physical properties of juvenile wood has been conducted on southern yellow pine (*Pinus palustris*) grown in the southeastern U.S. and radiata pine (*Pinus radiata*, various clones) grown in New Zealand and Australia. The lack of an abrupt transition means that the number of growth rings that make up the juvenile zone is variable. The juvenile zone is often considered to encompass the first 15 to 20 growth rings and anatomical characteristics and physical properties that would result in more durable in-service performance are lower in the juvenile region (Bendsten 1978).

Juvenile wood in trees is inevitable, but when silvicultural practices are followed that result in an increased growth rate (and therefore increased volume) during this period, the relative proportion of the tree's cross-section that is comprised of "juvenile wood" is much larger than when silvicultural practices result in slower growth. When processing siding, deck boards, or posts from the tree, the chance of a given piece having some juvenile wood is greater in the "managed for volume" tree.

Can silvicultural practices that influence growth rate be modified with redwood in such a way to maximize quality attributes? The purpose of this preliminary study was to begin exploring whether there was an effect of different growth rate in redwood based in quality measures that have traditionally been relied for long-term in-service performance. This portion of the study was limited to evaluating the effect of growth rate on the MFA in the S2 layer (and inferring the resulting impact on longitudinal shrinkage). The decay resistance of side-matched samples is being evaluated at another institution and is not currently available.

Methods

Experimental study site and management history

The forest stand where the tree cross sections were taken is located near Arcata, California. The forest was classified as a Site II – III redwood. The stand was originally harvested in the 1890s and again partially harvested in the 1950s.For these entries the largest and best formed trees were harvested. Since in the 1980's, three commercial thinnings have occurred. Today redwood is the most common tree species with lesser components of Douglas-fir (*Pseudotsuga menziesii*) and grand fir (*Abies grandis*). The trees used in this study were harvested from the stand in 2010.While it is known that wood quality is responsive to both silvicultural and genetic manipulation (Zobel 1984), no information was collected about the genetics of this stand.

Sample collection – stump to cross-section

In January 2011, four recently harvested stumps from different sprout clumps were selected for study. Five centimeter thick cross section samples were taken from the entire top of each stump at stump heights ranging from 20 cm to 68 cm above the ground (*table 1*). The sample trees were characterized as either "slow" or "fast" growing as defined by the amount of growth between rings.

Table 1—*Growth information on trees selected for use in this study.*

Tree number	Number Of rings detected	Growth rate classification	Stump height Mm (in)	Diameter mm (in)	Average growth (rings/mm)
2	72	Fast	685 (27)	775 (30.5)	10.7
6	74	Fast	225 (9)	940 (37)	12.7
3	88	Slow	335 (14)	520 (20.5)	5.9
4	100	Slow	200 (8)	445 (15.5	4.5

Sample processing – creating radial rectangular specimen

The sections collected were delivered to the University of California Richmond Field Station for processing the samples into square rods approximately 18mm by 18 mm by the radial (pith to bark) length of the sample. Two sets of side matched square rods were cut from each tree sample. One square rod specimen was used for the MFA analysis and one was reserved for use in a decay resistance test. After processing, all square rods were placed in a controlled environment room to dry. After equilibrating, one set of the square rods was shipped to FP Innovations, Paprican Division (Vancouver, British Columbia, Canada) for further processing and MFA analysis. The second set was shipped to Mississippi State University for additional analysis and testing to determine decay resistance. The specimens were further processed at Paprican, cutting the 18 x 18 mm square rods into rectangular sections that were 7 mm in the longitudinal direction, 2 mm in the tangential direction, and the length from pith to bark in the radial direction. The MFA was then evaluated using a Silviscan instrument that utilizes x-ray densitometry, x-ray diffraction and image analysis to measure a range of wood properties (age, density, and the modulus of elasticity (MOE) that indicates stiffness).

Results and discussion

The relationship between growth ring and MFA is shown in *fig.1*. Elevated MFA was most evident in tree #5. In this case the MFA did not decrease with tree age as was the case with the other trees. Local increases in MFA were observed in trees # 6 (around year 40), #2 (around year 30) and #3 (around years 50 and 80). Based on results from studies on southern yellow pine variation in MFA can be expected, but according to Megraw (1985), a negative relationship exists between MFA and tree age during the first 15 to 20 years, and at which time MFA remains relatively constant with increasing tree age. These localized increases in MFA could result from changes in growth rate during those periods. Trends in MFA that do not agree with published trends, such as those observed here, are possible with small sample sizes[3]. As previously stated, the samples for this study were taken from the tree stumps and this sampling location may drive some of the observed relationships reported here, with higher MFA typically being observed at lower heights above the ground (Megraw 1985). Future research should incorporate a much larger sample size and include multiple samples from a given tree.

The relationship between MFA and modulus of elasticity (MOE and a measure of stiffness) and MFA and density are given in *figs. 2* and *3*. In the Silviscan analysis, MOE is generated using a correlation developed with several species between MFA

[3] L. Schimleck, Professor, University of Georgia, personal communication, 2011.

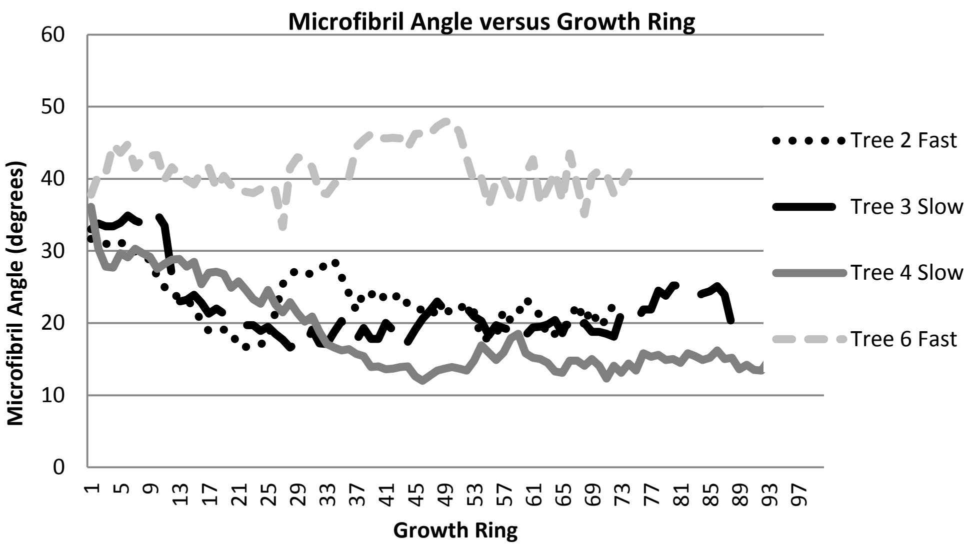

Figure 1—Microfibril angle as a function of growth ring in the four sampled trees characterized by their rate of growth.

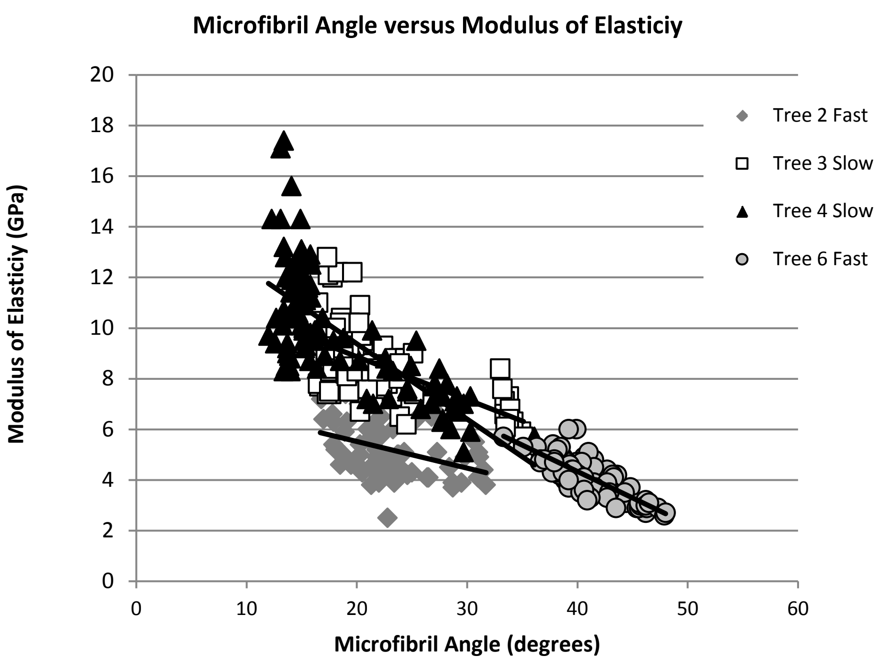

Figure 2—The relationship between microfibril angle and modulus of elasticity in the four sampled trees characterized by their rate of growth.

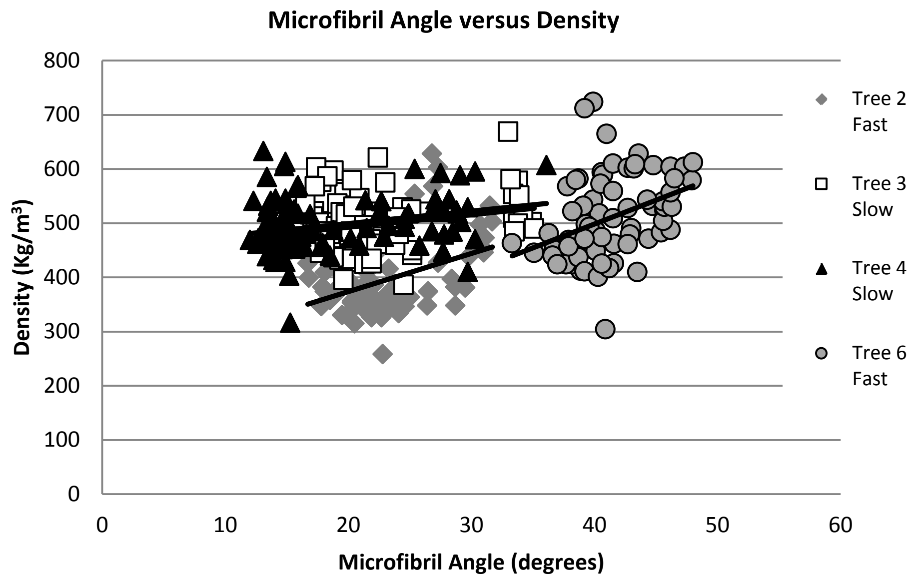

Figure 3—The relationship between microfibril angle and density in the four sampled trees characterized by their rate of growth.

and MOE. As such, it is unlikely that redwood samples were included in the development of the correlation. However, increasing MFA would result in between cellulosic unit hydrogen bonding to predominate during the tension test used to determine MOE. With lower MFA, the covalent ß (1-4) linked D glucose units that comprise the cellulose molecule would dominate. It is therefore reasonable to expect increasing MFA to result in lower MOE (in tension parallel to the grain). As seen in *fig. 2*, higher MFA resulted in lower MOE. Again, this was most clear with tree #6. The MFA in tree #2 was not consistently lower than the MFA in trees #3 and #4, but in tree #2 MOE is consistently lower than the MOE in trees #3 and #4. Based on the results of the SilviScan analysis, the average density of tree #2 was approximately 25 percent lower than trees #3 and #4. Since strength is highly correlated with density (Bowyer et al. 2003), this difference in density can help explain the differences in MOE observed in trees #3 and #4 ("slow growth rate") and #2 ("fast growth rate"), even though the MFAs were similar. This relationship is confirmed when viewing the relationship between MFA and density (*fig. 3*).Density (specific gravity) is one of the physical properties that will increase with age during the juvenile period (Bendsten 1978). The density values for tree #6 ranged from near the minimum to maximum values observed for trees # 2, 3 and 4. Because of the consistently high MFA in tree # 6, the predicted MOE is consistently lower than that for trees # 3 and 4.The overall lower density in tree #2 samples, and that corresponding effect on MOE, explains the relatively low MOE even though tree #2 MFA values are similar to the MFA values for trees #3 and #4.

Summary and conclusions

Higher MFA was observed in the faster growth tree #6. The trend observed in tree #2 was similar to those observed in trees #3 and #4. Particularly with tree # 6, MFA did not decrease with age as would be expected. Inconsistent results can be expected when measurements are made on an individual tree at a single stem location. A future study would be greatly enhanced by considering genetics and a range of sample locations within the tree bole. However, except for places where redwood is grown in plantations, there will always be genetic diversity in managed redwood forests and silviculture techniques may be used to compensate for some genetic characteristics.

Decay resistance and extractive content are very more important factors regarding the effect of second-growth issues on the in-service durability of redwood. When these data are available it will help round out the results reported here.

With this preliminary study, we have shown that when considering all measurements available with Silviscan, differences in faster growth and slower growth redwood could be discerned that may affect in-service performance of redwood lumber. These redwood samples followed the general trends observed in other conifer species where wood quality measurements have been investigated.

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Forest Restoration at Redwood National Park: a Case Study of an Emerging Program

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Abstract

For more than 30 years, Redwood National Park has been working to establish a Forest Restoration Program to rehabilitate its impaired, second-growth forests. This case study outlines the Park's history of using silviculture as a restoration tool, which began in 1978 after the Park's expansion. The most recent effort was the 1,700 acre South Fork of Lost Man Creek Forest Restoration Project where two silvicultural prescriptions were used. Low thinning on ridge-top sites reduced basal area by 40-percent, and wood generated was sold as forest products. Crown thinning on steep mid-slope sites reduced basal area by 25-percent, and the wood was lopped-and-scattered. Permanent plots were established before thinning and were re-assessed after thinning. Data were analyzed to determine whether the silvicultural prescriptions altered stand structure and species composition to promote redwood dominance. Before thinning, Douglas-fir dominated stand density. Both prescriptions shifted composition in favor of redwood. The ridge-top prescription resulted in 191.3 percent more redwood trees/acre than Douglas-fir, and 80.2 percent more redwood basal area, making redwood the dominant species. The mid-slope prescription resulted in 4.3 percent more redwood trees/acre than Douglas-fir, but Douglas-fir had 5.9 percent more basal area. This project is the Park's first successful attempt at large-scale forest restoration.

Key words: crown thinning, low thinning, monitoring restoration, redwood, silviculture

Introduction

Today over 50,000 acres of second-growth forests occur within Redwood National Park. These forests have developed into dense stands characterized by deficient redwood (*Sequoia sempervirens* [Lamb. ex D. Don] Endl.) composition, low tree vigor, homogeneous structure, and little bio-diversity (Chittick and Keyes 2007, Teraoka and Keyes 2011). This case study outlines the Park's 30 year history of forest restoration planning and how the history culminated in the South Fork of Lost Man Creek Forest Restoration Project.

Redwood National Park was created in 1968 to preserve intact, old-growth redwood ecosystems. Under the Redwood National Park Expansion Act of 1978, 48,000 acres were added to protect the Park from encroaching logging activities. The expansion included 38,000 acres that had been logged, so the new law mandated a program of rehabilitation to restore these lands. This marked the beginning of the Park's forest restoration planning efforts.

In 1978, the Park established a 200 acre experimental thinning referred to as the Holter Ridge Thinning Study. The area had been clearcut in 1954, was dominated by

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Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), and averaged 1,000 trees/acre 24 years after clearcutting (Chittick and Keyes 2007). A low thinning had been conducted to determine its effects on tree growth, forest composition, and understory vegetation (Chittick and Keyes 2007). This marked the first time that trees were cut down at Redwood National Park as a restoration effort. National Park Service executives, however, disagreed with the notion that thinning could be used as a restoration tool; others expressed frustration that the Park was cutting down trees on land removed from timber production. Finally, a moratorium on thinning second-growth forests in the Park was established in 1979, and the Park's first attempt at forest restoration came to an end.

In the 1990s, the listing of the northern spotted owl (*Stix occidentalis caurina* [Merriam]) as a threatened species made forest management on public lands a contentious issue and led to the development of the Northwest Forest Plan. Research began on ecosystem management practices that could maintain timber production while protecting, and creating, spotted owl habitat. Redwood National Park used this shift in paradigms to revive forest restoration planning, which resulted in a 40 acre experimental thinning in 1995 referred to as the Whiskey 40 Thinning Study. The area had been clearcut in 1962 and averaged 2,200 trees/acre 33 years after clearcutting. The thinning had been conducted in part to promote redwood composition (Stuart and Cussins 1994) and to demonstrate potential management for a larger forest restoration program.

A funding opportunity to support forest restoration came about when a neighboring timber company planned to log over 500 acres of old-growth stands that were acquired as a land swap connected to the creation of Redwood National Park. Because the marbled murrelet (*Brachyramphus marmoratus* [Gmelin]) was a newly listed threatened species, the timber company proposed to mitigate their logging by providing the Park $1.7 million for thinning. The Park quickly drafted a forest restoration plan (United States 1996) and it was distributed for public review in 1996. A local environmental group responded with a legal challenge, arguing the plan lacked cumulative effects analysis on marbled murrelets from the timber company's proposed logging and failed to state where the Park planned to thin. The Park's plan could not be approved and the mitigation dollars were not accepted; thus, the Forest Restoration Program was once again put on hold.

This unsuccessful planning effort, however, increased public awareness of the Park's impaired forests and increased public support for forest restoration. In 1999, the Park's General Management Plan (United States 1999) was approved, and provided the compliance needed to revive the Forest Restoration Program. To build momentum for the program, a park-wide forest inventory was conducted to describe the range of resource conditions and to use the data to prioritize areas for treatment. In addition, the Whiskey 40 Study was revisited in 2002 and the Holter Ridge Study was revisited in 2003 to aid managers in prescribing future silvicultural practices for restoring forests. The results from both studies were similar in that they both showed that residual trees responded well, there was a significant increase in biodiversity, and the overstory composition was unaffected as Douglas-fir continued to dominate because thinning intensities were too conservative or too much redwood was removed (Chittick and Keyes 2007, Teraoka and Keyes 2011).

Based on the assessment of the Park's second-growth forests, the South Fork of Lost Man Creek drainage was chosen as the first major forest restoration project

(United States 2009). Planning began in 2004 and the plan was approved in 2009. The plan's objectives were in part to reduce stand density to promote redwood composition, initiate understory vegetation, and promote vigorous tree growth. To address logistic, economic, and regulatory constraints, five silvicultural prescriptions were used; the two most prominent prescriptions are discussed in this case study. Regardless of prescription, Douglas-fir was primarily targeted for removal. On ridge-tops, where slopes were less than 30 percent, low thinning was used to reduce the stand basal area by 40 percent, and the Park sold the merchantable timber, a byproduct of the restoration, to the contractor that conducted the restoration work. Non-merchantable wood was sent to a cogeneration power plant. On mid-slope sites, where slopes exceeded 30 percent, crown thinning reduced the stand basal area by no more than 25 percent, and felled trees were lopped-and-scattered. Crown thinning removes trees in the dominant and co-dominant crown classes to benefit trees of the same crown classes. This method was used on mid-slope sites to benefit redwood and reduce stand density while only cutting a few trees. Since logs were not extracted from mid-slope sites, the accumulation of cut wood was a major constraint.

Methods

For this analysis, a subset of preliminary data was used from the project's monitoring effort, which began in 2009. Data collected after thinning were analyzed to determine whether the two prescriptions immediately altered stand structure and species composition to promote redwood dominance.

The South Fork of Lost Man Creek Project comprises 1,700 acres. Most of the area was clearcut in 1954; the rest was clearcut in 1962. The latitude/longitude is N41° 17 W123° 58. Elevation ranges from 800 ft to 2,200 ft, and annual precipitation averages 65 inches.

Species and diameter at breast height (dbh) of live trees larger than 10.5 inches dbh were recorded in six 3/5-acre square plots. A 1/40-acre square subplot was nested at each corner of the 3/5-acre plot (four subplots per 3/5-acre plot). In the 1/40-acre subplots, the same data were recorded for trees 4.5 to 10.5 inches dbh. Plots were re-inventoried immediately after thinning. Three plots represented the thinning at the ridge-top site and averaged 15 percent slope. Three plots represented the thinning at the mid-slope site, and averaged 50 percent slope. Both sites were on northwest-facing aspects and were adjacent to each other. Means and standard deviations for stand density (trees/acre, basal area/acre) and quadratic mean diameter (QMD) were calculated for both inventory phases. Diameter distributions were plotted for both inventory phases. Data for western hemlock (*Tsuga heterophylla* [Raf.] Sarg.) and grand fir (*Abies grandis* [Dougl. ex D. Don] Lindl.) were grouped and named "Other" species.

Results

Thinning on ridge-top site

Prior to treatment, the ridge-top site's stand density consisted of 566.1 trees/acre and 366.7 ft^2/acre basal area (*table 1*). The majority of the stand consisted of three species: Douglas-fir, redwood, and tanoak (*Lithocarpus densiflorus* [Hook. & Arn.]

Rehder). Before thinning, redwood and Douglas-fir had similar average diameters, both larger than tanoak. Douglas-fir dominated stand density in terms of trees/acre and basal area. However, redwood dominated the 22 inch diameter classes and greater (*fig. 1*, *t*op). The thinning reduced stand basal area by 38.4 percent (the target reduction was 40 percent), reduced the number of trees by 46.7 percent and increased stand QMD by 6.3 percent, demonstrating the thinning was successfully implemented as designed (*table 1*). All removals were in the 4 to 20 inch diameter classes (*fig. 1*, bottom), with a QMD of 9.9 inches for all removed trees.

Table 1—*Stand conditions before and after thinning at the ridge-top site expressed in number of live trees/acre, basal area (ft^2/acre), species composition (%), and quadratic mean diameter (QMD, inches) for redwood, Douglas-fir, tanoak, and other species.*

	Trees/acre			Basal Area (ft^2/acre)			QMD (inches)
	Mean	**SD**	**%**	**Mean**	**SD**	**%**	
Before Thinning							
Redwood	167.2	55.4	29.5	132.5	18.7	36.1	13.9
Douglas-fir	221.1	80.0	39.1	185.9	31.8	50.7	14.0
Tanoak	177.2	134.8	31.3	47.3	38.2	12.9	7.6
Other	0.6	1.0	0.1	1.0	1.7	0.3	17.9
Total	566.1	69.9	-	366.7	14.3	-	11.2
After Thinning							
Redwood	150.6	50.4	49.9	125.6	16.2	55.6	14.6
Douglas-fir	51.7	24.2	17.1	69.7	21.2	30.8	18.9
Tanoak	98.9	74.4	32.8	29.6	27.4	13.1	8.0
Other	0.6	1.0	0.2	1.0	1.7	0.4	17.9
Total	301.7	46.4	-	225.9	19.3	-	11.9

The thinning increased redwood composition relative to Douglas-fir (*table 1*). Prior to thinning, there were 32.2 percent more Douglas-fir than redwood and Douglas-fir had 40.3 percent more basal area than redwood. Immediately after thinning, there were 191.3 percent more redwood than Douglas-fir and redwood had 80.2 percent more basal area than Douglas-fir.

Stand QMD increased by 6.3 percent after thinning. This small boost in stand QMD was associated with leaving redwood virtually uncut and retaining approximately half of all the tanoaks in the smallest diameter classes (*fig. 1*). Douglas-fir represented 64.1 percent of the total number of trees removed and 82.5 percent of the total basal area removed. The removals concentrated on smaller diameter Douglas-fir (*fig. 1*); thus, its increase in QMD was 35.0 percent (from 14.0 to 18.9 inches), while redwood and tanoak QMD increased by 5.0 percent and 5.3 percent, respectively.

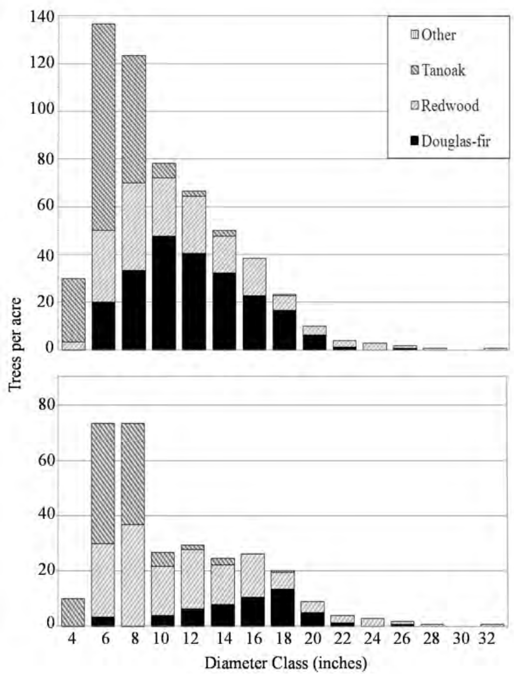

Figure 1—Diameter distributions at the ridge-top site stacked by species immediately before thinning (top) and after thinning (bottom).

Thinning on mid-slope site

Prior to treatment, the mid-slope site's stand density consisted of 391.7 trees/acre and 364.3 ft^2/acre basal area (*table 2*). The majority of the stand consisted of two species, Douglas-fir and redwood. Tanoak occurred in the stand, but was a minor component. Before thinning, redwood and Douglas-fir had similar average diameters. Although Douglas-fir and redwood had similar compositions, Douglas-fir represented most of the stand density in terms of trees/acre and basal area. However, redwood dominated the 20 inch diameter class and greater (*fig. 2*, top). The thinning reduced stand basal area by 11.0 percent (the target reduction could not exceed 25 percent), reduced the number of trees by 14.0 percent and had virtually no effect on stand QMD. The thinning was successfully implemented as designed; however, the reduction in basal area was low. All removals were in the 4 to 18-inch diameter classes (*fig. 2*, *bottom*), with a QMD of 11.5 inches for all removed trees, and concentrated on Douglas-fir. Douglas-fir represented 68.7 percent of the total trees/acre removed and 87.9 percent of the basal area removed.

The thinning had minor effects on redwood composition relative to Douglas-fir (*table 2*). The thinning virtually inverted the proportions of Douglas-fir to redwood in terms of number of trees, but failed to do so in terms of basal area, although it brought balance to the overall basal area composition. Prior to thinning, there were 11.4 percent more Douglas-fir than redwood, and Douglas-fir had 27.2 percent more basal area than redwood. Immediately after thinning, there were 4.3 percent more redwood than Douglas-fir, but Douglas-fir had 5.9 percent more basal area than redwood, which was an improvement to the pre-thinning condition.

Table 2—*Stand conditions before and after thinning at the mid-slope site expressed in number of live trees/acre, basal area (ft^2/acre), species composition (%), and quadratic mean diameter (QMD, inches) for redwood, Douglas-fir, tanoak, and other species.*

	Trees/acre			Basal Area (ft^2/acre)			QMD
	Mean	SD	%	Mean	SD	%	(inches)
Before Thinning							
Redwood	160.6	34.3	41.0	148.0	51.2	40.6	13.0
Douglas-fir	178.9	41.4	45.7	188.3	16.5	51.7	13.9
Tanoak	36.1	42.8	9.2	9.0	9.4	2.5	6.8
Other	16.1	10.0	4.1	18.9	4.2	5.2	14.7
Total	391.7	43.1	-	364.3	50.2	-	13.1
After Thinning							
Redwood	147.2	34.7	43.7	145.0	55.0	44.6	13.4
Douglas-fir	141.1	46.0	41.9	153.5	27.9	47.3	14.1
Tanoak	32.8	37.3	9.7	8.4	8.6	2.6	6.9
Other	15.6	9.2	4.6	17.8	5.5	5.5	14.5
Total	336.7	20.5	-	324.7	63.8	-	13.3

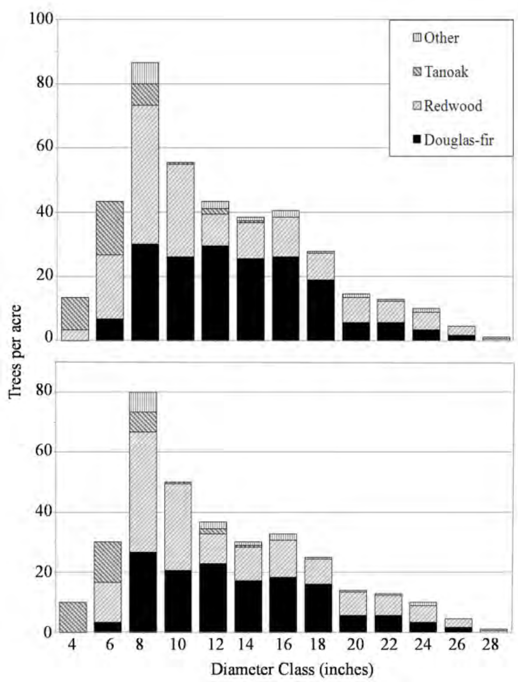

Figure 2—Diameter distributions at the mid-slope site stacked by species immediately before thinning (top) and after thinning (bottom).

Discussion

More than 30 years after the first attempt at establishing a Forest Restoration Program at Redwood National Park, regulatory agencies, the environmental community, and conservation organizations have embraced the idea that thinning young forests can improve ecosystem values. The effects of restoration thinning have been documented throughout the Pacific Northwest (Carey 2003, O'Hara and Waring 2005). Interest in the restoration of young-growth redwood forests has grown in part because the range of old-growth forest continues to decline (Evarts and Popper 2001), fragmented old-growth redwood stands have been deemed inadequate as reserves leading to increased landscape-scale efforts to connect these fragmented old-growth stands (Noss et al. 2000, Porter et al. 2007), and young forests in reserves exhibit declines in vigor and redwood dominance.

Teraoka and Keyes (2011) found that applying low thinning that targets all species for removal can result in a reduction of redwood composition relative to Douglas-fir and an intense low thinning targeting Douglas-fir could better achieve restoration objectives. This issue was addressed in the current case study. At the ridge-top site, the objectives of immediately altering stand density conditions while enhancing redwood dominance was successful because it was intense enough to immediately make redwood dominant. Although redwood dominated the largest size classes after thinning, the largest Douglas-firs were retained and Douglas-fir had the largest increase in QMD. This suggests that a gain in Douglas-fir's competitive potential in the upper canopy classes may occur during subsequent stand development, but this remains to be determined.

At the mid-slope site, the objective of immediately altering stand density conditions while enhancing redwood dominance was not fully met by low-intensity crown thinning; however, stand density conditions and redwood composition were improved. Results may have been more favorable if the basal area reduction was closer to 25 percent. The intent of this prescription was to remove fewer trees, while targeting Douglas-fir in the upper crown classes. Thus, a minor reduction in stand density could be achieved while releasing redwood in the upper crown classes. Regardless of the thinning intensity, all removals focused on redistributing growing space for redwood. Whether the thinning provided redwood with a more competitive position over the course of subsequent stand development remains to be determined.

The lop-and-scatter prescriptions were funded by the Centennial Challenge Initiative and the American Recovery and Reinvestment Act of 2009, which provided the Park $629,084 to thin 1,267 acres, a rate of $497/acre. In contrast, approximately 2.5 million board-feet were removed from 226 acres of ridge-top sites and sold to the contractor, which resulted in the contractor paying the government $221/acre to perform the restoration work. This project has demonstrated that excess wood, a byproduct generated from the restoration work, could be sold to offset the cost of implementing the project without compromising restoration objectives.

The Park is planning its next forest restoration project, which calls for managing 1,200 acres within the Lost Man Creek Drainage. In addition to traditional thinning methods, the Park proposes to implement variable density thinning, which has shown to be effective as a restoration tool in the redwood region (O'Hara et al. 2010, Keyes et al. 2010). In addition to ground-based equipment, the Park proposes to thin intensely on steeper slopes using cable yarding systems, which would allow the Park to remove excess biomass.

The challenge ahead for managers is to develop diverse silvicultural restoration treatments that are founded on the knowledge of forest dynamics in young stands and structural references from old-growth ecosystems, suit a variety of stand conditions, and are appropriate for an array of social and economic constraints.

Acknowledgments

I thank Leonel Arguello and Scott Powell for their many contributions. I thank Emily Teraoka and Christopher Keyes for their reviews of the manuscript.

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Whiskey Springs Long-Term Coast Redwood Density Management; Final Growth, Sprout, and Yield Results

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Abstract

Multi-decadal studies of commercial and precommercial thinning in redwood stands are rare and consequently of value. The Whiskey Springs study at Jackson Demonstration State Forest has a data set spanning 35 years. In addition to growth and yield response to commercial thinning, the results provide important information for evaluating regeneration and alternative silviculture treatments.

In 1970, the original treatment thinned a 40 year-old redwood forest from below to densities representing 25, 50, and 75 percent of the initial basal area (approximately 400 ft^2/ac). Three 0.2-ac plot replicates within each of the four density levels were assigned in a randomized design. In the 35 years after thinning, the basal area of the 25, 50, 75 and 100 (control) percent retention areas increased by 168, 232, 226 and 210 $ft^{2/}$ac of basal area respectively. There was some indication that mean annual increment (ft^3/ac) had culminated at age 70 years for some treatments.

Redwood sprouts and other conifer seedlings were abundant in the 25 and 50 percent retention areas. Precommercial thinning subplots were established within these two treatments in 1986, reducing the understory to 200, 300, and 400 sprouts/ac. The understory diameters ranged from 2 to 6 inches at DBH for the 25 percent retention 35 years after the commercial thinning.

Key words: *Sequoia sempervirens*, coast redwood, commercial thinning, precommercial thinning, regeneration, sprouting

Introduction

This report provides the final measurements for what began as the Whiskey Springs Redwood Commercial Thinning Study. It was established by the USDA Forest Service and the California Department of Forestry and Fire Protection staff in 1970 on Jackson Demonstration State Forest (JDSF) in 1970 near Fort Bragg, California. This study spans 35 years after thinning of a 40 year-old, well stocked second growth stand. The original experimental design included three sites located in the redwood region. Oliver et al. (1994) reported the first 15 years of growth (1971 to1985) for all three sites. James Lindquist has summarized periodic remeasurement

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data at JDSF (1988, 2004). The now 75 year-old stand has literally outgrown the study design and has transitioned into an educational demonstration site.

The study was designed to test treatments to help realize the productive potential of young redwood forests. The original study plan called for a final harvest at 70 years of age based on the projected culmination of the mean annual increment on a cubic foot volume basis for a 100-year site index of 180 ft.

Vigorous coast redwood (*Sequoia sempervirens* [D.Don] Endl.) sprouting occurred after the commercial thinning. A secondary regeneration focused study was established to examine this sprouting. The understory sprouts were precommercially thinned to a range of densities. Natural regeneration and redwood sprout growth were measured periodically beneath the thinned overstory.

The objective of the experiment was to test a range of commercial thinning intensities for the purpose of comparing total yield. The experiments' growth and yield data aids in planning silvicultural prescriptions in redwood to achieve maximum board foot volume production or other attributes such as target tree size. This data provided stand structure information to help inform decision-making regarding multiple goals in redwood forest management. The sprout response provided information for transitioning from even to multi-aged stand management.

This paper will provide a focused update on the final 35 year results and explore total yield and some density attributes. This long term experiment will be discussed in more detail in a California Forestry Report under preparation.

Methods

Site

The Whiskey Springs study was located approximately 9 miles from the Pacific coast, in redwood forest. There was an east to northeast aspect and an elevation range from 600 to 900 ft. The old-growth stand was clear-cut in the late 1920s. Natural regeneration of a nearly pure stand of redwood sprouts followed. There was evidence as well as records (Caspar Lumber Co.) that the area was planted with redwood seedlings around 1929. Breast-height age of redwood ranged from 39 to 41 years in 1970. Site index ranged from 172 to 212 ft, using 100 years as the base age (Lindquist and Palley 1963). The pretreatment basal area of trees greater than 4.5 inches in diameter at breast height (DBH) was 401.3 ft^2 per ac. The pre treatment species composition of trees by basal area was 91.8 percent redwood, 7.6 percent other conifers (Douglas–fir, *Pseudotsuga menziesii* [Mirbel]-Franco); grand fir, *Abies grandis* [Douglas] Lindley) and the remainder tanoak (*Lithocarpus densiflorus* [Hook &Arn.] Rehder).

Design

The main study established 12, 0.4-ac treatment blocks (132 x 132 ft). Measurement plots were established in a central 0.2-ac (93.3 x 93.3 ft.) section of each block. The blocks were located so that inter-plot basal area, site index, and species composition were as uniform as possible. Four thinning treatments were assigned to 12 blocks in a randomized design (Lindquist 1988). *Figure 1* shows the layout of the main study plots.

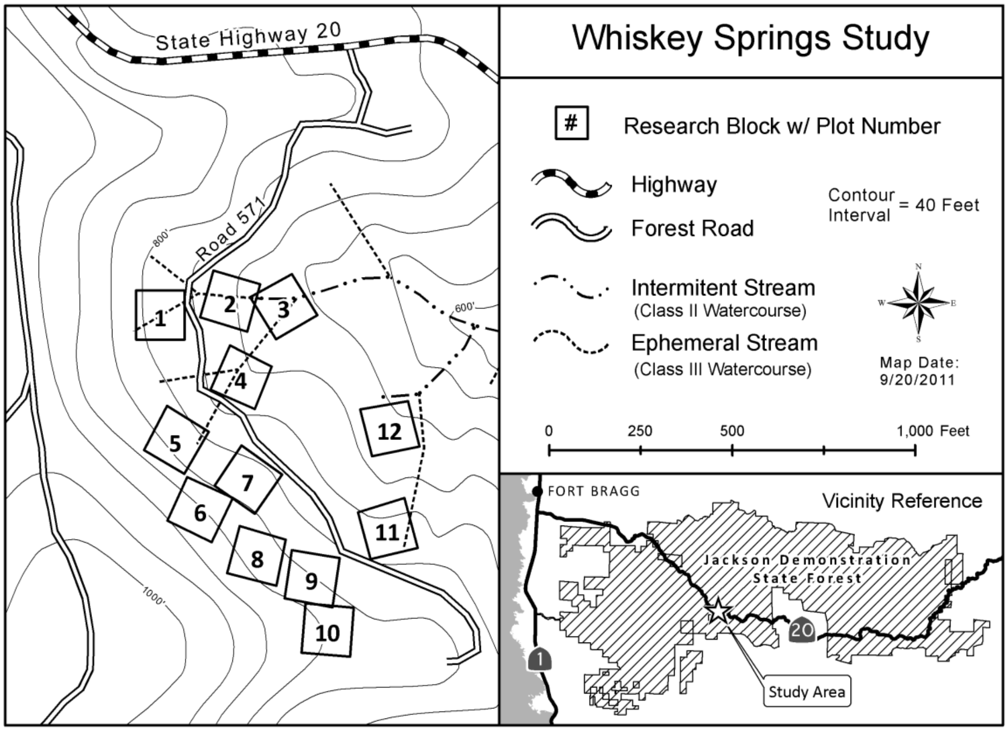

Figure 1—Plot locations at study site on topographic map and vicinity map for Whiskey Springs.

Commercial thinning in 1970 reduced basal area to approximately 100 ft^2/ac (25 percent retention), 200 ft^2/ac (50 percent retention), 300 ft^2/ac (75 percent retention), and retained an untreated control of 400 ft^2/ac.

Selection of leave trees was implemented to favor healthy dominant and codominant redwoods while leaving trees well distributed on the plot. Hardwoods and any conifers less than 4.5 inches DBH were removed. All trees were permanently tagged. After treatment in 1970 a total of seven plot measurements were conducted at approximately 5 year intervals ending in 2005.

Monitoring the effects of redwood sprouting under the four different treatment levels began in 1972. The secondary precommercial thinning study was implemented in 1986 in the 25 percent and 50 percent retention plots. The secondary study design divided treatment blocks into 1/10 ac quadrants. Each quadrant was randomly assigned to an understory regeneration stocking level of 100, 200, or 300 trees per ac and an untreated control. Leave trees for the precommercial thin were based on diameter and height, but desirable spatial distribution was also a goal (Lindquist 1988). All of the leave trees were tagged then diameters and heights were measured. Subsequent periodic measurements of the thinned understory occurred in 1991, 1999, and 2007.

Data analysis

Data was summarized for each of the seven measurement dates. It was evaluated by the statistical functions in Microsoft Excel 2007. Two way ANOVA tests were conducted for treatment averages though the 35 year study period. For some attributes, one way ANOVA with replicates were conducted for the 2005 data.

New redwood site index equations have been developed with a base age closer to the stand age (50 vs.100 year) since the study was established. Krumland and Eng (2005) site was calculated using individual tree site values though the 35 year span for base age 50. For each plot the five best sites were averaged.

Because the stand structure became more complex as a result of the sprouting, SDI was calculated using the summation method (Tappeiner et al. 2007): SDI= $\Sigma(D_i /10)^{1.0605}$. Understory diameters were adjusted to the measure date for overstory (2005).

At the beginning of the study, a limited number of trees were measured for heights. Lindquist (2004) described the method used to estimate heights for all trees. Over the 35 year study period, methods for computing volume have advanced. For this final measurement a new computation method was used. To establish height for all other trees, a nonlinear regression model (R Development Core Team, version 2.2.0, 2006) was used to develop equations based on all the measured trees (by species) for each measure year. Equations used to calculate cubic and board foot tree volumes were developed by the Redwood Yield Cooperative (Wensel et al. 1987) and locally calibrated with the JDSF fall-and-buck study (Griffen 1986). Board foot volumes were computed by the Scribner log rule using a one-foot stump height and a top diameter of 6 inches.

Because prior reports indicated between treatments cubic foot volume growth increments were similar (Lindquist 2004), and date of measurement varied during the study period; seasonal growth adjustments were made to diameters. Data from JDSF (Bawcom et al. 1961) was used to adjust diameters to a common measure date.

Results

Site quality

The treatment-wide averages for site index (Krumland and Eng 2005) were 140 ft for 25 percent retention, 133 ft for 50 percent retention, 139 ft for 75 percent retention, and 124 ft for the untreated control using 50 year base age. Small watercourses are found in about half the treatment blocks, but none in the control blocks. The average site index for plots by treatments indicated the site quality of the treatment plots were not significantly different in terms of site index (P = 0.370).

Mortality

During the study period, the untreated plots exhibited the highest mortality of 50 to 75 year old trees. The number of trees per ac (greater than 4.5 inches DBH) declined steadily after the first decade to 133 trees per ac or 21 percent less than the initial measurement. The 75 percent retention experienced limited mortality starting in 1985, totaling six trees per ac or about three percent of the initial measurement. No mortality was observed for overstory trees in the 25 and 50 percent retention plots.

Diameter

Heavier thinning from below (removing the smallest stems) resulted in larger average stem diameters as shown in *fig. 2*. Error bars of one standard deviation are graphed illustrating that 50 percent retention had the most variation between plots. Quadratic mean diameter was significantly different between treatments through the 35 year study period (*P* <.01).

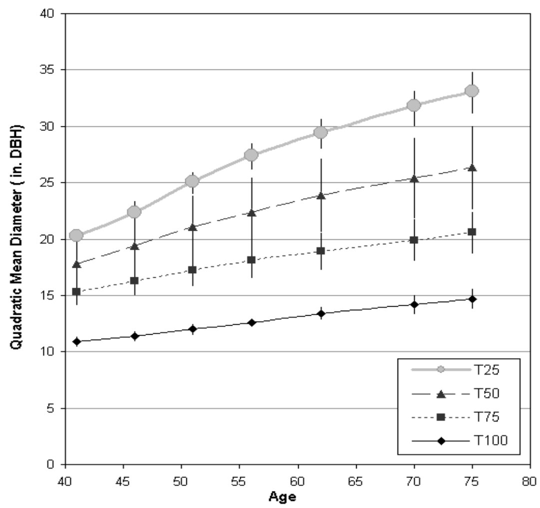

Figure 2—Mean tree diameter by treatment through time. Error bars represent one standard deviation.

Basal area

The initial treatment basal area of trees greater than 4.5 inches in diameter are depicted in *fig. 3*. The treatment's effect on total basal area per ac remained significant throughout the 35 year study period *(P* <.01).

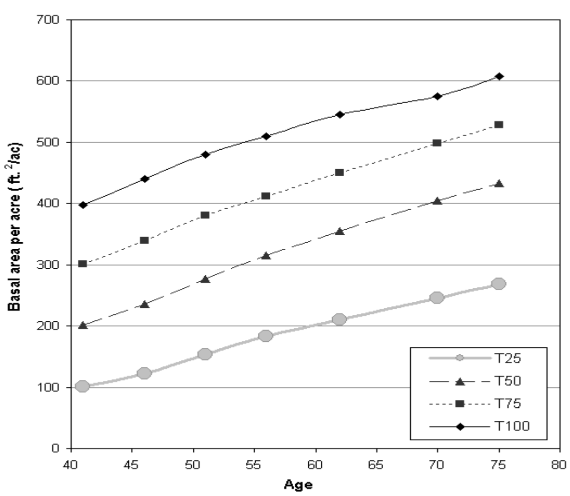

Figure 3—Basal area in square feet per ac by initial thinning density though the 35 year study period.

Volume

The untreated controls continued to have the largest cubic foot volume through the study period. For cubic foot volume there were significant differences between treatments though the 35 year study period (P <.01).

Mean annual increment

A biological characteristic that is of management interest is the change in growth rate by treatment through time. Mean annual increment (MAI) is the average annual growth over the life of the stand. For cubic foot volume per ac MAI; there were significant differences by treatment (P <.01), but not through the 35 year study period (P = 0.0228).

One purpose of the initial study was to determine when mean annual cubic foot growth increment peaks. For all treatments except the control, the MAI for cubic foot growth per ac reached its highest value at stand age 70 (*fig. 4*). Vertical error bars of one standard error help illustrate variably between plots for a single treatment. The decline in MAI was less than one percent per year, so at this point the projected culmination of mean annual increment at age 70 was suggested but not definitive.

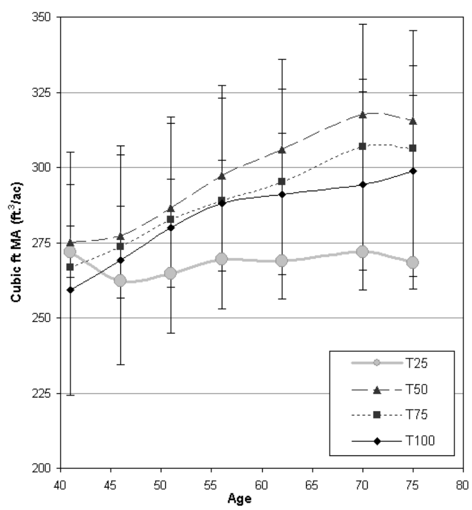

Figure 4—Mean Annual Increment of cubic foot volume per year by thinning retention treatment. Vertical error bars indicate variability of data.

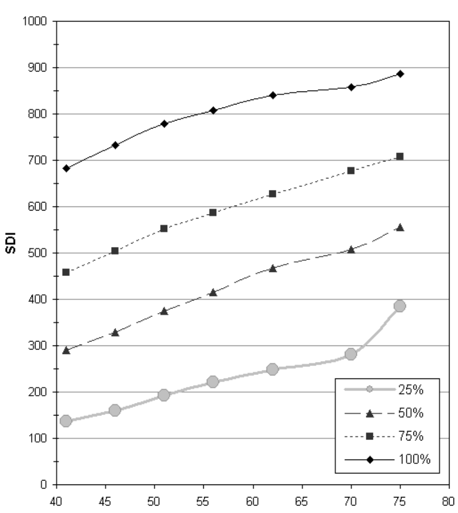

Figure 5—Stand Density Index (SDI) by treatment through time. The 25 percent and 50 percent retention at age 75 includes regeneration.

In contrast with the MAI for cubic foot volume, the board foot volume MAI for the final measure interval continued to increase for all treatments over 35 years, but the increases were smaller than the standard errors between plots. The final MAI in board feet volume was 1,467 for 25 percent retention, 1,694 for 50 percent retention, 1,704 for 75 percent retention and 1,528 for the unthinned control. For board foot volume MAI there were significant differences for both through the study period and by treatment ($P < .01$).

Total yield

To address the original management question of effects of different commercial thinning intensities on total yield, both the commercial thinning yield at year 40 and the theoretical final harvest yield at year 75 were considered. At the Thirty-five years post-harvest, the highest yield was realized in the 50 percent treatment block (*table 1*).Total cumulative cubic foot volume per ac results differed slightly from board foot results. The highest two cumulative cubic foot volume per ac in thousands are for the 75 and 50 percent retention (25.0 and 24.6 respectively) with the control at 23.1 and 25 percent retention at 21.3.

Table 1—*Compilation of total yield at stand age 75 using thousand board feet volume (Mbf) per ac. Commercial thinning treatments retained a range of initial basal area (100 to 400 ft^2/ac).*

Treatment	Age	Thousand board ft/ac volume		
		Cut	Standing	Cum-ulative
25%	41	31	17	48
	46		23	54
	51		32	63
	56		43	73
	62		53	84
	70		70	100
	75		78	109
50%	41	17	31	48
	46		40	57
	51		51	68
	56		64	81
	62		79	96
	70		100	117
	75		111	128
75%	41	5	40	46
	46		50	55
	51		61	66
	56		72	77
	62		85	90
	70		106	111
	75		116	121
100%	41	0	39	39
	46		49	49
	51		60	60
	56		73	73
	62		87	87
	70		105	105

75		119	119

Stand density

Stand Density Index (SDI) is a relative density measure that describes the level of inter–tree competition experienced by individual trees growing in a stand (Tappeiner and others 2007). The maximum theoretical SDI value for coast redwood is 1,000 (Reineke 1933) on an ac basis. The highest SDI value (887) was derived for the control in 2005 (*fig. 5,* above). The final SDI for 75 percent retention was 707. The final SDI for 25 and 50 percent retention treatments including regeneration were 385 and 556 respectively. The understory comprised about 22 percent of the entire SDI for 25 percent retention and 4 percent for the 50 percent retention.

Calculation of the SDI by the summation method provided lower values than those calculated in prior work (Lindquist 2004). Using the equation in prior studies; SDI= TPA $(D_q/10)^{1.605}$ where Dq is the quadratic mean diameter the result is 917 versus 887 for the control in 2005.

Understory regeneration

The redwood sprouting response after commercial thinning was inversely proportional to the overstory density. No sprouts were found in the un-thinned controls. The 75 percent retention plots had 5,130 redwood sprouts per ac in 1972. Twenty one years later these sprouts had diminished to the point that they did not merit further study (Lindquist 2004). The 25 and 50 percent retention plots had a similar number of sprouts initially (more than 10,000 sprouts/ac in 1972).

The regeneration measured in 25 percent retention treatment had a larger diameter than in the 50 percent retention treatment at the end of the study period as shown in *figures 6* and *7* ($P < .01$). Seedling in-growth tallied in 2007 had a higher proportion of western hemlock and grand fir than the first wave of regeneration in this study; 36 and 80 percent respectively in the 25 and 50 percent overstory retention treatments. After 35 years the 25 percent retention plots regeneration stems (> 4.5 inches DBH) was measured to be 451 cubic feet per ac.

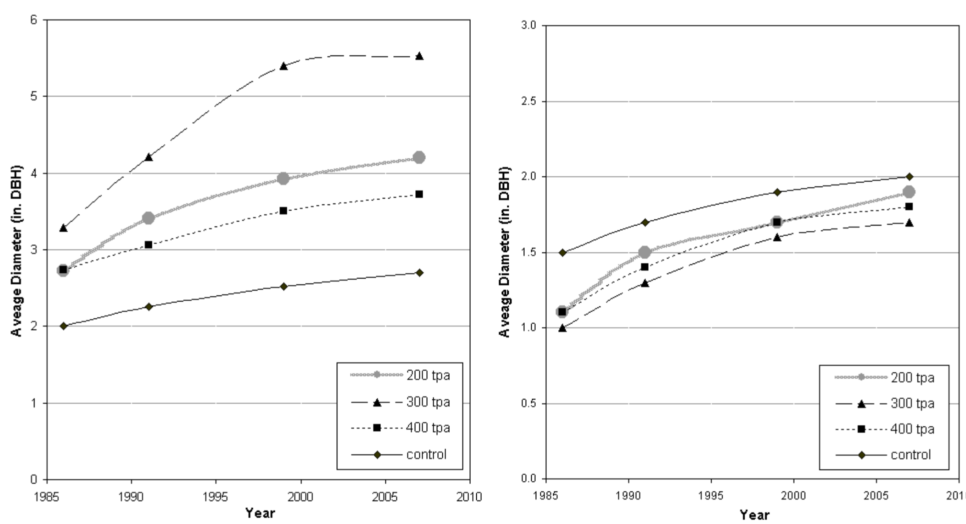

Figure 6 and 7—Growth of the redwood sprouts precommercially thinned to a range of densities within a commercially thinned to retain 100ft.^{2}/ac of basal area per ac (*fig. 6,* left) or 200 ft.^{2}/ac (*fig. 7*, right). Note the diameter scale for the 50 percent retention on the right is twice as large as the 25 percent retention on the left.

Discussion

As a result of the 35 year study period, each experimental treatment resulted in unique and quantifiable stand structures. The greatest mortality occurred in the control plots. Self thinning (onset of mortality) is expected to occur above an SDI of 550 (Tappeiner et al. 2007). The control was above this density threshold for the entire study period.

A substantial portion of the stems in the untreated control remained below the 10.5 inch threshold for production of board feet, but above the 4.5 inch threshold for cubic foot volume. Volume results vary between treatments depending on which volume measurement attributes were used.

The updated height estimation methodology resulted in slightly greater height estimates for the control trees than Lindquist's prior methods (2004). Diameter/height relationships in the study area are influenced both by site quality (plot basis used by Lindquist) and stand age (this study). Though height data is limited for early measurements, further exploration of this data could clarify results.

The 75 percent retention produced a total yield with slightly lower board foot volume than the 50 percent. The 75 percent retention consistently had the highest cubic foot MAI. Some sprouting was observed during the first decades after treatment, but the sprouts did not persist. O'Hara and Berrill (2010) noted complete sprout mortality at low light levels. The 75 percent retention surpassed the SDI threshold of 550 in 1980 and the first mortality was noted in 1985.

The 50 percent retention had quadric mean diameter of 26 inches at the end of the study period. This density level may offer log value or aesthetic benefits when the final diameters are compared to other treatments.

At the end of the study period the 35 year old stems under the 50 percent retention treatment averaged less than two inches in diameter. The SDI for this treatment exceeded the 550 threshold at the end of the 35 study period. Leaf area index is considered to be more integral to stand productivity and growing space occupancy than other measures (Berrill and O'Hara 2009). These authors conclude that removing a large fraction of the basal area such as this 50 percent treatment can promote a vigorous understory and high productivity. The observed slowing of understory diameter growth with time since treatment suggests that the reentry period is critical to maintaining understory vigor.

The 25 percent retention provides about 85 percent of the highest board foot yield (*table 1*), and has a mean diameter of 33 inches with an established understory. The initial 40 year old even aged stand had an average diameter of slightly over 10 inches in contrast to the now 35 year old regeneration with a treatment-wide average diameter of approximately four inches. This treatment does seem to provide a viable route to creating a multi-age stand if reentry periods are carefully planned.

Conclusion

The first phase of the Whiskey Springs commercial thinning study has concluded in 2011 with a second thinning. The next phase will have a demonstration and education objective because the plot size and corresponding edge effects make future experimentation problematic at this site. The trees are now twice as tall as they were 35 years ago.

Long-term permanent plots in complex terrain present opportunities as well as challenges. An experimental site must provide adequate uniformity so that treatment effects are not confounded by site influences. Treatment buffers surrounding study plots need to be sufficient size to minimize edge effects. In addition, remeasurement must be continued over decades. The Whiskey Springs provided valuable research for the first 35 years. It confirms the importance of long-term studies and has provided insight beyond the original focus. A future California Forestry Note will include more details and additional analysis. The study area's resulting stand will continue to tangibly illustrate density management tradeoffs.

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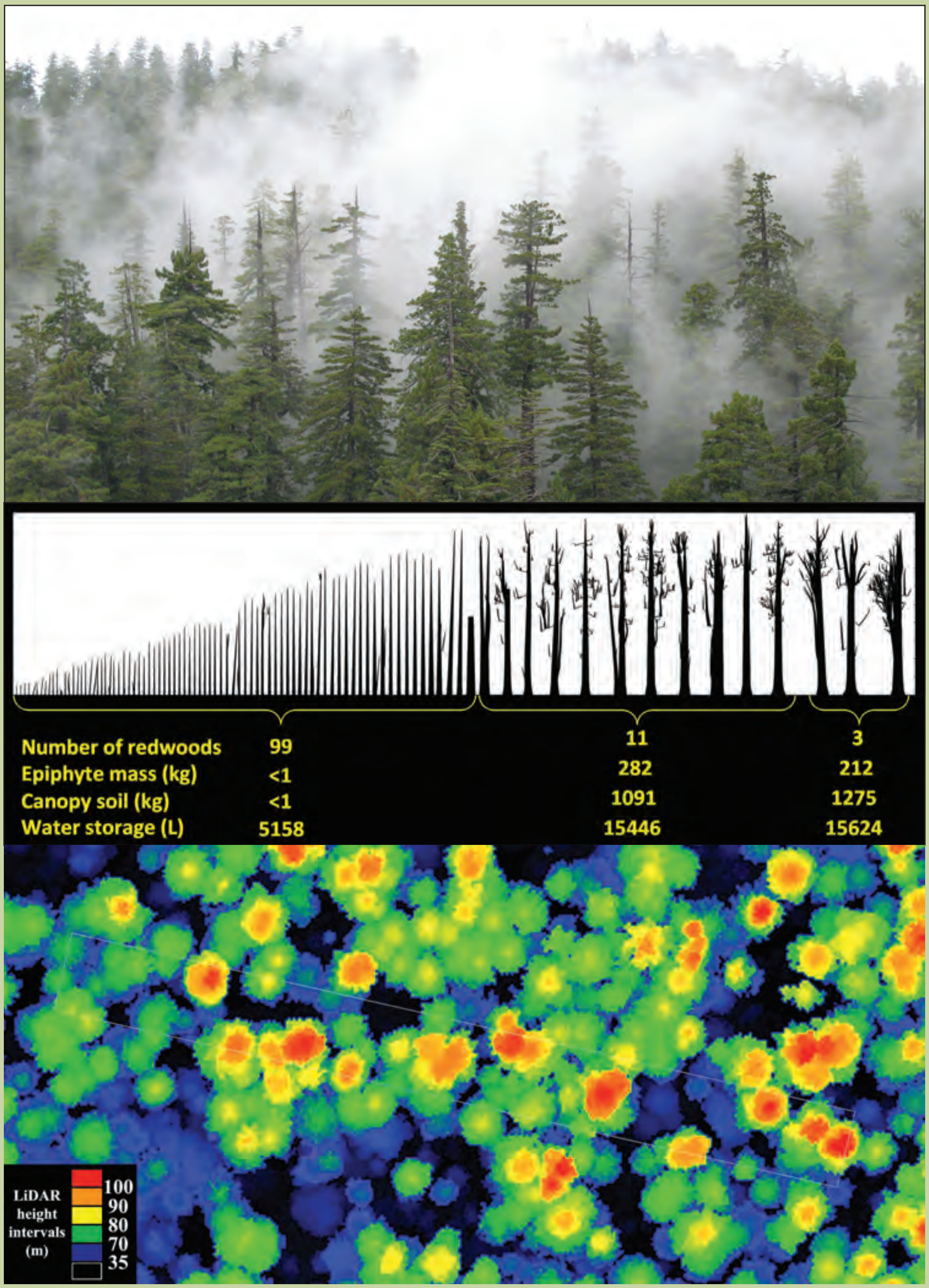
Economics


Number of redwoods 99 11 3
Epiphyte mass (kg) <1 282 212
Canopy soil (kg) <1 1091 1275
Water storage (L) 5158 15446 15624
LiDAR height intervals (m)
100
90
80
70
35

Observations About the Effectiveness of Utilizing Single Tree Selection Silviculture in Redwood Forestlands

Bob Berlage and the Big Creek Lumber Company Forestry Department Staff[1]

Abstract

Harvesting in predominantly redwood forests has been ongoing in the Santa Cruz Mountain region for over 150 years. Under California Forest Practice rules specific to the Southern Subdistrict of the Coast District, clearcutting has been outlawed since 1970. Since that time, single tree selection has been the only silvicultural practice allowed in the Southern Subdistrict. Big Creek Lumber Company has been practicing some form of selective harvesting throughout coastal redwood forestlands in the Santa Cruz Mountains since 1946. Sixty-five years of experience makes it is possible to form general observations about the effectiveness of this silvicultural practice within a redwood forest environment. Increasing population and urban sprawl have created pressures on redwood forestlands in California, and particularly on the Central Coast. Tensions resulting from population increases and ongoing urban encroachment into forestlands in the Santa Cruz Mountains have increased over time. This has created significant logistical and socio-political challenges for the local forest products industry. Not surprisingly, these challenges are now beginning to be seen elsewhere in the redwood region. Selection harvesting can provide positive benefits, particularly adjacent to densely populated areas. These benefits include providing local, sustainable products for local consumers, supporting working forestlands that provide a buffer against the pressures of land conversion and urban sprawl, as well as being a mechanism for maintaining complex redwood forest ecosystems.

Introduction

This paper is not intended to be a scientific analysis. Rather it is a case study and compilation of experiences gleaned from forest resource professionals over the past sixty years. Particular thanks go to Frank (Lud) McCrary, Homer T. (Bud) McCrary and Dale Holderman, Registered Professional Forester No. 69, for their perseverance in seeing the concept of single-tree selection forestry through to a successful outcome in the Santa Cruz Mountains.

Harvest history

The extraction of forest products in the Santa Cruz Mountains by European settlers undoubtedly commenced on their arrival. However, the first recorded commercial lumber ventures were those of Thomas Larkin and Jose' Amesti during the year 1832, in what is now Santa Cruz County. These operations were very crude. Logs were placed straddling a pit and a worker above and below the log would run a whipsaw down the length of the log to create boards. Mechanical sawmilling began

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around 1841. The first mechanical mills used water power to drive the saws. During this time, draft animals (oxen and horses) were primarily used to transport logs to the mills from the forest and lumber to the end user.

By the 1850s, steam began to replace water flow as the power source in many sawmills. Steam driven log yarders (steam donkeys) were used in woods operations starting in the latter 1880s. With the development of new technologies, the rate of harvest increased. Two seminal events contributed to increased forest resource extraction in the Santa Cruz Mountains. The first was the 1849 California gold rush. While few local forest products were actually delivered to the Sierra gold country, there was a tremendous demand in San Francisco, which had become the primary hub for materials and manpower destined for the gold fields. The second event that spurred forest resource production in the Santa Cruz Mountains was the 1906 San Francisco earthquake and fire, which destroyed many of the existing wood-frame structures. By 1897, there were 32 fully operational sawmills in Santa Cruz County alone. The procurement, production and transporting of forest products was the primary industry in Santa Cruz County from 1850 to 1925.

Not all felled timber was turned into dimensional lumber. In fact, it is likely that more old-growth redwood was cut in the Santa Cruz Mountains to make “split stuff” (shakes, fence posts, railroad ties and grape stakes) than lumber. Hardwoods, logging debris and mill ends were used for fuel in heating, cooking and the generation of steam power. Tanoaks were felled and the bark transported to local tanneries. Clearcutting of the redwood/Douglas-fir conifer forests of the Santa Cruz Mountains continued more or less unabated until the mid-1920s. By 1930, most of the contiguous stands of old-growth timber had been cut or transferred to park land.

Big Creek Lumber Company

Big Creek Timber Company was established in 1946. The Company’s first sawmill was located approximately 5 miles up Waddell Canyon on land that is now part of Big Basin State Park. This forestland had been previously harvested by William Waddell from the mid-1860s until 1872. In 1946 this forestland was comprised of some residual old-growth and robust stands of small second-growth redwood and Douglas-fir ranging from 70 to 80 years old.

State regulations at that time required that four “seed trees” per acre, 18 inches in diameter or larger, be retained. All other trees could legally be cut. However, Big Creek Lumber decided to not cut that heavily and instead selectively harvested some of the residual old-growth trees and some of the larger second-growth. Trees that were too small to be economically profitable and trees that were too large for the sawmill were left uncut. The company continued sawmill operations in Waddell Canyon until 1955 when Waddell Creek experienced a significant flood. The company moved its sawmilling operation to the west side of California Highway 1, near its present location. The company became incorporated in 1960 and the name was changed to Big Creek Lumber Company. After 1955, Big Creek Lumber began to concentrate on harvesting second growth timber.

Development of single tree selection silviculture in the Santa Cruz Mountains

The development of selection harvesting in the Santa Cruz Mountains was more a gradual transitional awareness than it was a singularly planned outcome. Big Creek Lumber Company concluded that the kind of forest management practiced on the North Coast of California at that time would not be acceptable to the local public. Urban development had slowly encroached into Santa Cruz Mountains forestlands and by 1946 there were few potentially harvestable forested parcels that did not have some degree of residential influence. From an aesthetic perspective, an acre with only four seed trees was not significantly different than a total clearcut. It made sense to consider a harvesting strategy that reflected local attitudes and concerns.

From Big Creek Lumber Company's perspective there was also a practical business consideration in its approach to forest management. The composition of redwood forestland in the Santa Cruz Mountains was considerably different in 1946 than prior to 1925. While there was no legal prohibition against harvesting old-growth, most of the larger contiguous stands of old-growth had been cut by 1946. Most harvesting at this time involved cutting scattered residual old-growth and was conducted on a smaller scale than the turn of the century operations. At that time there were far more acres of young second-growth. It became clear that the second-growth constituted a larger and rapidly growing resource base. However, given the improvements in logging equipment, it was also evident that harvesting these second-growth lands under the existing regulatory standard of retaining four trees per acre would have resulted in the depletion of the resource within several decades. Harvesting at more conservative levels not only addressed neighborhood concerns, it also insured a sustainable resource.

Absent any regulatory mandate, local timber operators voluntarily formed the Central Coast Timber Operators Association in July 1956. The purpose of this organization was to create a mutually agreeable set of logging standards beyond what state and local regulations required. The impetus for these self-imposed voluntary standards was the increasing public concern over logging operations and their potential affect on streams, roads and particularly drinking water. At that time there were a few careless logging operators whose lack of consideration for these legitimate public concerns resulted in increasing conflict between neighbors and timber harvesting. On August 14, 1956, the Central Coast Timber Operators Association adopted self-imposed rules which included an assessment of surface water on every proposed timber harvest site to determine whether the water was being used for domestic purposes, rigorous confirmation of property lines and rights-of-way, strict attention to logging slash treatment and a prohibition of log hauling on weekends and legal holidays. The Central Coast Timber Operators Association also began discussions regarding developing practices for improving stream crossings, road and landing construction as well as establishing buffer zones adjacent to creeks.

In 1967, representatives of Big Creek Lumber Company participated in Board of Forestry sub-committee discussions of county-specific forest practice rules. It was during these discussions that the basic principles of selection silviculture began to take shape. These principles would eventually be formally established as government regulation in the Santa Cruz Mountains.

Three operational standards were adopted at this time which formed the basis for single-tree selection silviculture in San Mateo, Santa Cruz and Santa Clara Counties. The first was the 60-40 Rule. This rule stated that no more than 60 percent of trees 18 inches in diameter or larger could be cut during any harvest entry and no more than 40 percent of the trees 8 inches to 18 could be cut per entry.

The second operational standard established at this time was the minimum reentry time period. It was set at 10 years, based on harvest entry intervals being practiced by several local foresters at that time.

The last standard was lopping. This operational concept was first tested for economic effectiveness by Big Creek Lumber Company on a harvest site in San Mateo County in the 1960s. The eventual outcome of this standard was that all logging slash now had to be cut to within 30 inches of the ground. Interestingly it was this requirement that had the most immediate impact on timber operations in the Santa Cruz Mountains. Once operators became responsible for the cleanup of logging slash, the quality of timber operations improved significantly. Timber fallers and equipment operators could no longer knock down or damage smaller conifers and hardwoods, at least not without incurring prohibitive cleanup costs.

In 1973, the legislature passed the Z'berg-Nejedly California Forest Practice Act, enabling legislation that charged the California Board of Forestry and Fire Protection with establishing the California Forest Practice Rules. The Act permitted individual counties to create their own separate logging regulations as long as those regulations were more protective than state regulations. On January 1, 1983 Senate Bill 856 was passed. This legislation removed county authority to regulate the conduct of timber operations. The impetus for SB 856 was that Santa Clara County Board of Supervisors had decided in 1980 to completely disallow timber harvesting within the county.

SB 856 also recognized the fact that counties might have specific needs. The legislation empowered individual counties to petition the Board of Forestry for Special County Rules. In the early 1980s, not long after the passage of SB 856, several counties, including San Mateo, Santa Clara and Santa Cruz, petitioned the Board of Forestry for such rules. Primarily using the threshold of necessity, the Board of Forestry passed some of the requested Special County Rules and rejected others. Interestingly, the rules that filtered out of this process were remarkably similar to the operational standards adopted by the Central Coast Timber Operators Association during the 1950s.

Observations from an industry perspective

From Big Creek Lumber Company's perspective, the development of an uneven-age silvicultural paradigm served several purposes. First, it was an effective strategy to address several public concerns regarding aesthetics and resource protection. Second, it had the potential to provide a dependable and renewable resource over time. There was also a sense of what forestlands managed under selection harvesting might look like over time, but it would be years before there was an understanding of the results of multiple harvest entries.

By the 1950s, foresters and timber operators were well aware of the ability of individual coastal redwood trees to exhibit significant, and at times phenomenal, release growth when competing trees were removed. Cross sections of residual trees frequently showed a small core section of very slow growth and a fast growing outer section. These were small, suppressed trees that released when the adjacent old-growth overstory was felled. The increased sunlight, coupled with an existing established root system created an opportunity for rapid growth. Foresters at that time theorized that selection thinning could result in increased growth on individual uncut trees. Over 60 years of periodic selective harvesting in the Santa Cruz Mountains has demonstrated that this is generally true.

Big Creek Lumber Company’s timber marking and harvesting philosophy has remained fundamentally unchanged both before and after the passage of the Z’berg-Nejedly California Forest Practice Act. Defective trees with little growth potential are harvested, allowing overall stand growth to be distributed on better formed trees. More recently, larger trees with recognizably pronounced cavities or other potential wildlife habitat attributes are generally retained. The remainder of the redwoods are thinned using the principal objective of increasing spacing between adjacent dominant and co-dominant trees. This would not be considered “thinning from below”. It is our experience that limiting harvesting to understory trees results in little growth release in the dominant and co-dominant trees and does not sufficiently open the canopy to allow for successful sprout regeneration.

Careful consideration is also given to the logistics of each proposed timber harvest. It is not sufficient to solely consider silviculture when marking trees for harvest. Logging is a mechanical process and selection harvesting has a value set that does not exist in even-age management: protection of the unmarked trees, also referred to as the “leave stand”. Foresters are trained to consider how a tree will be felled and yarded when they mark timber. It is counter-productive to mark timber that cannot be felled or yarded without breaking the marked tree or damaging the leave stand. Our newly hired foresters spend significant time working directly with logging crews in order to understand the mechanical principals of logging operations. Big Creek Lumber has always placed the responsibility for road and log landing layout with its foresters. At the same time, logging crew personnel are trained to minimize damage to the leave stand when conducting yarding operations.

Three timber yarding methods are utilized in local selective harvesting; ground-based tractor/skidder yarding, cable yarding and helicopter yarding. Terrain and infrastructure usually dictate which yarding method is used.

Ground based yarding

This method is utilized when the terrain permits and there is either an existing skid road system or the ability to construct skid roads without creating potential environmental problems. Cable and helicopter yarding methods are used when the terrain is excessively steep, the forestland is inaccessible or new road construction is not feasible. Ground-based yarding is the least expensive. Particular care is taken to minimize soil disturbance when conducting ground-based operations and all skid roads are seeded, straw mulched or covered with logging slash to prevent erosion following harvest operations. A direct advantage of this yarding method is that tractors can be used to distribute logging slash on skid roads and then crush the slash to create an effective barrier against erosion.

Cable yarding

This method is primarily used when the terrain is too steep for ground-based yarding. Cable yarding is operationally more challenging in a selection harvest because of the need to protect the leave stand. Cable corridors are designated and trees are felled on an angle towards the corridor in order to minimize damage to standing timber. Local operators are able to maintain cable corridors no wider than 15 feet. Within several years the canopy closes in over these corridors and they are no longer visible.

Helicopter yarding

This is the most expensive method, roughly twice the cost of ground-based yarding. It is primarily used when there is neither existing road infrastructure nor the ability to create new roads. It is the least intrusive relative to ground disturbance. Because of steep terrain and limited road access, lopping of logging slash on cable and helicopter operations is restricted to hand crews and is considerably more expensive and less effective than lopping operations on tractor-logged ground.

The 60-40 cutting rule has been in effect since the 1960s, first as a self-imposed operational standard, and then as California Forest Practice Rules specific to the Southern Subdistrict of the Coast District. While this rule technically allows sixty percent of the trees over 18 inches in diameter to be harvested every ten years [2], the reality is that few properties have been cut that heavily during an initial entry and almost no forested parcels have been harvested at that rate repetitively. In reality, logging in the Santa Cruz Mountains has evolved into a self-regulating system that has resulted in harvest levels that have consistently fallen short of the regulatory limits. There are several reasons for this.

Under certain circumstances in a densely stocked second-growth forest, a heavy initial harvest level can be justified from a silvicultural perspective. It can stimulate accelerated growth in the leave stand and promotes successful regeneration. An argument can be made for a slightly less aggressive harvest during a second entry, but heavy cutting on short intervals over time does not make sense economically and could theoretically result in smaller average tree size. Simply put, the growth may not be sufficient to make future harvest operations economically desirable. The loggers, sawmill and landowners do not necessarily want small trees and the landowner has an expectation of a reasonable return from harvest operations and their commitment to responsible land stewardship. To be certain, there are a number of variables that affect decisions about harvest levels and entry periods. For example, the growth potential of the forestland is an important factor. Nevertheless, the result over time is that the harvest levels in the Santa Cruz Mountains have not approached the regulatory limits and the average size of trees harvested has consistently increased.

Looking forward

Responsible forest management in the Santa Cruz Mountains faces considerable challenges. These challenges are not dissimilar from those occurring elsewhere in California or the nation, but they began earlier here than in other areas and are

[2] In Santa Cruz County, if the harvest level is between 50 and 60 percent, the minimum reentry period is 14 years.

frequently more intense. The Santa Cruz Mountains have been subjected to urban sprawl, including development encroachment into forestlands for more than a century.

More recently, the transition of the Santa Clara Valley into a regional economic powerhouse has predictably placed extreme land use pressures on adjacent rural lands and particularly on local forestlands. Properties that historically were owned and maintained with periodic selective harvesting as an objective have now became desirable as an upscale bedroom community for Silicon Valley. This dynamic not only creates competing economic pressure on local forestlands, it has also resulted in a population of new residents who have little knowledge of local logging practices of the area's longtime history sustainable forest management. Communicating this information to a constantly changing general public, their elected representatives and various government regulatory agencies is an ongoing necessity for the local forestry community.

One of the current challenges to maintaining sustainable forestry and a forest products industry in the Santa Cruz Mountains is the loss of viable timberlands to preservation policy. In the last 30 years, tens of thousands of acres of potentially harvestable forestland have been taken out of production by park and open-space acquisition. While many of these lands technically could be harvested, that has generally not been the policy objective of those administering these lands.[3] Encouragingly, several open space organizations are now considering the potential of responsible forest management as a mechanism to achieve their conservation goals. Time will tell what decisions are made.

In conclusion, selection harvesting has evolved into an effective and sustainable forest management methodology within the Santa Cruz Mountains. It was not created by government regulation alone, but is better explained as the result of multiple inputs which include: industry initiative, social and political pressure, as well as environmental considerations. While this kind of forest management works well in the predominantly redwood forestlands of the Santa Cruz Mountains, it remains to be seen whether it would be appropriate in other forest ecosystems.

[3] A significant exception to this dynamic is the Byrne/Milliron forest owned and managed by the Land Trust of Santa Cruz County. This sustainably managed and harvested parcel has generated over $1.1 million, funds which in turn have been used by the Land Trust to support their conservation work.

The Watershed TMP: a Proposal to Manage the Redwood Ecosystem Under Convergent Environmental, Economic and Social Goals

Frederick D. Euphrat[1]

Abstract

Under present California Forest Practice Rules, mandated by the Legislature and codified by the State Board of Forestry, Non-industrial Timber Management Plans (NTMPs) give small landowners the flexibility to operate under a specific set of rules 'forever,' allowing short notice (3 days or less) for pre-approved harvest operations. This permit is presently restricted to forestland ownerships less than 2,500 acres, requires uneven-aged management (selection harvest), and allows forest managers to both react to market conditions and have certainty of regulation. There is no comparable tool for larger ownerships. Using this concept for larger ownerships, with an explicitly directed primary outcome of wildlife, watershed benefits or carbon sequestration, and timber as a subsidiary output, would improve environmental quality, reduce landowner costs and reduce regulatory time for forestry under the California permit process. A Legislative bill or Board of Forestry initiative could create a Watershed Timber Management Plan as a low-intensity, watershed-scale forest management permit. This tool would be particularly useful in California's coast redwood ecosystems, where selection harvest is a demonstrated way of managing for both ecosystem services and timber production.

Key words: silviculture, watershed, forest policy, forestry, California, redwood, NTMP, natural capital, ecosystem services

Background

Private timberlands in California fall under control of the State Legislature. The Z'Berg-Nejedly Forest Practice Act of 1973 and amendments give the State Board of Forestry (Board) the formation language for the California State Forest Practice Rules (FPRs). That legislation reflects the intent of the State Legislature to:

> "create and maintain an effective and comprehensive system of regulation and use of all timberlands so as to assure that:
>
> (a) Where feasible, the productivity of timberlands is restored, enhanced, and maintained.
>
> (b) The goal of maximum sustained production of high-quality timber products is achieved while giving consideration to values relating to sequestration of carbon dioxide, recreation, watershed, wildlife, range and

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forage, fisheries, regional economic vitality, employment, and aesthetic enjoyment.[2]

The implementation of this intent is the responsibility of the Board through its rulemaking procedures, then accomplished through the agency of the California Department of Forestry and Fire Protection (CalFire). In practice this has created a set of 'classes' of permits for timber harvesting: timber harvest plans (THPs), modified timber harvest plans (MTHPs), non-industrial timber management plans (NTMPs), sustained yield plans (SYPs) (still requiring a THP), program timber environmental impact reports (PTEIRs), exemptions and emergency notices. Additional federal permitting requirements for large ownerships may also mandate or justify a Habitat Conservation Plan, to be implemented and referenced in subsequent THPs. While this plethora of approaches exist, choices for landowners narrow quickly.

THPs are flexible, may cover large areas, and are good for 3 years, with extensions granted by CalFire. All silvicultural (harvest) methods are allowable in THPs. MTHPs are for parcels 100 ac and less, and do not allow even-aged (clearcut and functional equivalents) methods, among other significant restrictions. MTHPs are time-limited in the same manner as THPs. SYPs are intended to be a supporting element for THPs over a broad area, but are used little because of the documentation required for the THP. PTEIRs are also for very large areas, establishing a local set of forestry rules through an EIR process, followed by subsequent Program THPs, smaller in scope than a full THP. PTEIRs are a very useful tool for local 'regulations', though they are acknowledged to be complicated, expensive, and require broad buy-in to begin.

NTMPs are allowed for landowners with less than 2,500 total ac, and require uneven-aged management (selection harvest). NTMPs are permanent. With 50 or more years of harvest and yield projections, NTMPs are intended to be continued into the future with the simple notification of a Notice of Timber Operations. Some elements of NTMPs must remain up-to-date, such as water quality and species considerations administered through other agencies.

It is important to note that these options trade size for silviculture (*table 1*). But this means that large landowners, such as ranchers, organizations and industry, cannot make the same trade. This is counterintuitive; the landowner, the public and the ecosystem all need the opportunity for and benefits of less intensive silviculture over large areas, particularly individual watersheds that may be of great natural capital value, such as for domestic water supply or listed fisheries. Without large, low intensity operations, forest managers tend to seek the most cost-effective solution of compressing harvests into a set of THPs, with concomitant high-yield cutting patterns. It is this approach that may define checkerboard clearcuts as the most cost-effective management approach (*fig. 1*). While this even-aged matrix may be as effective at protecting forest resources as an uneven-aged system, no option presently exists for landowners who specifically want to devote land to ecosystem services while still harvesting trees.

[2] Z'Berg-Nejedly Forest Practice Act, Division 4, Chapter 8, Public Resources Code, sec. 4513. Effective January 1, 2010.

Table 1—*Selected California Forest Practice Rule Harvest Permits.*

CalFire Permit	Maximum size (acres)	Silvicultural limitations	Time Period	Additional Permit required
MTHP	100	Uneven-aged	3 years*	No
NTMP	2,500	Uneven-aged	Permanent	No
THP	None	None	3 years*	No
SYP	None	n/a	Permanent	THP
PTEIR	None	Within PTEIR frame	Permanent	PTHP
Exemption: Dead, Diseased and Dying	None	10% cut, only damaged trees	1 year	No
Emergency Notice	None	Only damaged trees**	1 year	No

* May be extended in one-year amendments.
** With exceptions.

Discussion

There is a bias in the options available for California private forest management. Managing below 2,500 ac favors uneven-aged management, granting unchanging rules and short notice harvests. No permit alternative exists for larger lands to ‘trade’ management intensity for increased certainty of harvest and reduced cost in the future. In CalFire’s own assessment of the cost-effectiveness of NTMPs:

> “The cost of preparing an NTMP is about 25 to 50 percent more than a typical THP, much of which comes from the required sustained yield analysis. However, this cost is recaptured over time because subsequent NTMP harvest entries can be conducted under a much simpler notice to CDF, which triggers the inspection and enforcement process.”(CalFire 2003).

The bias exists because the Legislature and the Board of Forestry identify two classes of landowners: Non-industrial private forest landowners (NIPFs) and industrial landowners. NIPFs do not own more than 2,500 ac of timberland in California and are “not primarily engaged in the manufacture of forest products” (see footnote 2). CalFire estimates that more than half a million additional acres would be available for NTMPs if the maximum were increased to 5,000 ac (CalFire 2003).

There is no ‘industrial’ alternative to NTMPs, trading uneven-aged silvicultural practices for long-term certainty in regulation. This seems counterintuitive; California’s sequestration of carbon dioxide, recreation, watershed protection and wildlife can all benefit from uneven-aged management. For example, carbon sequestration of an uneven-aged forest is effective because (1) Climate Registration Protocols require a baseline of timber to be in place before ‘additional’ timber can be valued (Climate Action Reserve 2010), and (2) large trees, in general, put on more volume of wood per acre per year than small trees (Lindquist and Palley, 1963). Some industrial owners operate successfully using only uneven-aged systems, such as Big Creek and Mendocino Redwoods (MRC), operating within the THP system and (in MRC's case) developing an HCP, as well.

The PTEIR may be a choice for large landscapes, but Calwater watersheds and other intermediate size areas, from 2,500 ac to 15,000 ac, are too small for such an investment. In addition, the PTEIR harvests, designed for multiple ownerships, are managed via individual Program THPs with their own costs.

Prudent management

In California and the redwood region, cut blocks in even-aged management are regulated by the FPRs, with a maximum size from 20 to more than 30 ac. Adjacent areas must be restocked for 5 years prior to even-aged treatment, with logical boundaries in between cut blocks. *Figure* 1 shows the Gualala River watershed, managed with even-aged silviculture.

Figure 1—Even-aged redwood silviculture in the Gualala River watershed.[3] (photo: Fred Euphrat).

Seen from a statewide perspective, permanent forestry plans with a lighter impact would be useful for managing important landscape elements, such as streams and watersheds, wildlife habitats and corridors, and urban interface areas. Consider that the average Calwater 2.2a planning watersheds units recognized by CalFire are 7,537 ac (Smith 2008); that habitat for many species is reduced by fragmentation (Andren 1994); that biological corridors require continuity (Noss and Cooperrider 1994); and that parcel development reflects a loss of cost-effectiveness of forest management. The corridor issue is addressed, to an extent, by the FPRs - stream corridors provide riparian passage. Other biodiversity hotspots, such as ridgetop and saddle corridors, edge to forest ratio, and the encroachment of urban and agricultural areas remain unaddressed. Larger, permanent uneven-aged plans may mitigate many of the

[3] www.krisweb.com. 1999. Gualala watershed, hypothesis no. 3. “Figure 9. Clear cuts in unstable, inner gorge locations of upper Little North Fork Gualala River in 1999. Arrows designate landslides.

biodiversity problems of industrial timber management while making it more cost-effective.

Indeed, the State Legislature finds forest management, in combination with uneven-aged methods, "prudent," in its enaction of NTMP legislation. The intent in section 4593 of the Public Resources Code is:

> (b) The Legislature further finds and declares that minimal environmental harm is caused by prudent management of nonindustrial timberlands because low volume production and dispersion around the state of these small tracts reduces damage to aesthetics, air quality, watersheds, and wildlife.[4]

Recognizing that the Legislature's finding of 'prudent' forestry also embraces low-volume harvest and dispersed production, the inclusion of uneven-aged management allows demonstrated protection of aesthetics and habitat (Thornburg et al. 2000), particularly with the inclusion of old individual 'heritage' trees and group selection.

It is a timely moment in California to transition to uneven-aged management. For example, a two-aged stand is shown in *fig. 2*. This is the kind of forest where single-tree selection can both harvest the larger trees and promote the growth of smaller ones. The larger trees put on more high quality volume; the smaller trees fill canopy gaps and increase growth in response to additional light and space. Uneven-aged stands may be harvested every 10 to 15 years and produce an even flow of timber while maintaining a continuous forest, depending on the site and other tree growth characteristics (Arvola 1978).

The watershed approach to forest management

This discussion has been oriented towards the use of uneven-aged silviculture as a method to increase the quality and quantity of forest benefits. This is reflective of the considered opinion of the Legislature, which supports uneven-aged management with the reward of the permanent NTMP. But this belies the fact that there are many approaches to uneven-aged management, and there is no direct causal link recognized between specific forest management and particular outcomes in the FPRs. Selection harvesting is considered better for NTMPs, but without benefits stated explicitly.

It stands to reason that, if goals were explicit, with cause-and-effect relationships that could be measured, inventoried and modeled, the State would get greater benefits from forest management. Goals of carbon sequestration, watershed or wildlife management could be implemented with certainty, and the State would be able to quantify those benefits. This does not mean moving away from timber; it means using silviculture to achieve other benefits, rather than expecting trickle-down benefits from a purely silvicultural approach.

A Watershed TMP would allow forestry for specific attributes determined by what is important in a locality. Examples are: watershed management above sensitive waterways, habitat management for particular species, or aesthetic and fire

[4] Division 4, Chapter 8, Public Resources Code, sec. 4593 (b).

Figure 2—Two-aged redwood stand in Jackson State Demonstration Forest (photo: Fred Euphrat).

management adjacent to residential zones. All of these require varied forest management. The Calwater Planning Watershed system is a reasonable first step of organizing and delineation. Multiple watersheds can be used to create larger elements but, under a Watershed TMP, would still be modeled, evaluated and permitted by watershed to address local issues.

Expanding the NTMP approach to more timberlands would create both increased certainty and social benefit. In the words of CalFire,

> "This change would benefit both landowners and the state by providing an opportunity for these additional timberlands to be placed into a sustained yield and uneven-aged management regime." (CalFire 2003).

It must be stated that industrial timberlands already meet criteria for sustained yield under the FPRs. Expanding an NTMP approach may not, in itself, make the lands more 'sustained,' as cutting patterns would shift, though timber output would not necessarily change. But it would clearly expand the opportunity for uneven-aged management to benefit California.

It must also be stated that, on a watershed level, it is not the intention of this paper to propose that selection cutting has less impact, over time, than even-aged methods. Good road practices, revegetation and harvest methods appropriate to stand dynamics are critical to any harvest. Rather, uneven-aged methods, with their concomitant combination of closed canopy forests and their continuous accumulation of carbon is preferred by the public, via the legislature, to be an action of less potential negative consequence, thus requiring less scrutiny, as in the NTMP legislation discussed above.

In the redwood region, the utility for Watershed TMPs may be greater than the rest of the state. Redwoods coppice and are relatively shade tolerant. The highly productive redwood forests produce effective (though not biological maximum) growth in uneven-aged conditions. Forests have been effectively managed through uneven-aged silviculture for decades, and are a sustainable source of wood and jobs.

Model rules

A Watershed TMP would require its own legislation and its own section in the FPRs. Elements that section would include are:

- A maximum size of one planning watershed.
- Disconnected sub-areas within one planning watershed OK.
- Ability to reasonably adjust Calwater lines for property and resource management.
- One landowner.
- No assessment of total landowner forestland holdings.
- No differentiation between industrial and non-industrial ownership.
- Only uneven-aged management.
- Primary resource NOT timber.
- Explicit goals for outcomes of primary and subsidiary resources.
- Inventory and Modeling of primary resource.
- Inventory and Modeling of timber.
- Re-inventory schedules.
- Accommodation for other agencies' review as necessary through time.
- Plan setting in context with adjacent parcels, permits and Watershed TMPs.
- Permit for harvest by Notice of Timber Operations.

The political landscape

In California there is a constituency for every side of an issue. Conversations regarding a Watershed TMP approach have yielded a variety of responses. One forest manager said, 'Of course, we already do that, we just don't get any credit for it.' The Watershed TMP may be welcomed by industries that are already using selection forestry to their current advantage. This is also a good tool for industry lands seeking compliance with Federal Habitat Conservation Plans. In most cases, this would reduce cost for the submitter and the agency, and may well bring significant acreage into uneven-aged management for specific ecosystem services.

An environmental advocate said 'No way will we accept permanent logging plans.' But advocates are not all of one mind. Others see this as an opportunity to focus on particular watersheds, species or aesthetics to the benefit of the community and its job base. It allows public trust advocates to look at a manager's big picture, rather than having to address watershed via THPs in a piecemeal fashion. Permanent plans with continuous monitoring allow for flexibility and focus on locally important resources. The Watershed TMP could spread the benefits of 'prudent' management throughout the redwood region and the state.

A Watershed TMP scenario

An industrial manager chooses to prepare a Watershed TMP, with combined goals of water quality, habitat, carbon sequestration and timber harvest. The plan will cover about 7,500 ac. That area will modify the Calwater watershed, to better encompass a watershed on the property. That watershed is adjacent to two other Watershed TMPs, so conjunctive planning will occur.

Emphasizing water quality and carbon accumulation, smaller watercourses will be cut less than allowed by the rules. Overall stocking will be maintained at 20 percent above FPR floors. While clearings will be limited to the regulatory maximum of 2.5 ac, their locations will avoid habitat elements, preserving saddles and creating continuity with adjacent watersheds. Cutting frequency will be increased relative to even-aged silviculture, necessitating an active road system, which will allow constant monitoring.

Following a higher up-front cost for the Watershed TMP, significantly less capital will be required for later permit compliance. Jobs will be less cyclic, because of the certainty of this and other plans, adding regional economic vitality. As now, the entire area will continue with owl surveys, murrelet surveys and wildlife inventory, and stay apprised of changes in State and Federal species listings. Carbon sequestration will continue to progress on site, due to standing trees on site and quickly increasing volume.

Conclusion

The Legislature has created a two-tier system of forest landowners in California, creating divergent opportunities and outcomes. Landowners with less than 2,500 ac are given the opportunity to manage forests with uneven-aged methods and a permanent plan in the NTMP process. The legislature considers uneven-aged methods part of a 'prudent' approach. Larger landowners, specifically forest industries, must manage with short-term permits and face changing CalFire rules. While landowners with greater than 2,500 ac may choose to use uneven-aged methods, there is no incentive to manage forestlands for resources other than for timber productivity.

A Watershed TMP would provide a reason for industry and other forest landowners with greater than 2,500 ac to produce specific benefits from uneven-aged management. The incentive is a permanent permit at a watershed scale of one Calwater planning watershed, which would require only a Notice of Timber Operations to proceed, in addition to statutory requirements from other agencies. The Watershed TMP would be oriented towards one or more of many benefits received from forests as well as timber, with inventory and monitoring to match.

The goal of Watershed TMPs is to allow large landholdings to use uneven-aged management across as broad a range of California forests as possible. It allows landowners to evaluate the range of positive outcomes from silviculture. It promotes multi-watershed planning of wildlife habitat. It helps create long-term certainty in jobs and local economies, and sequesters forest carbon more effectively than other means.

At this writing, the California Legislature is considering SB 455, Watershed Timber Harvest Plans[5]. While similar in some ways to the proposal above, it differs in that it would not require uneven-aged management, plans would be larger than one planning watershed, and requires both a watershed and fish and wildlife assessment. The bill also affects conversions of forest to non-forest uses.

Watershed TMPs may be a tool most effective in the redwood region, because of the relative shade tolerance and resprouting ability of redwoods, the steep slopes and the sensitive species. The natural capital of our forests is great, the ecosystem services they provide are priceless It is time for the law to encourage silviculture to become and active agent in watershed protection, carbon sequestration and habitat protection in California.

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Protecting Forests Across Landscapes and Through Generations: the Sonoma County Forest Conservation Working Group

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Abstract

There are approximately 513,000 acres of coniferous forests and oak woodlands in Sonoma County, California, situated about 50 miles north of San Francisco. Most of the oak woodland, and over 68 percent (132,000 acres) of the coniferous forestland, is in private ownerships of 50 acres and less. These forests are unique, with 10 species of true oak and 19 species of conifer in the county. While Sonoma's forests provide a suite of benefits, factors create pressure to parcelize, convert to viticulture, and fragment the forest ecosystem. In order to address these threats, an ad-hoc committee created the Sonoma County Forest Conservation Working Group (Working Group).

The Working Group provides information and resources to forest landowners with the goal of protecting the health and long-term tenure of forests and woodlands. Lessons learned highlight landowners' interest in fire and forest management, but also a lack of wildland and forest management knowledge, a reluctance to spend money, and institutional gaps that ultimately increase fire danger and ecosystem fragmentation. This experience demonstrates the importance of our mission, 'Protecting Forests Across Landscapes and Through Generations.'

Key words: community forestry, extension, Sonoma, fire control

Introduction

Sonoma County is unique among forested counties in California. Because of its proximity to San Francisco, fertile soils, abundant water and quicksilver mines, the area was settled early relative to much of the state. Its timber resources were accessed by rail in the 1870s for the development of San Francisco (Codoni and Trimble 2006). Most of the land ended up in private hands, generally going back to Spanish Rancho demarcation. For this reason, there is very little public ownership of timberlands or oak woodlands in Sonoma County. Forest health and ecosystem continuity depends, in a very large part, on the private landowner.

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The Working Group grew out of a meeting convened in February, 2005, by Sonoma Land Trust, to talk about issues including small, private parcels of forest in Sonoma County, California. The 'problem' with these parcels is their abundance in the County; Sonoma County is the most highly parcelized county in California (Greenbelt Alliance, 2006). North of San Francisco and Marin, the million-acre county has more than 513,000 acres of true oak woodland (*Quercus* spp.) and coast redwood-Douglas-fir (*Sequoia sempervirens* and *Pseudotsuga menziesii*) forest, about one-third of which is held in parcels of 100 acres or less. In coniferous forests, parcels of 50 acres and less constitute 68 percent, or 132,000 acres[9]. This relatively small size of ownership of a wide-ranging ecosystem creates parcels that are uneconomic for timber harvest or rangeland, historic uses of this land when it was in larger parcels. The general use at this time is 'no-action'.

The no-action alternative is a concern to the colloquium of forest, land trust and trail managers. The county has unique forests with 10 native species of true oak and 19 native conifers[10] (CalFlora Data Base 2011). While most of the old-growth redwood and Douglas-fir forest was logged since the California gold rush, the ecosystem remains. The second growth redwood-Douglas-fir forest is in various stages of succession in response to previous harvests. Most of the seral stages are now fire-prone in this once frequent-fire ecosystem. The oak savannahs are similarly subject to the effects of fire, an indigenous practice in Sonoma's valleys (Anderson 2005). Some species, such as knobcone Pine (*Pinus attenuata*) are fire-dependent. In addition, Sudden Oak Death (caused by *Phytophthora ramorum*), irregular management practices, and land tenure changes all contribute to fire hazard. And most importantly, the smaller size of ownerships with dwellings increases both the likelihood of fire and the severity of fire damage, including the potential for loss of life.

The Working Group formed from a group of the Sonoma Land Trust meeting's participants and has focused on finding those private landowners with 'small' parcels, and providing the tools they need to plan, manage and conserve their land for themselves and the generations to come. The mission of the Working Group is: Protecting Forests Across Landscapes and Through Generations.

Methodology

The core partners in the Working Group are: the Sonoma Land Trust; the Greenbelt Alliance; the State Department of Forestry and Fire Protection (CalFire); the University of California Cooperative Extension; Sotoyome and Gold Ridge Resource Conservation Districts; Sonoma County Agricultural and Open Space District; Fire Safe Sonoma; the Bodega Land Trust; Coast Ridge Community Forest; and interested individual and forestry professionals. People and organizations contributed space for meetings. There is no paid staff. Volunteer members rotate duties to provide agenda, facilitation, and take minutes at monthly meetings, as well as representing their respective organizations, interests, and capacities.

[9] Sonoma County Agricultural Preservation and Open Space District. 2009. Analysis of forested parcels and ownerships in Sonoma County, California. Unpublished.
[10] Sonoma County Agricultural Preservation and Open Space District. 2010. Oak woodland plan. Unpublished.

With the combination of capacity and will, the Working Group has been engaging with the public and learning from its experiences. The Working Group has successfully coordinated Forest Stewardship workshops and two house meetings to date, as well as engaging large rural-subdivision neighborhood residents in dialogue about their needs, possibilities and future options for forest management. House meetings are informal get-togethers in a rural neighborhood, hosted by a resident, with two or three Working Group members to discuss the local situation and to familiarize the community with resources for rural landowners.

Forest Stewardship workshops are all-day classes, with a morning session focusing on forest ecology, roads and fisheries, fire safety, Sudden Oak Death, and conservation easements as a tool for family succession. The afternoon is a field tour for participants to visit sites that highlight the practices discussed during the morning session. Because the Working Group represents a broad base of knowledge and resources, many individuals can connect with the people or program they most need: cooperative extension, land trusts, foresters, federal and state grant coordinators, or even free chipping of dead and thinned brush and trees.

In addition, we have had a series of ‘brownbag’ seminars in Santa Rosa, presenting topical issues related to forests and their management. These are hour-long seminars with one speaker on a specific topic. Presenters’ subjects have included the carbon cycle, carbon sequestration, and monetizing carbon credits; oak ecology and history in Sonoma County; Sonoma County fire history and ecology; and climate change and its potential impacts to the county. The audience for the seminars includes many land and resource professionals from local and regional government, creating an opportunity for people to ‘get to the woods’ while eating their lunch in town. The continued success of the Working Group has included the melding of the brownbag program in Santa Rosa with workshops and house meetings in Sonoma’s rural areas. This balance has developed a needed rapport between the county’s urban center and its forested periphery.

This success has also been driven by the diverse nature of the core partners of the Working Group, which includes representatives from farm, forest, environment, government, academia, non-profit agencies and local volunteers. Based on workshops and outreach, the partners seek to fulfill the public’s need: How do I manage my forest?

Case studies

The cumulative effect of combining agencies and volunteers for a common goal is that many resources are already available, others can be had inexpensively, and some can be derived from the community that receives the benefit. The combination of resources which has worked for the Working Group includes: help with outreach from all three local Resource Conservation Districts and UC Cooperative Extension; educational funding and support from UC Cooperative Extension; grant funding from the Community Foundation of Sonoma County and the Ernie Carpenter Fund for the Environment; donations of food, coffee, field transportation and meeting space from neighborhood groups; and donated time from both volunteer and professional educators. With this, the Working Group was able to engage the most important resource to the forests of Sonoma County, a cross-section of its landowners.

Six workshops and six brownbags have been conducted by the Working Group over the last four years in Sonoma County. They are presented in *table 1* as categorized case studies, because they all differ in content, ecosystem and community. Workshop attendees (*table 1*) do not include Working Group members, and are dominantly local landowners. Because of coordinated outreach and topics of interest, some people travel many hours to attend workshops.

Table 1—*Summary of outreach activities by the Sonoma County Forest Conservation Working Group*.

Workshop or series	Date	Atten-dance	Ecosystem	Field tour
Mark West Creek	9/29/2007	40	Mixed Douglas-fir	Yes
Occidental	10/13/2007	50	Redwood-Douglas-fir	Yes
Cazadero	10/4/2008	38	Redwood-Douglas-fir	Yes
Mill Creek	10/18/2008	48	Redwood-Douglas-fir	Yes
Cavedale Road	3/19/2011	40	Mixed Douglas-fir	No
Shone Farm	5/7/2011	28	Mixed Douglas-fir	Yes
Brownbags	Six to date	25 per	Santa Rosa	No

An average of 40 percent of workshop attendees report either having changed or planning to change land management practices as a result of the information learned. The attendees also felt they had significantly increased their understanding of forest processes, ecology, fire and fire breaks, disease and succession.

Our now-standard Healthy Forests curriculum is forest ecology, watersheds and roads, Sudden Oak Death, and fire safety. We have also presented panels on funding sources, economics of forest management, conservation easements, carbon credits, biomass utilization and lessons from Trinity County.

We speak of the classes as “101” and “102”, because people consistently want more in-depth knowledge of particular topics. In addition to fire planning, heritance planning and ecosystem management, future Forest Stewardship workshops which will address Healthy Forests, Healthy Economies, and Healthy Communities on an on-going basis. Brownbags will continue to address carbon, wildlife management, ecological services, oak woodland management, wood cogeneration, and the forest assets of Sonoma County. House meetings will continue to address the individual concerns of specific neighborhoods.

The Cavedale Road workshop, requested by the community, focused on “What’s killing our trees and what to do about it.” The agenda covered tree health and beetle outbreak in Douglas-fir and madrone (*Arburtus menziesii*), fire safety and the chipper program, and how to protect wildlife habitat while doing fuels management work. This was a less organized, less formal workshop in a person’s home, which generated lots of interest and good feedback. The community has requested that the Working Group present a full Forest Stewardship workshop next fall.

The Shone Farm workshop was also a new approach for the Working Group. On land managed by Santa Rosa Junior College, the theme was Community Forests, with updates on selected forest conservation projects presently in the county. The field trip was a tour of the Shone Farm forest led by the students of recreation at the College.

Discussion

Our lessons learned may be summarized in the following points:

- Stop looking for money to hire someone else. The experts are already in the room.
- Find all the agencies that may have an interest, public and private. Both efficiency and synergy can be created with existing agencies, though they may seem ‘unlikely.’ Everybody wants to help the forest.
- People are hungry for information. While ‘old-timers’ may remember the methods, contractors and agencies related to land management, new landowners are eager to learn.
- People generally do not perceive fire danger in forested lands, nor understand how to prepare for it. With information, they become more motivated to protect their homes and property.
- People are ready to meet and learn, but not so ready to spend money. While fire danger may threaten valuable homes, people are reluctant to pay full costs for clearance. The free chipper program from Fire Safe Sonoma is an effective, tangible incentive.
- The general forest management goals of most landowners are relatively uniform: fire danger reduction, large tree dominant ecosystems, and liberty to manage their lands.
- Many landowners have wildlife, pest, disease or other forest health goals, but generally have neither the knowledge nor the capability to effectively achieve those goals.
- There is a significant need for management strategies to coordinate multiple owners for multi-parcel management. For instance, one rural subdivision near Santa Rosa has 104 parcels, ranging in size from 4 to 40 acres, in about 1000 acres.
- People want to manage sustainably and achieve their goals, but do not have a definition for sustainability.
- People are generally not familiar with the conservation easements that may be used to protect elements of their land for succeeding generations.

Lessons learned are only a part of this; much of the benefit has gone to the individual members of the Working Group. It serves as a networking and support group for all those who do or want to do forestry education and outreach to the interested public.

The Working Group has just finished its first brochure for outreach to the community. We are working to have a web presence, and to develop, publish and distribute a forest landowner’s manual. That manual would outline and elaborate on the information presented in the workshops.

The intention of the Working Group is not to create new information, only to get good information into the right people’s hands. In Sonoma County, behind almost every vineyard is a forest, and above almost every grassland is an oak savannah. In a region that depends on private land ownership to achieve public goals of (1) fire and watershed protection, (2) ecosystem maintenance and (3) aesthetic protection, yet where economic decisions drive forest management, it appears critical that an organization like the Sonoma County Forest Conservation Working Group exists.

In retrospect, the Working Group has learned a great deal in conducting outreach to hundreds of forest landowners in Sonoma County. We have also learned how to focus our efforts to achieve the goals at hand, moving forward on volunteer and lunchtime energy. It is important to note that, while many people have served on the ad-hoc committee that forms the Working Group, everyone does so with the enthusiasm, readiness-to-work and positive attitude that often accompanies the word ‘volunteer.’ Conducting outreach has been our goal, but the creation and perseverance of the Working Group has been our pleasure.

Conclusion

The Sonoma County Forest Conservation Working Group was formed in 2005 to address the needs of forest landowners throughout the county. The need for the Working Group was perceived due to the paucity of public lands in the county, and its extensive parcelization. Most of the oak woodland and the coniferous forestland are in private ownerships of 50 acres and less.

The Working Group was established to address threats to Sonoma County’s forestland. A number of factors, including population growth, regulatory requirements, estate issues and changes in the economy have created pressure to convert Sonoma’s forests to other uses.

The Working Group has used community workshops, seminars and neighborhood meetings and lunchtime ‘brownbag’ seminars to disseminate information and connect experts, landowners and resource managers to access technical and financial support. Four years of outreach experience and feedback has allowed the Working Group to look back and assess lessons learned.

Our experience demonstrated the need for learning both in the Working Group and the public. The Working Group found that the synergy of interested individuals and agencies could create a cost effective, well-equipped educational forum. With small grants and donations, we used our own energy and expertise on our mission. Landowners showed an interest in information on fire, ecosystems, wildlife and sustainability. The outreach also highlighted a lack of wildland management knowledge, a reluctance to spend money, and a no-management practice across the landscape that ultimately degrades forest health and increases fire danger and ecosystem fragmentation.

This experience demonstrated the importance of our mission, ‘Protecting Forests Across Landscapes and Through Generations.’ At this time, forests remain vulnerable, and significant education, policy, and public will are necessary to achieve long term success.

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California's Coast Redwood in New Zealand

Tom Gaman[1]

Abstract

New Zealanders are making a significant effort to develop their forest industry to benefit from rapid growth exhibited by *Sequoia sempervirens* on both the North Island and South Island. US and New Zealand forest products companies have established redwood plantations in the past decade, and have found that microclimate, site preparation, soil chemistry, fertilization and precise silvicultural practices are all of utmost importance. Farm foresters have established experimental redwood tree crops alongside *Pinus radiata* and many other western US and NZ native species. The NZ Farm Forestry Association Redwood Working Group has many experimental growth monitoring plots scattered around the country. Nursery producers are testing several California redwood clones that have shown exceptional promise. Researchers have been monitoring plantations of these and other clones here in California and in New Zealand. Genetic traits such as wood density, growth rates, and knot size are known to affect final product quality. Tree growers recognize the importance of investment in improved growing stock. While New Zealand plans to develop domestic and Australian markets for redwood products, growers hope to develop a redwood timber that will be competitive in international forest products markets.

Key words: redwood, New Zealand, plantation, tree improvement, farm forestry

Introduction

As New Zealand farm foresters and forest industry seek to diversify the country's plantation forest, they recognize California's coastal redwood as a key species for plantation forestry. Many New Zealand coastal districts have climatic conditions very similar to California and redwood grows rapidly there. With careful site selection, ongoing research, clonal propagation of improved provenances, and careful timing of silvicultural practices, New Zealanders have no doubt that they can produce a consistent product with desirable qualities. They hope that such timbers will favorably compete with second and third growth California-grown redwood in the global marketplace.

History of coast redwood in New Zealand

During the boom of the 19th and 20th centuries, enormous agricultural industries were developed in the isolated New Zealand economy. Sheep outnumbered people by 30 to 1. As lamb and wood exports declined, a plan in the 1970s projected that timber exports could provide 20 to 25 percent of foreign exchange earnings for the fragile New Zealand economy (O'Neill 1975) by early in the 21st century. *Pinus radiata*, known locally as simply "radiata", had been introduced in the 1860s and was widely planted on old field and volcanic sites by government, industry and farm foresters starting early in the 20th century. Also in the 1860s, coast redwood seed was imported from California. Today the Kaingaroa Forest, in the center of the North Island, is a 240,000 ha plantation almost entirely of radiata. Also, many of the early redwood

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plantations are magnificent groves today. Recognizing its potential and value New Zealand foresters in 1923 imported and planted a ton of redwood seed from California. Though wood quality turned out to be poor, and many trees were lost due to frost damage, weed competition, and inappropriate site selection, generations of redwood trees have been successfully grown at hundreds of New Zealand locations. Little of this redwood, however, was utilized for forest products and the tree was widely viewed as a novelty giant from afar.

As with much change in the world, visionary leadership is vital. Bill Libby, forest genetics professor emeritus of UC Berkeley, has been involved in New Zealand tree improvement since his 1969 sabbatical at the New Zealand Forest Research Institute in Rotorua. In a 1995 address to the New Zealand Farm Forestry Association (NZFFA) Libby urged the membership to establish commercial California redwood plantations. Foresters Wade Cornell, Rob Webster, forest nursery owner Robert Appleton and many others responded enthusiastically to Libby’s call for a new New Zealand redwood industry, and the NZFFA established its Sequoia Action Group. On this side of the Pacific, Libby and Kuser selected exemplary redwood trees from native stands in every California county, and established a clonal research trial plantation at University of California’s Russell Reservation in the East Bay hills.

New Zealand farm foresters and several US companies started forestry plantations on both the North and South Islands. Even the California forest industry became interested. In 2001 California’s Soper-Wheeler company established the New Zealand Redwood Company and purchased a 2500 ha tract, a former sheep station, at Conway River near Kaikoura on the east coast of the South Island. Later the company acquired the adjoining 2500 ha property. Today the New Zealand Redwood Company has 2000 ha of redwood planted on the South Island and 1200 ha on the North Island, the country’s largest redwood plantations. Port Blakely Tree Farms of Seattle created Blakely-Pacific Tree Farms and has embarked upon redwood planting on its lands in New Zealand. Though radiata remains by far the dominant species planted, scores of New Zealand tree farmers are also actively planting California redwood.

Research, tree improvement, clonal propagation, and product quality

With the goal to expand export timber production in the face of intense competition from Chile and many other southern countries, New Zealanders know they must produce a high quality product that will survive in discerning world marketplaces, particularly Australia, North America and Asia. The NZFFA Sequoia Action Group found that eight clones planted at 34 sites had highly variable performance ranging from 55 to 455 cm height in 4 years (Brown et. al. 2011), so it became clear that careful attention to growth rates, genetic selection, site selection and preparation, fertilization, plantation practice, pruning practices, branch size, wood density are all critical factors in selecting the proper clone to plant.

In the early years of the radiata industry most questions were answered by trial and error, but it was soon evident that continuing research and education programs are necessary in order to assure measured and successful development of the New Zealand plantation forest industry. The NZ Forest Service established the Forest

Research Institute at Rotorua in 1947 at the site of 1901 experimental plantations, including the Whakarewarewa Redwood Grove. Though the FRI facility was created primarily to support the radiata plantation industry, the fine plantations of redwood and Douglas-fir (*Pseudotsuga menziesii*), macrocarpa (*Cupressus macrocarpa*) and other species are of continuing interest. Today the research campus is operated for Crown Forest Research by Scion, a private organization with over 350 researchers and staff. Interestingly, the Whakarewarewa Redwood Grove is protected as a memorial grove and a site of historic trees.

The NZFFA has a membership of over 300 farm foresters from around the country, all with intense interest in forests and trees of the world. Almost 50 members have voluntarily established test plots on their farms throughout the country. The NZFFA Sequoia Action Group has field days, collects data, does provenance testing, and tracks the performance of selected clones. Farm foresters, the industry and researchers are working together on their common interest in redwood research.

Today virtually all redwood planting stock is clonally produced via tissue culture. The genetic material is obtained from the most outstanding provenances. Many new clones have been imported from California while other proven trees derived from the early historic plantings were also selected for development. Fifty new clones, derived from the 1923 seed import project, are tested each year by NZ Redwood Company. The California clones, originally selected from all the counties in California where redwood is native, are being tested for a variety of qualities on the NZFFA membership test plots. The original clones were selected either by complete chance, or because the parent trees seemed to possess desirable traits for New Zealand plantations. Meanwhile foresters have recognized that redwoods grown rapidly with very wide rings often do not produce the most desirable wood products. Clonal sources for tissue culture are being selected for moderate growth rates, smaller knot size and desirable heart color. Redwood clones are also selected to represent the average of natural range densities of 330 to 350 kg/m^3. Genetic testing is ongoing, but early results show that some of the California-sourced clones, with names such as R2 and R52, outperform their peers in growth and uniform small knot size[2], while some of the naturalized New Zealand clones are more adapted to varying New Zealand climatic conditions and growing sites[3]. Further wood density, durability, and structural tests are now underway to assure that the selected clones possess desirable market characteristics. Nicholas, Silcock and Webster of NZFFA, have calculated that investments in redwood plantations that can produce timber with these desirable traits show an internal rate of return of 9.3 percent net of land costs (Nicholas 2011, Chapter 9).

Redwood growers and silviculture

New Zealanders expect to produce redwood timbers from the selected genotypes that compete very handsomely with second and third growth redwood on the market in California. Trees are planted on frost-free or low frost prepared sites below 700 m elevation with medium to good quality soils in areas of moderate rainfall that are naturally sheltered from prevailing winds. High soil moisture capacity is important

[2] R. Appleton, Appleton Nurseries, Brightwater. 2010. Personal communication.
[3] R. Coker, New Zealand Redwood Company. 2011. Personal communication.

but fog drip is not essential. Sites require suitable redwood mycorrhizal inoculation to perform well. Weed control is critical for establishment. NZ Redwood Company originally used mechanical site preparation methods to clear weeds, mostly gorse, bracken and broom, on abandoned sheep paddocks. This method has been replaced with a combination of herbicide application and burning, which is cheaper and preserves soil structure (see footnote 3). Redwoods have performed adequately on clay soils on the North Island. As in California, New Zealand redwood plantations have been relatively pest free. Fungicide applications are in some locations necessary for establishment. There are enormous brushtail possum (Trichosurus vulpecula) populations in some areas, and they have damaged redwood leaders, so possum control is necessary. In wet areas drywood termites are sometimes problematic. Minor problems with leaf rollers and cankers have been reported. There has been some branch node dieback that may be attributable to soil boron deficiencies (see footnote 2). Minute boron applications on such sites seem to cure the problem. Robert Appleton has imported natural boron from Africa and South America and applied it in the Tasman Region using aerial dusting.

Plantings increase each year. Trees are planted at 500 to 600 stems per ha. After 9 years trees are generally about 8 to 9 m in height. Pruning would be necessary to produce the clear heart redwood prized by the market. However the species responds to moderate or heavy pruning by producing epicormic branches. Therefore pruning trials, which field test pruning of varying intensity and stages of development, are ongoing. Ideally trees would be pruned to about 4 m so that a clear log is produced in each tree.

Trees are planted with the intent of production of clear saw logs within 25 to 40 years. New Zealanders recognize that consistent quality is going to be vital when it comes time to develop an export market. If the clones selected today produce timber with traits such as durability, high specific density, good heart color, and small knot size, the market return should dramatically exceed that from radiata pine, and this group of creative researchers and land managers will have developed yet another valuable forest product and a source of foreign exchange for New Zealand.

Acknowledgments

Thanks to Robert Appleton, Russell Coker, Simon Rapley, and Bill Libby for sharing their insights, which contribute to this paper.

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Subdivide or Silviculture: Choices Facing Family Forest Owners in the Redwood Region[1]

William Stewart,[2] Shasta Ferranto,[3] Gary Nakamura,[2] Christy Getz,[2] Lynn Huntsinger,[4] and Maggi Kelly[2,4]

Abstract
Families or family businesses own nearly all of the private redwood forestland in California. Family forest owners have practiced both subdivision and silviculture for decades but the dominant theme for most family owners is environmental stewardship. Parcel size is more important than expressed values as a predictor of resource management activities. All landowners with more than 50 acres undertook high levels of resource stewardship regarding controlling invasive species, protecting water quality, improving fish and wildlife habitats, and removing individual trees to promote forest health. Timber harvesting was undertaken by 80 percent of the ownerships with 500 acres or more, but became progressively less common for smaller ownerships. Sustainable timber production is the most significant legal revenue-generating alternative to real estate for forest properties. Unlike most other resource management activities that are discretionary, timber harvesting in California can require permission from up to five state agencies and three federal agencies. High transaction costs limit involvement to forest owners with both large land holdings and strong skills in business management. It is not inconceivable that California's forest ownership pattern could become more like Washington State over time where small rather than large holdings represent the majority of family forestland.

Key words: family forests, land use change, sustainable forestry, timber harvesting

Introduction

Over the past 40 years, resource management of private lands in the redwood region has been a source of controversy as old growth redwood forests were logged and regenerated, wild and hatchery raised salmon populations fluctuated, and invasive species such as *Phytophthora ramorum* (cause of Sudden Oak Death) spread rapidly. A recurring theme in the public debate is whether timber harvesting is positively or negatively correlated with environmental stewardship activities. While the practices of the large industrial ownerships of forestland are well known due to their extensive permitting and planning documents, much less is known about resource management practiced by families with smaller ownerships. To increase understanding of the family owned forest and rangeland properties, a team of

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University of California Cooperative Extension and campus-based researchers surveyed California forest and rangeland owners in 2008. This paper uses a subsample of forest landowners (half from coastal forests and half from inland forests) to test hypotheses concerning resource management on family forests. We conclude with some hypotheses on the potential interaction of resource management and forest ownerships over the coming decades.

Methods

A mail questionnaire was sent to a sample of forest and rangeland owners on parcels greater than 3 acres in size from 10 representative counties in California. The counties included in the study were Humboldt, Sonoma, Mendocino, Shasta, Sierra, Plumas, El Dorado, Santa Barbara, San Diego, and Contra Costa.

Within each county, individual survey recipients were selected based on a stratified random sampling design. The sample was drawn from a statewide land parcel database created in 2003 by CalFire for the Forest and Range Assessment (CDF 2003). Parcel size was then sub-categorized into four groups: 3 to 9 acres, 10 to 49 acres, 50 to 499 acres, and greater than 500 acres. Equal samples were pulled from each size category. The resulting sample demographics have more small and medium size landowners compared to the proportion of total private forest and range land they own (Butler 2008). Due to the high standard errors on statewide estimates and the large variation in county-to-county forestland ownership patterns, definitive regional implications cannot be drawn from the statewide data analyses. The greater value comes from insights into factors such as ownership size, values, affluence, and local residency on resource management actions.

A statewide subsample of 200 forest owners was taken from the 670 forest and rangeland respondents for this analysis. Although only half of the forest owner respondents were from Sonoma, Mendocino and Humboldt counties, we used the total data set of statewide forest owners to provide greater statistical power. There was no statistically significant difference in the level of resource management activities between the coastal and inland forest areas.

Results and discussion

The survey results also included responses related to positions, values, actions, and additional information requests. We used the answers they provided on a 17-part question on their reasons for owning their land to create four unique groups based on environmental, family and community, and financial values (*table 1*). The results were analyzed using the K means cluster routine in the JMP software program (SAS Institute 2009). We gave the resulting four groups different names that match the results of Butler et al. (2007) in their analysis of a nationwide data set of family forest owners. It is important to note that our ‘forest investor’ group often own multiple parcels and had considerably larger ownerships than the other groups.

The names given to the different values-based clusters capture a dominant trait of each group. The percentages in the following tables refer to the number of answers that considered an attribute to be an important reason for forestland ownership.

Table 1—*Summary of characteristics of California family forest owners clusters based on environmental, family and community, and financial values.*

	Forest investor	Business family	Gardener family	In-heritor
Percent of total sample	16%	49%	22%	13%
Butler et al. (2007) typology	Supplemental income	Working the land	Woodland retreat	Ready to sell
Sample Size	30	91	41	25
> 50 acres	25	43	18	10
< 50 acres	5	42	23	14
No size reported	0	6	0	1
Median size in acres	1260	50	40	35
Reasons to own forestland				
Environmental values (n=5)	51%	71%	79%	20%
Family and community values (n=6)	16%	58%	56%	26%
Financial values (n=11)	50%	45%	23%	27%

The forest investor type captured all of the very large land holdings in the sample and was the only group where less than half the group did not live on the land or within 50 miles. Notwithstanding their moniker and large land holdings, their median annual income was considerably lower annual incomes than either of the two family groups. Annual income did not have a significant positive or negative influence of resource management actions across the whole sample. While the two family types have very similar values on most topics, the business family group had a stronger focus on the positive financial values related to owning forestland. The gardener family appears to most closely represent the archetypal city dweller that chose to move to a rural area. The inheritor group (the ‘ready to sell’ group in the Butler (2007) typology) generally had lower incomes, less interest in a children-friendly environment, and a greater interest in selling rather than managing the forestland. *Table 2* provides demographic information on the specific responses used to create the groups in the preceding table.

Table 2—*Positive responses to "reasons for owning land" questions from family forest owners in the University of California Landowner survey.*

UC Landowner Survey Family Forest Typology	Forest investor	Business family	Gardener family	In-heritor
Sample Size	30	91	41	25
DEMOGRAPHICS				
live on land or nearby	37%	59%	78%	72%
income > $100,000/year	37%	45%	51%	20%
retired	20%	36%	37%	28%
timber/range job	20%	19%	15%	8%
ENVIRONMENTAL				
protect environment	80%	68%	83%	4%
grow own food	0%	44%	59%	4%
preserve open space	67%	67%	85%	8%
live near natural beauty	50%	95%	98%	52%
recreational opportunities	57%	81%	68%	32%
simpler life	13%	84%	80%	44%
FAMILY AND COMMUNITY				
small community	20%	82%	76%	28%
good to raise children	13%	54%	61%	20%
escape city crime pollution	20%	87%	68%	36%
ex-city dweller	37%	32%	49%	36%
close to friends and family	0%	36%	15%	16%
connect to higher power	10%	31%	46%	0%
FINANCIAL				
add land	73%	46%	2%	20%
source of income	53%	40%	15%	16%
support local economy	43%	27%	17%	4%
family tradition/business	47%	52%	34%	16%
good financial investment	80%	82%	15%	36%
land appreciation	80%	84%	32%	52%
harvest timber	67%	38%	27%	32%
approached to sell	53%	36%	39%	36%
will sell	3%	12%	7%	32%
will give to children	47%	73%	68%	52%
will donate	0%	0%	0%	0%

How long have families owned their land?

The large number of willing buyers and willing sellers of family forestland in California as well as succession within families has led to a significant turnover in family forest ownership. *Figure 1* compares the length of time the forest property has been in the family for the different clusters. Both the business family and the gardener family had very high percentages of relatively recent land purchases. Across all groups, nearly 80 percent of the current owners became owners of their property after 1970.

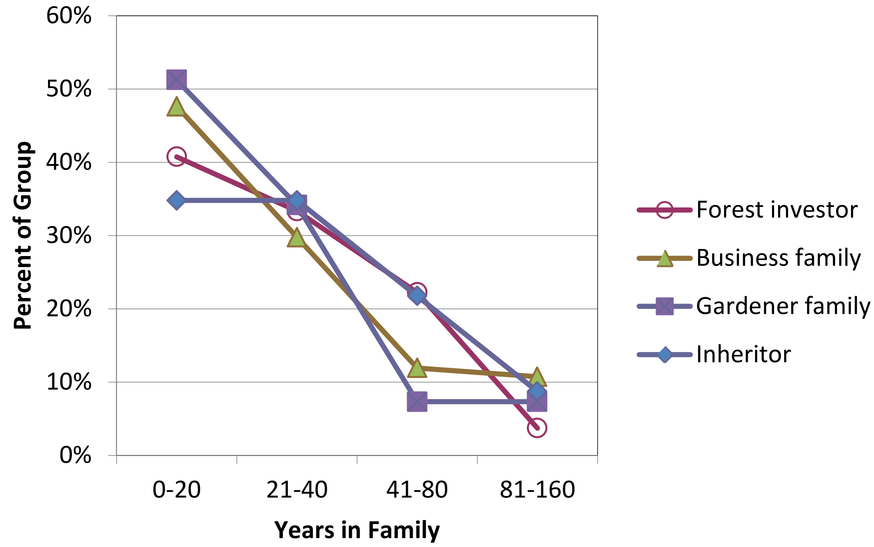

Figure 1—Length of time of family forest ownership.

Is ownership size or values the more important driver of resource management actions?

In addition to responses on positions (e.g., "I would like access to more land for hunting and fishing in my area"), and values (e.g., "I own land to help protect the environment"), we also collected responses on resource management activities undertaken by landowners. We used a small subset of those action responses that range from simple (remove exotics by pulling up weeds) to complicated (file a timber harvest plan) to compare the actions undertaken by different sub-samples of owners.

Figure 2 illustrates the resource management activities undertaken by the four value-based clusters for the medium and large ownerships (greater than 50 acres).

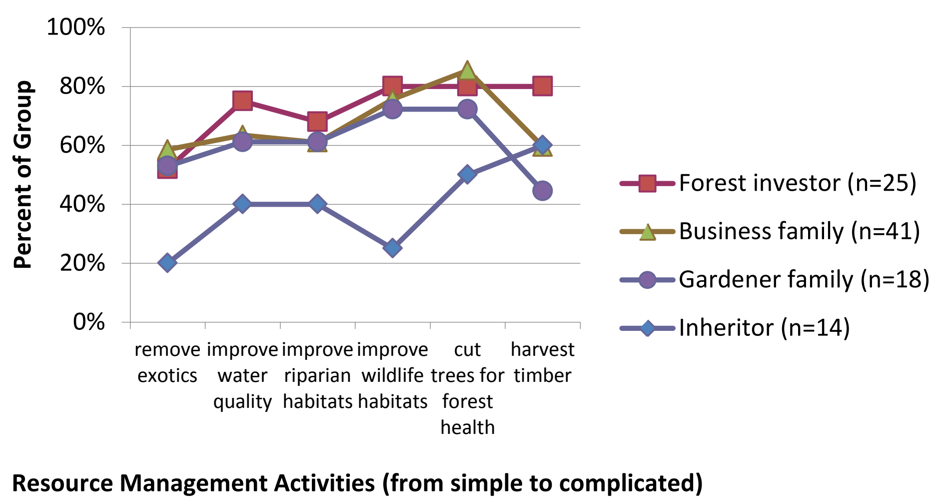

Figure 2—Resource management activities for ownerships larger than 50 acres by value group.

Among these ownerships, all the owners with strong environmental, family and community values also reported high levels of environmental stewardship activities. There was less consistency with regard to timber harvesting across the value-based clusters.

As shown in *fig. 3*, ownership size had more explanatory power across the range of resource management activities. The first five actions in both figures require expenditures of time and money by the landowner and can be considered as environmental 'gifts' since the benefits spill over across the landscape. It is noteworthy that few of the forest landowners reported being recipients of cost share awards or grants for these activities. They also did not have positive opinions regarding the quality of management advice they received from state and federal agencies associated with water quality and habitat issues. (Ferranto et al.[5]). Overall, it is apparent that forest owners with parcels larger than 50 acres are willing to invest considerable amounts of their own resources in environmental stewardship. Timber harvesting was practiced by 80 percent of owners with 500 acres or more, but dropped by half for each successively smaller ownership size class in our study.

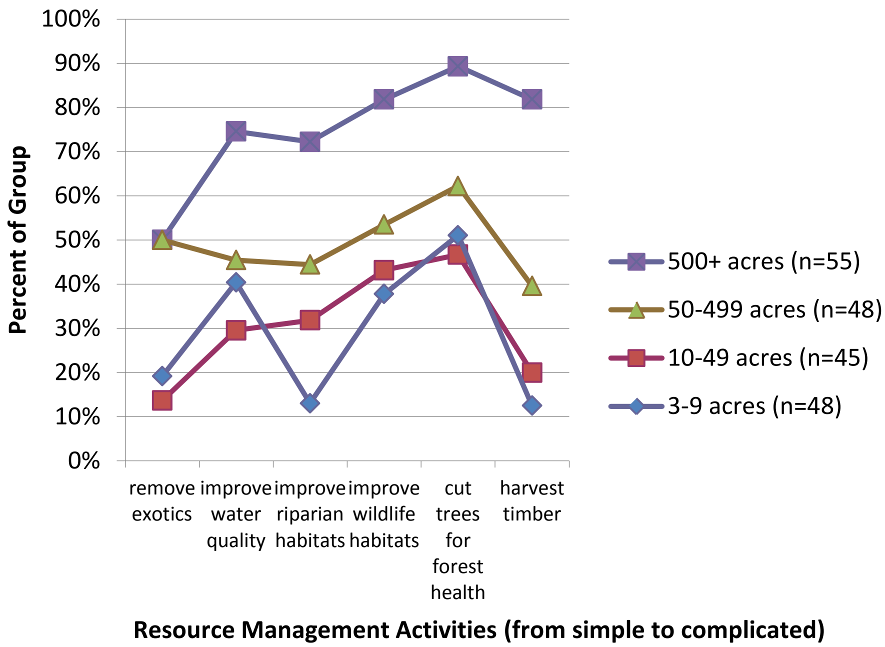

Figure 3—Resource management activities by ownership size.

For ownerships larger than 50 acres, there is a strong correlation between timber harvesting and four of the five other resource management activities. *Table 3* summarizes the probability of an owner with more than 50 acres undertaking an action as a function of whether they reported timber harvesting.

[5] Ferranto, S.; Huntsinger, L.; Stewart, W.; Getz, C.; Nakamura, G.; Kelly, M. 2011**. Consider the source: the impact of media and authority in outreach to California's forest and rangeland owners.** Unpublished.

Table 3—*Correlation between environmental stewardship activities and timber harvesting for forest ownerships larger than 50 acres (n=96).*

Environmental stewardship activity	Harvest timber	Do not harvest timber	p value
Remove exotics	.525	.457	.5245
Improve water quality	.738	.389	.0007
Improve riparian habitat	.721	.389	.0012
Improve wildlife habitat	.836	.486	.0003
Cut trees for forest health	.921	.500	.0001

When the two analytical approaches are compared, ownership size has a much stronger explanatory effect than the expressed values, especially for ownerships larger than 50 acres. Timber harvesting was less common than the other activities for most forestland owners. Unlike most other resource management activities that are discretionary, timber harvesting in California can require permission from up to five state agencies and three federal agencies. Done successfully, it can provide enough revenue to justify the ownership of land that has a high market value for alternative uses. The relatively low percentage of timber harvesting among medium (40 percent) and small (20 percent) sized ownerships suggests that there are a significant disincentives involved to this activity.

Does 'local' matter?

Since many of the environmental stewardship outcomes can only be appreciated by the owners if they visit the land regularly, we tested whether having a primary residence on the forestland or living within 50 miles of the forest property had an impact of resource management activities (*table 4*). For ownerships larger than 50 acres, the 'local' bonus varied from 19 percent to 49 percent. The flip side is that non-local forest owners are considerably less active in both environmental stewardship and timber harvesting activities.

Table 4—*Influence of local residency on resource management activities for ownerships larger than 50 acres.*

	Remove exotics	Improve water quality	Improve riparian habitats	Improve wildlife habitats	Cut trees for forest health	Harvest timber
Not Local (n=42)	45%	51%	50%	56%	67%	48%
Local (n=56)	54%	68%	70%	81%	87%	71%
Local/Non Local ratio	1.19	1.33	1.40	1.45	1.30	1.49

How many forest ownerships are at risk of getting smaller in the redwood region?

The preceding analysis illustrates very strong resource management values and actions among family forest owners in California with more than 50 acres of forestland. They place considerable value on the self-provision of environmental outcomes such as improved water quality, fish and wildlife habitats, and forest health or resiliency. Although their goals seem to match those of the regulatory agencies, there appears to be only limited cooperation. However, the general trend of increasing real estate prices compared to flat or declining commodity prices suggests that current land use and resource stewardship patterns could change. The following figures (*fig. 4*, *fig. 5*) show the distribution of forestland by parcel size (large ownerships are typically comprised of a number of medium sized parcels) for Sonoma, Mendocino and Humboldt counties. There are large acreages that potentially could be owned and managed as much smaller operational units without any need to request a parcel split.

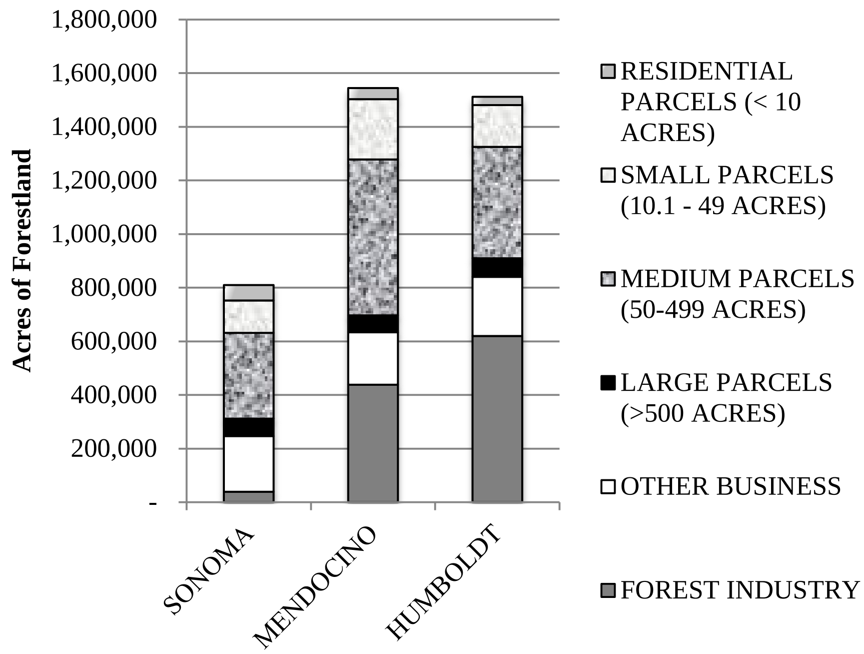

Figure 4—Area of forestland in family and business ownership in Sonoma, Mendocino, and Humboldt Counties.

While the rates of other environmental stewardship activities were positively correlated with timber harvesting, only timber harvesting generates revenue. For all forest landowners, there is a vibrant real estate market ready to purchase the ownerships ‘as is’ or to be further broken down into smaller units. California currently has a distribution of family forest ownerships very similar to Oregon where larger ownerships comprise most of the family forestland. However, the comparative economic advantages of non-timber valuation for small and medium sized ownerships could shift family forest ownership patterns towards those of Washington state.

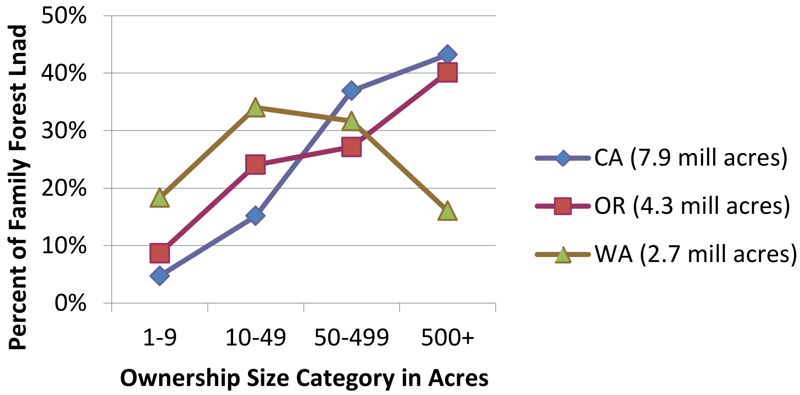

Figure 5—Distribution of family forestland in California, Oregon, and Washington (from Butler 2008).

Conclusion

Families own a substantial portion of the forests of the redwood region. Most of these parcels have changed ownership in the past forty years. Family forest owners have practiced both subdivision and silviculture for decades but the dominant theme for most family owners is environmental stewardship. The results of a 2008 survey of forest landowners documented very high levels of environmental stewardship activities that produce no revenue and provide benefits that extend beyond the property boundaries. Environmental stewardship actions were positively correlated with timber harvesting. All actions were more common for ownerships larger than 500 acres and declined as ownerships became smaller. It is smaller ownerships, however, that would be most affordable to the next wave of urban and suburban residents who would like to buy their own forest retreat. While the goals of stricter state environmental quality regulations seem to be closely aligned with the goals of most family forest owners in California, there does not appear to be much mutual recognition of common goals. The survey data also suggests that the high transaction costs limit involvement in timber harvesting to forest owners with large land holdings and strong skills in business management. Since sustainable timber production is the most significant legal alternative to real estate sales as a means for generating revenue from forest properties that are too large to be used as a single residential property, it is not inconceivable that California's forest ownership pattern would become more like Washington State over time where small rather than large holding represent the majority of family forest land.

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Is it Economical to Manage Jointly for Wood and Carbon Under the Climate Action Reserve Protocol?

Richard P. Thompson[1] and Steve R. Auten[1]

Abstract

To quantify the benefits and costs of modifying forest management for additional climate benefits, Cal Poly's Swanton Pacific demonstration forest was used to test the Climate Action Reserve's protocol and identify management strategies for both wood and carbon markets. Residing in the Southern Sub-district with its clearcutting restrictions, Swanton offers an ideal test site for landowners transitioning from even-aged to uneven-aged management. However, the increased minimum baseline condition for the region effectively precludes joint production of wood and additional carbon. The much lower baseline on the North Coast could allow for a joint production strategy that could meet some landowners' goals.

Key words: baseline condition, carbon registry, climate action reserve, CO_2e, forestry-offset

Introduction

The perception of an impending 'cap and trade' system for greenhouse gases in the United States has spurred the growth of tradable carbon-storage credits, or offsets, from a wide range of proven and new sequestration technologies. One of the most obvious is to add forests, known as forestry-offset projects, since trees are the only plants that retain woody tissue comprised of about 50 percent carbon by weight. Another reason for interest in forests for sequestering carbon is they are relatively inexpensive to establish and also provide a host of other environmental services. Nevertheless, there are a number of obstacles in the path toward full use of forestry-offset projects in meeting national and international policy goals on reducing atmospheric greenhouse gas (GHG) emissions.

For carbon-storing forestry-offset projects to be viable, the net revenue from carbon services must be competitive with expected cash flow for the forest landowner. This can be fairly low for forest landowners who prefer a reduced or zero harvest rate but will be much higher for forest landowners managing their land for maximum sustained harvest levels. The costs of additional inventory and monitoring actions required by carbon exchange protocols loom as a major issue in competitiveness (Mader 2007, Rudell et al 2007). Assuming that additional carbon inventories can be counted as a tradable benefit, Simpson (2008) reports that minor modifications for carbon storage of loblolly plantation management on a 30-year rotation can increase rates-of-return from 7.0 percent to 9.2 percent (NPVs from \$364/ac to \$520/ac) with just \$5/tonne CO_2e prices, if all three CCX forestry-offset

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protocols are used. He also noted that carbon monitoring and trading costs were 25 percent of the carbon revenue and that thinnings have a negative effect on carbon revenues. The CCX offset market closed in 2010 and is quite different than the CAR protocols (CAR 2010) that are better known in California. The goal of our research is to provide small to medium-sized forest landowners with information on carbon credits available under uneven-aged management, the costs of offering carbon credits on the market, and the likelihood of receiving sufficient revenues that justify the investment.

Forestry-offset projects – the basics

Although forestry-offset projects have been voluntarily used worldwide to comply with Kyoto, they remain controversial due to the transitory, albeit long-term, nature of trees. The CCX, and the more recent CAR protocols, use verifiable standards that forestry-offset projects must satisfy in order to “sell” carbon credits. The interpretation of the following four accounting issues are very important in how many offset credits are considered marketable for any individual project.

1. Permanence – sequestered carbon must be long-term to be of value in mitigating climate change.
2. Additionality – forest management must result in additional carbon storage in the forest and where the harvested products are used that would not have occurred with business-as-usual (“Common Practice”), estimated over a 100-year time horizon.
3. Baseline – carbon that would have been stored in lieu of new activities to store additional carbon, aka, *business-as-usual.*
4. Leakage –increased carbon storage can induce reduced carbon storage due to higher local timber prices, those leakages must be subtracted from the measured local benefits.

There are a growing number of carbon registries for forestry and other offset projects.[2] The CAR protocol is one of the newest (Version 2.1 in 2007, Version 3.2 in 2010); this study used Version 3.2.

Climate Action Reserve offset projects

The Climate Action Reserve (CAR or the “Reserve”), the successor to the California Climate Action Registry, is a non-profit organization created by the State of California in 2001 to “promote and protect businesses early actions” designed to reduce or offset their GHG emissions (CAR 2010). The Reserve is financed by GHG emitters interested in creating tradable offset credits that will presumably be less expensive than other forms of GHG reductions. It includes a number of detailed and very specific protocols, setting high standards for those applying to sell carbon storage credits. Climate Reserve Tonnes (CRTs or “carrots”) can be sold to parties that want to offset current emissions or resell these offsets when there is a state or national system of required emission reductions.

[2] for example, American Carbon Registry (Environmental Resources Trust. Winrock International 1996), Verified Carbon Registry System (Verified Carbon Standard Association, Version 2, October 2006); Markit Environmental Registry (formerly TZ1. 2009).

Figure 1 illustrates the distribution of the CAR's offset projects. "Listed" projects are those that have been submitted to the Reserve but have not been verified yet by a Reserve-approved auditor. Once verified, the project is then "Registered" with a specific number of CRTs per year (known as "vintages"). The most common Reserve projects are offsets for landfill gases, nearly half of which have been registered. There are three types of forestry offset projects: (1) Reforestation, (2) Improved, formerly Conservation-Based, Forest Management (IFM, CFM), and (3) Avoided Conversion. Forestry-offset projects are the third largest in number among all projects. Although IFM/CFM are the most common among forestry-offset projects, only four have been registered: Lompico Forest Carbon Project in Santa Cruz with 11,708 CRTs in 2010, Garcia River Forest in 2004 with 715,268 CRTs from five vintages, The van Eck Forest with 274,403 CRTs from four vintages (2006 to 2009), and Big River/Salmon Creek Forests in 2007 with 736,517 CRTs from three vintages. Thus far, all registered forestry-offset projects are in the coast redwood region of California, and all three CFM registered projects are in the North Coast redwood region. However, none of these projects involve managing for carbon without significant reduction in (or elimination in Lompico's case) timber production.

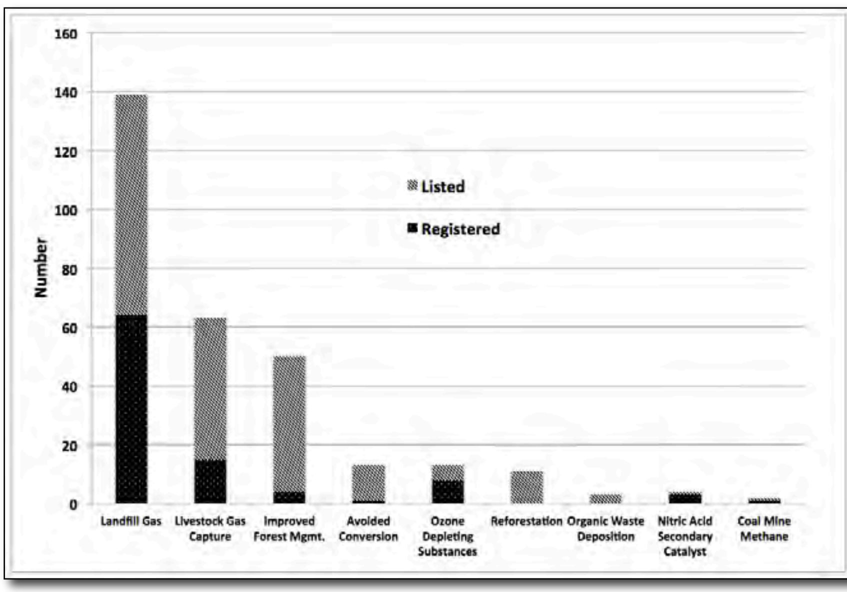

Figure 1—Climate action reserve offset projects.

Climate Action Reserve forestry-offset protocol

CAR's forest-offset protocol has gone through five iterations with the first announced on June 13, 2005 and the current version (3.2) approved on August 31, 2010. Listed and Registered projects can differ in requirements and standards depending upon the protocol version pertinent at the time of listing. Calculating CRTs under the IFM protocol is an involved and costly process. The process described hereafter greatly simplifies what is a much more detailed calculation.

One starts with an inventory of the subject forest property; this sets the year of the CRT vintage. Precision has a significant effect on available CRTs, where a sampling error (SE) greater than 20 percent eliminates any carbon credits, from 5.1 to 20 percent reduced CRTs by 5 percent, and 5 percent SE or less has no deduction. Inventories should include all standing timber with volumes estimated using CAR-approved cubic foot volume equations for tree boles. Next total above-ground metric biomass is calculated where bark and live crown biomass is added based on species and diameter. Carbon biomass is 50 percent of total above-ground bone-dry biomass which is then converted to metric tonnes (0.001 tonnes/kg). Under the current protocol, carbon weight must be converted to CO_2 equivalent (CO_2e) to represent the atmospheric effect of CO_2 emissions on climate (carbon tonnes x 3.67). Thus the Actual Above-ground, Live Carbon Stock (AC) in year y is calculated as follows:

$$AC\ (CO_2e) = \Sigma\ (cfvol_i)(kg/cf_i)(0.5\ C)(.001\ C\ tonnes/kg)(3.67\ CO_2e/tonne),$$
where i = species

The CAR protocols are based on creating offsets for owners who hold higher than ‘average’ inventories (common practice). While half of all parcels are going to be above the regional average in any case, the 100 year contract would require owners to maintain a high inventory for the following century.

Once AC is calculated, one then determines the minimum baseline (MBL). First, the carbon stock based on the Common Practice (CP) for the Assessment Area of the subject project is obtained from Appendix F of the protocol. The Common Practice for Version 3.2 was estimated using Forest Inventory and Analysis (FIA) plots on private forestlands. The equation for MBL depends on whether the initial carbon stock (actual above-ground carbon stock, AC) is greater or less than CP. The following MBL equation is used when AC > CP.

$$MBL = max\ (CP, min(AC, CP+AC-WCS),$$
where WCS is the weighted average of above-ground AC for all landholdings in addition to the subject property. If there are none, then AC = WCS, resulting in MBL = CP.

At this stage, it is important to estimate whether there exists any additionality as a result of the Initial Carbon Stock (ICS) being greater than the MBL If not, the remaining calculations are unnecessary since there is no additionality available in the ICS. Further, additionality per CAR protocol is only derived from existing, not predicted (from growth & yield modeling), carbon stocks.

If additionality appears to exist, one proceeds to model the baseline carbon stock (BC) for all required (in other words, live, above-ground timber) and optional carbon pools (e.g., shrubs, soil) through a series of growth and yield cycles over 100 years. This is needed to demonstrate that carbon credits (CRTs) can be maintained based on reliable predictions. The model must reflect all legal and financial constraints. The average of the growth and yield results is used for the BC calculation and must not fall below MBL.

If AC is predicted to still be greater than BC, one then adds below-ground biomass and standing dead biomass estimates to AC, along with other optional carbon pools. Interestingly, the equation for below-ground biomass (BBD) is an exponential functional of above-ground biomass (ABD) regardless of species [BBD = exp(-0.7747 + 0.8836 x ln(ABD))]. Equation 6.1 provides the calculation that leads to CRTs, or Quantified Reductions (QR) for each year’s (y) carbon stock:

$QR_y - [(\Delta \square C_{onnsite} - \Delta BC_{onnsite}) + (AC_{wood\ products,\ y} - BC_{wood\ products,\ y}) \times 80$ percent $+ SE_y] + N_{y-1}$

where,

$\Delta AC = (AC_{onnsite,\ y})(1\text{-confidence deduction}_y) - AC_{onnsite,\ y-1})(1\text{-confidence deduction}_{y-1})$, if y is the first year, then $(AC_{onnsite,\ y-1}) = 0$.

$\Delta BC = (BC_{onnsite,\ y})(1\text{-confidence deduction}_y) - BC_{onnsite,\ y-1})(1\text{-confidence deduction}_{y-1})$, if y is the first year, then $(BC_{onnsite,\ y-1}) = 0$.

$AC_{wood\ products,\ y}$ = actual carbon in wood products produced in year y projected to remain stored for at least 100 years.

$BC_{wood\ products,\ y}$ = average annual baseline carbon in wood products that would have remained stored for at least 100 years.

SE = Secondary Effect GHG emissions caused by the project activity in year y.

N = Any negative carryover from the prior year (see footnote 9, p. 38 Version 3.2).

The first two terms are referred to as the Primary Effect. Wood products can either be processed by mills, with adjustments for mill efficiency, or "stored" in landfills for some types of projects. The 80 percent adjustment is intended to model market response to reduced wood product production where every tonne diverted to carbon storage will be compensated with an increase in harvesting 0.2 tones on other lands, i.e., leakage. While the 20 percent leakage may be a realistic estimate of local substitution, leakage at the timber market scale for North America has been estimated at over 80 percent (Murray et al. 2004). Wood residues that are harvested but used to generate energy are also ignored in the CAR accounting system. Over a 100 year lifetime, wood used for energy can constitute over half of initially harvested volume (Smith 2009).

Once QR is calculated, all remaining deductions (mainly risk of loss/reversal) are formula-driven. These deductions address the financial, managerial, social and natural disturbance risks that could result in release of CRTs. Finally, CRTs are reduced by the calculated risk percentage, and contributed to a Buffer Pool. However, as with leakage, little or no scientific support or rationale is offered for these constants. Table 6.4 of the Protocol provides a step-by-step procedure for these calculations.

The Lompico Project – an example

To exemplify the Version 3.2 protocol, the Lompico project was chosen for two reasons. First, it's the only CAR forestry-offset project adhering to the current protocol; the other earlier registered forestry projects were not required to provide as much information. Second, Lompico is in the Southern Sub-district of the Coast district where this study's project is located. The only drawback to Lompico for our purposes is their management objectives exclude timber harvesting.

Lompico, located about 10 miles north of Santa Cruz, was purchased by the Sempervirens Fund from Redwood Empire Co. as part of the Fund's mission to preserve redwood forestland. Carbon storage from precluding timber harvests on 285 ac of Timber Production Zone (TPZ) parcels within the 425 ac ownership was the basis for their application for IFM carbon credits through the Reserve.

First, Common Practice (CP) was determined to be 280 CO_2e tonnes/ac according to the protocol in order to establish the minimum baseline (MBL). Version 3.2 increased CP to 280 CO_2e tonnes/ac from the earlier version's 224. The new CP is quite high for the average of managed timberlands in the Santa Cruz and San Mateo Assessment Area, an issue that will be addressed again later in the paper. Second, the Fund's initial carbon stock in 2007 was estimated at 322 CO_2e tonnes/ac (with a 5.6 percent SE). Next, growth and yield modeling was performed based on thinning 30 percent of the timber greater than 18" DBH, giving a baseline condition (BC) that averaged slightly above MBL. Subtracting BC from the ICS resulted in 42 CO_2e tonnes/ac (11,970 total) of additionality for the 2007 vintage (*fig. 2*). Further additionality is derived from timber growth from the inventory date to the CAR application - 2008 and 2009 vintages of about 4 CO_2e tonnes/ac each year.

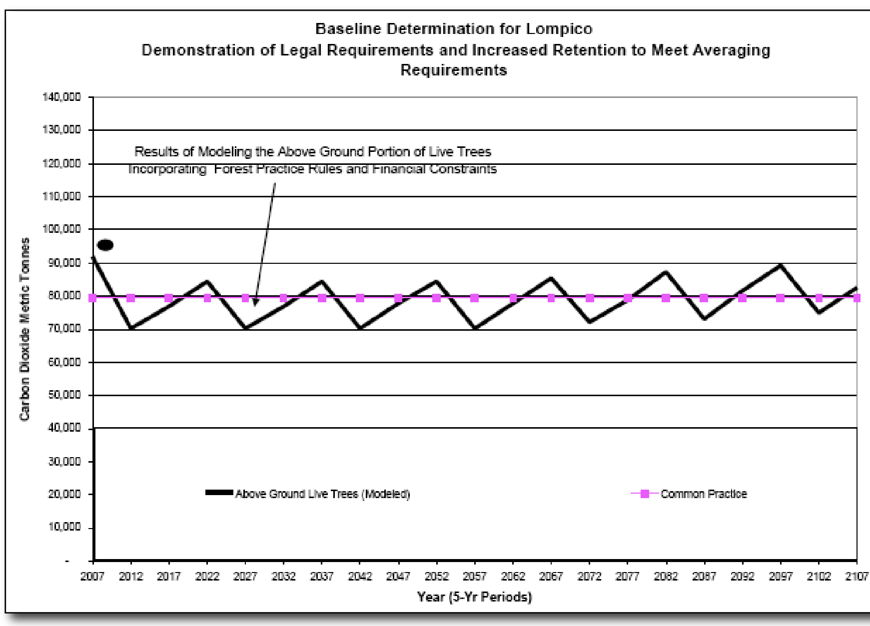

Figure 2—Lompico modeled baseline condition for live, above-ground carbon stocks.

What remains to determine CRTs after the calculation of the carbon stock is much smaller in magnitude but more detailed in procedure. After calculating the below-ground carbon stock, the second part of the Primary Effect is to calculate net wood product carbon stocks - planned harvests less baseline harvests. This is followed by estimates that are essentially deductions required to ensure permanency and avoid risks of human and naturally caused losses. In Lompico's case the net harvest carbon storage was negative because of the Fund's non-harvest philosophy. In the end, Sempervirens Fund was successful in selling 11,798 CRTs.

Swanton Pacific Demonstration Forest – Valencia Unit

Cal Poly owns two properties in Santa Cruz County for education and research – Swanton Pacific Ranch (3200 ac) near Davenport, and Valencia (620 ac) near Aptos. The Valencia unit is almost completely forested with the following characteristics:

- 480 ac of redwood-fir production lands.
- 60 percent redwood, 10 percent Douglas-fir, 30 percent hardwoods.
- Average high Site Class III.
- Areas with two to three age-classes from light thinnings in the 60s and 70s.
- Well-stocked, over 40 mbf/ac in conifer volume, 260 BA.
- FSC certified, approved NTMP.

Both forested properties are managed for educational and research purposes. Southern Sub-district Rules prohibit even-aged management. Valencia is managed for an uneven-aged structure using Guilden's BDq model is to guide silvicultural decisions (Guilden 1991). *Fig. 3* displays the current Valencia structure for the average stand compared to the BDq guide curve, with the following residual stand parameters: BA = 175 ft^2/ac, max DBH = 34", DeLiqourt's q = 1.15.

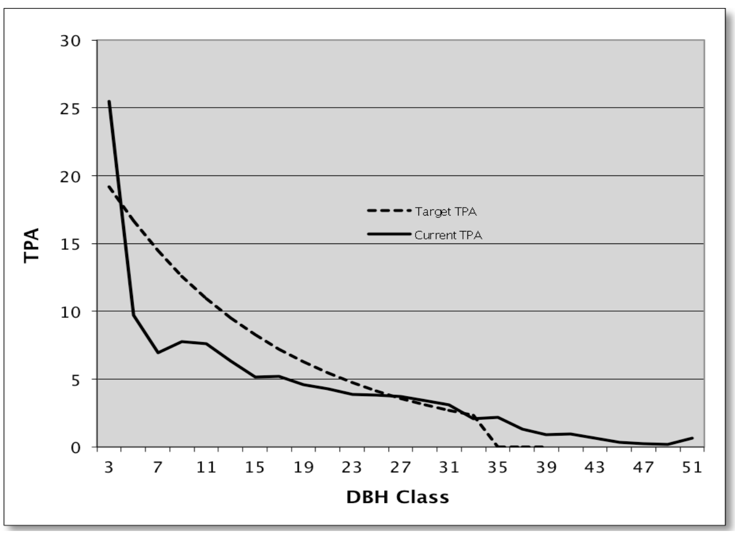

Figure 3—Current average Valencia stand structure compared to the target stand structure.

Although Valencia has a CFI system in-place and was scheduled for re-inventory, the Lockheed fire at Swanton delayed it making it necessary to install a separate inventory for this project. A VPS inventory was conducted in August 2010 according to the CAR protocol at a cost of $5000, resulting in the volume estimate stated above. Inventory costs were somewhat less than normal since DBH and height data were available from past CFI inventories. The sampling error was only 1.8 percent, thus no CAR deduction is necessary. The live, above-ground carbon stock was calculated to be 376 metric tonnes/ac of CO_2e.

As shown in the Lompico Carbon Project, the Santa Cruz Assessment Area's CP is 280 CO_2e tonnes/ac (MBL). Since Valencia's ICS > MBL, it may appear that additionality exists, but the average baseline carbon stock projected over 100 years must also exceed MBL. No reasonable harvest level could be found such that the average baseline carbon stock exceeded MBL. This is not economical since current stumpage prices of around \$500/thousand board feet (BOE 2011) are far above the \$10/metric ton of CO2 (roughly equivalent to a stumpage price of \$75/mbf).

North Coast uneven-aged management for wood and carbon

The initial concept for a carbon project on Cal Poly's forest properties in Santa Cruz County presumed revenue possibilities existed based on earlier Protocol versions. However, the current version raised CP so high as to preclude joint timber and carbon management at Valencia. A possible explanation for this high value will be discussed at the end of this paper. Nevertheless, an opportunity for combining wood and carbon production exists on the North Coast where CP on equivalent "low" site quality is drastically lower - 165 CO_2e tonnes/ac (even North Coast high site is only 188 tonnes). Therefore, we propose to describe Valencia's carbon-offset potential as if it were in the heart of the coast redwood region, where Sub-district rules do not apply.

Figure 4 illustrates two sustained yield analyses: (1) baseline condition (BC) growth and harvests that are sustainable where the average carbon stock is at or above Common Practice (MBL), and (2) the average actual carbon (AC) resulting growth and yield modeling using residual stand BDq parameters stated above. The BC analysis reduced growing stock to about MBL, required a longer 15-year growing cycle. The average carbon stock was about 165 CO_2e tonnes/ac, or about 20 tonnes/ac greater than the Sub-district's MBL.

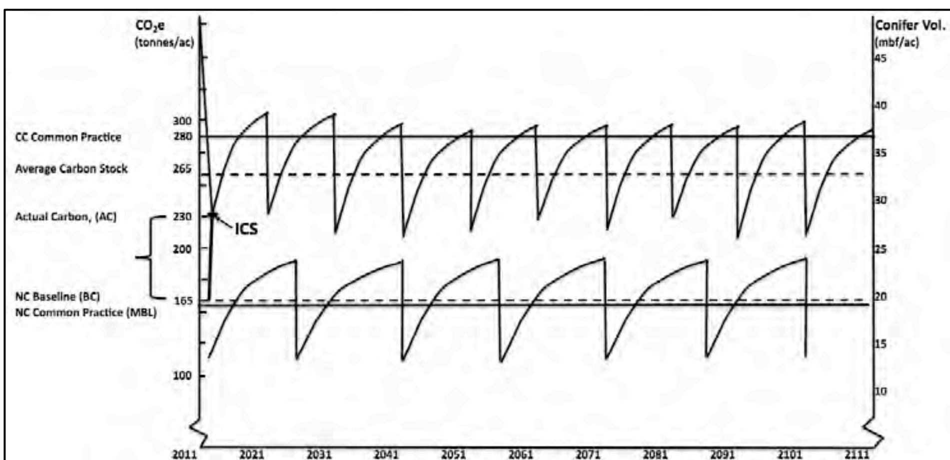

Figure 4—Sustained yield analysis for both baseline condition and uneven-aged timber management objectives.

Sustained yield analysis for the actual carbon stock started with a thinning that somewhat aggressively reduced live, above-ground CO_2e from 376 to about 230 tonnes/ac by harvesting nearly all trees above max DBH and drastically reducing the hardwood component. This reduced standing volume from about 43 mbf/ac total to about 28 mbf/ac. Afterward, a 10-year thinning cycle varies roughly between a residual volume of 28 mbf/ac and growing to 38 mbf/ac. The result is a growth rate of approximately 1000 bf/ac/yr, or 3.5 percent/yr. At a stumpage price of at least

$500/mbf, that is an annualized revenue of $240,000. It would not make economic sense to substitute this one-time carbon revenue for this wood product income stream, even at conceivably higher carbon prices.

If a forestland owner in the North Coast region were to manage a property like Valencia on an uneven-aged management regime as prescribed, there would be an Initial Carbon Stock (ICS) well-above the North Coast MBL, providing about 65 tonnes/ac of additional live, above-ground CO_2e carbon stock. Further additionality from below-ground carbon adds about another 5 tonnes/ac. (Note that residual carbon stocks drop slightly below 230 requiring either adjustments to the sustained yield analysis or slightly reducing ICS.)

The remaining term in the Primary Effect is the solidwood product carbon stock. Equation 6.1 (CAR 2010) gives credit for actual carbon sold from harvests after project initiation, plus the average from predicted harvests, minus the average from baseline predicted harvests. The vast majority of the timber would be sold with the residues treated as if landfilled. After adjusting for mill efficiency and market response, solidwood product carbon stock could add roughly another 35 CO_2e tonnes/ac, mostly coming from the initial harvest of about 17 mbf/ac.

Given manuscript size limits, the remaining secondary effects are not described in detail. In the end, at least 80 CRT tonnes/ac. should be verifiable, or 38,000 tonnes toal. At recently heard OTC CO_2e prices of $10/tonne, or a one-time revenue of $380,000 to compensate for the unharvested volume that must be protected from all disturbances for 100 years. Even allowing for estimation error, such revenue certainly justifies the cost of application and additional inventories, if necessary. However, one must keep in mind that annual reports, periodic inventories and audits are required to certify that the live tree CRTs sold remain on site.

Final thoughts

Joint management for carbon and timber under the current CAR protocol can be profitable under specific conditions. There appears to be profitable opportunities for joint management in regions where the “common practice” reflects stocking levels from a balanced range of stand ages. Such is the case in the North Coast redwood region; not so in the Sub-district region where heavy cutting 80 to 100 years ago spurred regulations requiring higher average stocking, the basis for the CAR’s regional baseline. It would not be economical for Sub-district forestland owners to substitute carbon for wood production goals at current or conceivable product prices.

Although Sub-district common practice carbon stocks are expected to be higher than on the North Coast, the magnitude of the difference seems unjustified, especially when comparing CO_2e estimates with equivalent volume and basal area averages in Appendix F. One explanation may be due to stratifying FIA plots only by private land. However, in Santa Cruz and San Mateo counties, zoning regulations further limit lands where timber management is permitted.

This study did not test whether even-aged management offers a joint production opportunity but the large price differential between redwood stumpage and generic carbon prices suggests disadvantages. However, for forestland owners who are considering a switch to uneven-aged management may be further encouraged to do so with these results. This result should be transferrable to other forest types where lower growth rates are offset by lower, more carbon substitutable, wood prices.

Finally, one wonders what effect contracting to sell CRTs would have on property values. Since these contracts "go with the land" (somewhat like a conservation easement) limiting management options, property values would be reduced, though partially compensated from carbon revenues. Further value reductions are likely when a prospective buyer discovers that they are committing to annual reporting, inventories and audits.

Acknowledgments

The authors wish to thank Dr. William Stewart, UC Cooperative Extension Specialist, Center for Forestry, for his advice and contribution to this paper. Any errors are the sole responsibility of the authors.

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