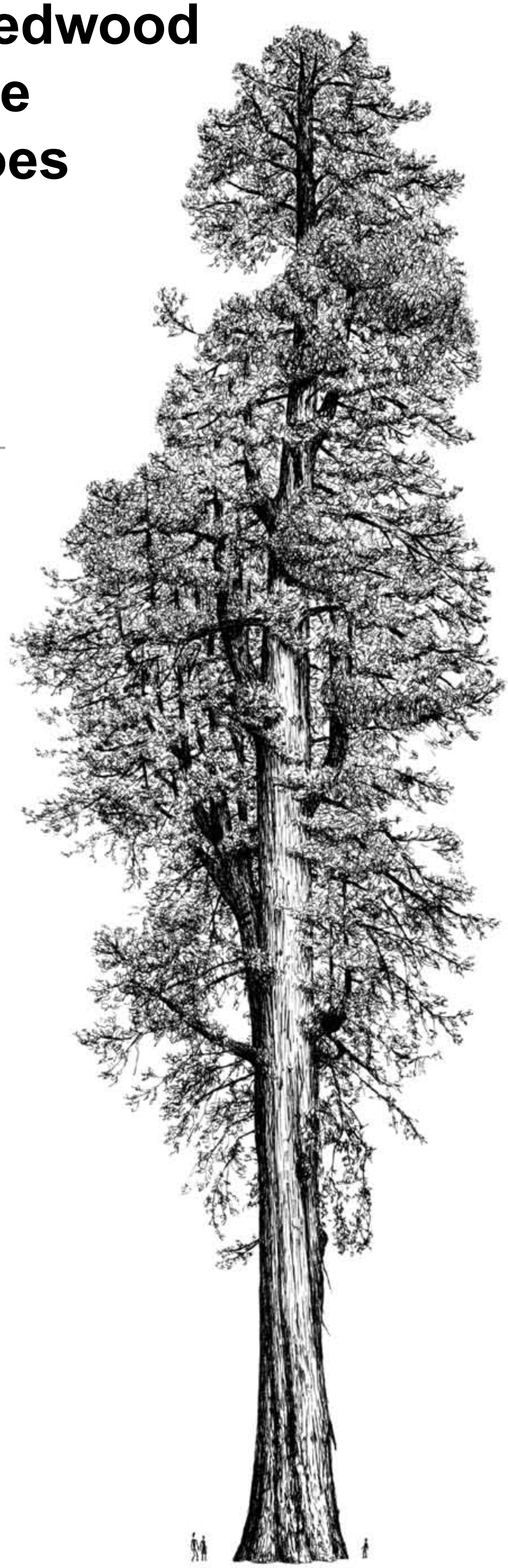
Proceedings of the Redwood Region Forest Science Symposium: What Does the Future Hold?

March 15–17, 2004



United States
Department of
Agriculture

Forest Service

Pacific Southwest
Research Station

General Technical
Report
PSW-GTR-194
July 2007

Technical Editors
Richard B. Standiford is Associate Vice President for the UC Division of Agriculture and Natural Resources, and Cooperative Extension Specialist, University of California, Berkeley, CA 94720; **Gregory A. Giusti** is Forest Advisor, University of California Cooperative Extension, Lake and Mendocino Counties, Lakeport, CA 95453; .**Yana Valachovic** is Forest Advisor, University of California Cooperative Extension, Humboldt and Del Norte Counties, Eureka, CA 95503; **William J. Zielinski** is a wildlife biologist, U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, 1700 Bayview Drive, Arcata, CA 95521; and **Michael J. Furniss** is a hydrologist, U.S. Department of Agriculture, Forest Service, Pacific Northwest and Pacific Southwest Research Stations at Redwood Science Laboratory, Arcata CA 95521.

Cover and chapter divider illustration by Robert Van Pelt from "Forest Giants of the Pacific Coast." The tree illustrated is within Prairie Creek Redwoods State Park. For more on this superlative tree, see "A redwood tree whose crown may be the most complex on Earth. SC Sillett, R Van Pelt - L'Arbre, 2000" (http://www.humboldt.edu/~storage/pdfmill/201/sillett.pdf)

Proceedings of the Redwood Region Forest Science Symposium: What Does the Future Hold?

Technical Editors
Richard B. Standiford, Gregory A. Giusti, Yana Valachovic,
William J. Zielinski, and Michael J. Furniss

March 15–17, 2004

Rohnert Park California

U.S. Department of Agriculture

Forest Service

Pacific Southwest Research Station

Albany, California

General Technical Report PSW-GTR-194

July 2007

Acknowledgments

Program Committee

Rick Standiford, UC Center for Forestry, (510) 643-5428, e-mail: standifo@nature.berkeley.edu
Greg Giusti, UC Cooperative Extension, (707) 463-4495, e-mail: gagiusti@ucdavis.edu
Yana Valachovic, UC Cooperative Extension, (707) 445-7351, e-mail: yvala@ucdavis.edu
Bill Zielinski, PSW Research Station, Arcata, (707) 825-2959, e-mail: bzielinski@fs.fed.us
Joni Rippee, UC Center for Forestry, (510) 642-0095, e-mal: rippee@nature.berkeley.edu

Coordinated by

University of California Center for Forestry, Berkeley Campus
Division of Agriculture and Natural Resources, UC Office of the President
College of Natural Resources, Berkeley Campus

Co-sponsors

California Department of Forestry and Fire Protection
Hawthorne Timber/Campbell Timberland Management
Mendocino Redwood Company
Pacific Lumber Company
Save-the-Redwoods League
Simpson Resource Company
The Nature Conservancy
USDA Forest Service Pacific Southwest Research Station

Steering Committee

Nolan Colgrove, Hoopa Tribe
Neal Ewald, Simpson Resource Co.
Greg Giusti (co-chair), University of California Cooperative Extension
Richard Harris, University of California, Berkeley
Ruskin Hartley, Save-the-Redwoods League
Eric Huff, Big Creek Lumber
Mike Jani, Mendocino Redwood Company, LLC
Wendy Millet, The Nature Conservancy
Doug Piirto, California Polytechnic State University
Richard B. Standiford (co-chair), University of California Center for Forestry
Bill Stewart, California Department of Forestry, Fire and Resource Assessment Program
John Stuart, Humboldt State University
Brad Valentine, California Department of Fish and Game
Yana Valachovic (co-chair), University of California Cooperative Extension
Bill Zielinski (co-chair), Pacific Southwest Research Station

Proceedings of the Redwood Region Forest Science Symposium: What Does the Future Hold?

Abstract

Standiford, Richard B.; Giusti, Gregory A.; Valachovic, Yana; Zielinski, William J.; Furniss, Michael J., technical editors. 2007. Proceedings of the redwood region forest science symposium: What does the future hold? Gen. Tech. Rep. PSW-GTR-194. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture. 553 p.

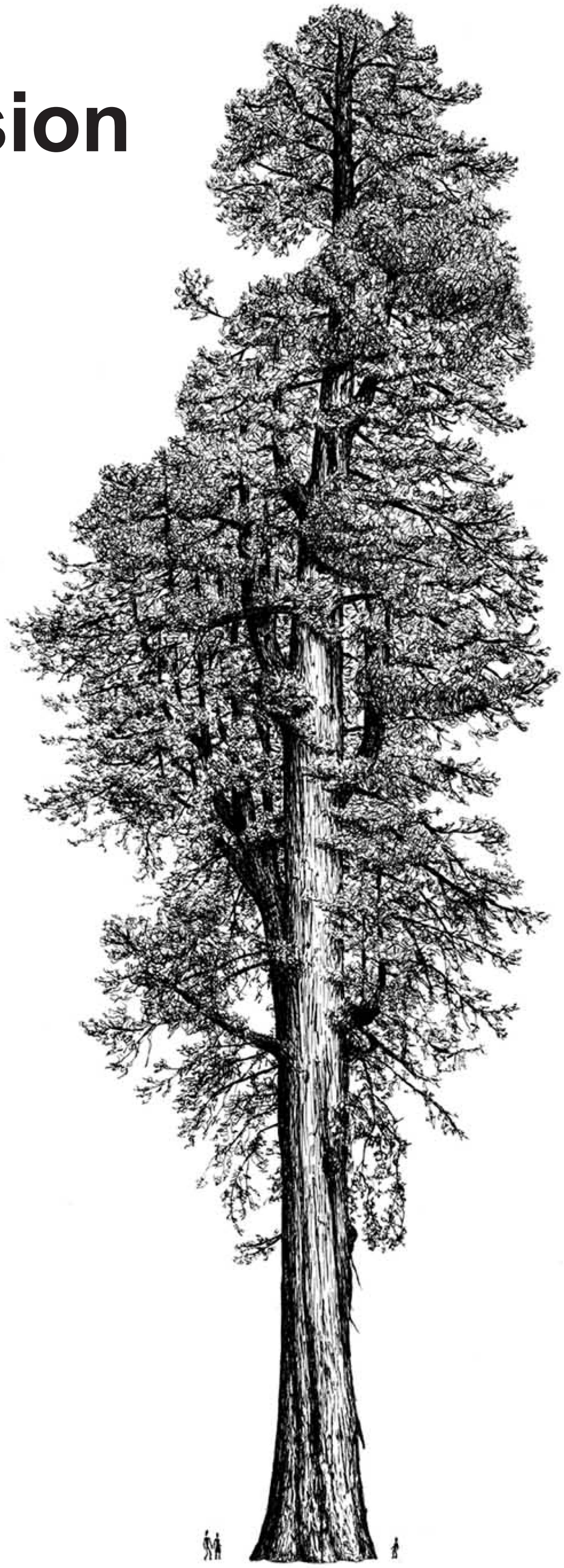
Policies and strategies that guide use and management of lands in the coastal ecoregion are dependent on objective scientific information. In recent years, attention to this region has increased. Correspondingly, there has been much new information collected. Efforts such as the Caspar Creek Watershed Conference and the Scientific Basis for the Prediction of Cumulative Watershed Effects illustrate both the great interest and effort that is devoted to collecting and using scientific information to support resource and land management in this region. Each year the array of decisions that affects lands and natural resources in the redwood region carry more weight; evidence the recent interest in TMDLs, watershed assessment, and fish and wildlife recovery efforts. How do we, therefore, promote the development and communication of scientific findings to inform management and policy decisions?

No single meeting or institution is capable of providing thorough coverage of current scientific findings and insights. It is the intent of the organizing committee to provide a sampling of current scientific work, to enable access to more detail and other sources of information, and to put these findings into a context where such information can be synthesized and interpreted for applications in land and resource management.

This symposium is intended to promote the dissemination of scientific evidence to managers, policymakers, other scientists and interested public, and, in turn, to inform policy decisions. Thus, the presentations will range from the discussion of recently gathered scientific knowledge to the integration of that knowledge into planning and management processes and tools. We support the many other efforts intended to achieve these and similar goals and acknowledge the need to coordinate all such efforts.

Contents

General Session



Opening Address[1]

Katherine Anderton[2]

We're all here because we all believe there are benefits in developing and sharing our understanding of the redwood forest. Why we want to understand the forest may differ. But if we have a shared understanding of the forest, we optimize the probability of resolving competing interests.

What we value in the forest; why we value the forest and its survival may be very different.

Some of us care because we see the redwood forest as a source of extraordinary interest and inspiration. We are motivated by the majestic beauty and peace of the ancient, tallest of all living trees.

Some of us care because of forest biodiversity. Globally, forests are one of the most species rich environments on the planet. They support 65 percent of the world's taxa: more forms of life than any other land. In the face of accelerating pressures that are leading to extinction of species at a rate greater than any time since the demise of the dinosaurs, some care about forests to avert loss of biodiversity.

Some of us care because we wish to preserve and enjoy the ecological services of the forest: forests act as a filter to clean water; they decrease erosion and sedimentation they improve air quality; remove pollutants; and function as carbon sinks.

Some of us care because of the commercial value of timber from the forest.

So we may value the forest for inspiration, for its biodiversity, for its ecological services, or its commercial return. Balancing these values, resolving the conflicts arising from the differences in these values drives public policies, policies that are often reflected in regulations.

If we look at policy as the intersection of fact and values, as suggested by Nate Stephenson of the USGS, we can appreciate the pivotal role of developing a broader and deeper shared scientific understanding of the forest. As the facts become clearer and we share an understanding of what the facts are, we gain a critical tool in developing policies that accommodate and resolve competing values.

The existence of the ancient redwood forest in the north coast of California today is the direct result of a policy dating back to 1918 developed by leading scientists who recognized the unparalleled nature of the ancient redwood forest and the undisputed fact that they were at imminent risk of elimination. They acted. They built a shared understanding of the value of the forest and marshaled the necessary resources for success.

---

[1] This opening address was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Executive Director, Save-the-Redwoods League, 114 Sansome Street, #1200, San Francisco, CA 94104.

It's an interesting story. In 1917, three old friends, scientists and conservationists who shared a keen interest in evolution, found themselves together in a gathering in the redwoods west of Santa Rosa. Recognizing a shared intrigue with reports of the ancient redwoods to the north, they undertook an expedition to explore the region recently opened up with the construction of the new State highway. John C. Merriam, Professor of paleontology and Dean of the Faculties at the University of California in Berkeley, Madison Grant, chairman of the New York Zoological Society, and Henry Fairfield Osborn, president of the American Museum of Natural History, traveled north to explore the fabled forest of immense redwoods, and encountered a landscape littered with fallen giants. For mile after mile, they found forests felled at an unprecedented rate, possible because of the highway and developing technology. Trucks had replaced oxen teams, hauling logs from the forest to the mill faster than ever before. Mills worked more and more efficiently, turning logs into lumber. The post World War I market demands and mechanization were laying waste to a forest that had stood for as long as two thousand years.

Imagine then, arriving on the banks of the South Fork of the Eel River where Bull Creek emerges from its stately meander to join the Eel river. Imagine Merriam, Grant and Osborn, stopping, getting out of their car, and walking into the forest, standing at the foot of those massive trees and looking upward, unable to see the tops that towered higher above ground than any other tree in the world. Imagine the profound quiet of the forest.

Today we don't have to imagine what they saw and the quiet they heard because they translated the awe they felt into resolve to save that forest for all time. They realized that no group, public or private, was working actively to preserve the ancient redwoods of the North Coast. On returning to San Francisco, they started Save-the-Redwoods League, dedicated to rescuing from destruction representative examples of our forest primeval.

At that time, there was no State Park system and the National Park system had only recently been created. The only National Parks were formed by Congressional designation of publicly-owned lands that had never been granted to private land owners or with lands that were donated by individuals. Congress was not appropriating money to purchase park lands. In the redwood region, virtually all of the forest land had been homesteaded or granted and were owned by individuals and timber companies, large and small.

In order to rescue the ancient redwoods from destruction, the League recognized it would be necessary to create momentum for the purchase of redwood groves with private resources.

In August of 1919 two of the League's founders again traveled north to Bull Creek and then to Eureka. By the end of their trip, the Chamber of Commerce had resolved to halt logging along the highway and the newly established League had pledged $30,000 to buy the first redwoods for permanent protection provided that the County of Humboldt match that gift. Within weeks the County matched the gift and redwood conservation in the North Coast had begun.

Again several years later, Humboldt County's Supervisors played a key role. The League had been urging the County to act to acquire the majestic ancient forest of Bull Creek and Dyerville Flats. Pacific Lumber Company had patiently deferred harvest for five years, but would delay no longer. In the heat of contentious public hearings representatives of the League came into the chambers with a telegram announcing an anonymous donation of one million dollars, secured in strictest confidence from John D. Rockefeller Jr. The momentum shifted and the vote to save that incomparable forest passed. Clear, irrefutable facts combined with community values to create policy and inspire action.

In the years since, the League has been instrumental in purchasing more than 165,000 acres of forest lands: more than six of every 10 acres in California's redwood state parks.

For 86 years, the decision about where and how to focus the conservation priorities of the redwood forest has been based on extensive surveys and studies. The League's early leaders recognized that improving our understanding of the redwood forests would be critical to effectively rescuing them from destruction. One of the League's long-standing objectives is:

To foster and encourage a better and more general understanding of the value of primeval Redwood or Sequoia and other forests of America as natural objects of extraordinary interest to present and future generations.

The League has long recognized that rescuing the redwood forest primeval from destruction requires more than a fringe of spectacular ancient trees along the traveled highways. Watershed level protection and linkages among core reserves on a landscape-scale have become axiomatic elements of successful conservation. But in the redwood region, many questions remain unanswered. Uncertainties are heightened by global impacts such as climate change, particularly critical in affecting the moisture regimes so pivotal to the health of the world's tallest trees.

In 2000, through Island Press, the League published a book edited by Reed Noss called *The Redwood Forest*. It is a compilation of contributions from some 33 scientists with expertise in the coast redwoods. This book sets the stage for prioritizing the League's work in the coast redwood region. The book includes a focal area model that spatially applies nine ecological criteria to identify the watersheds in the coast redwood range with the highest probability of contributing to the long term survival of the redwoods. Building on that framework, the League is now engaged in a finer scale assessment of conservation opportunities throughout the redwood's two million acre range. In the course of completing the "master plan," questions arise continuously about the design and adequacy of the conservation strategy.

The League also continues to sponsor research through a relatively modest grant program. Several of the papers being presented in this symposium are the results of League-funded studies over the last few years. And we are pleased to announce that we have established research fellowships at Humboldt State University and Cal Poly to support graduate student research with a focus on ancient redwoods.

Today, as we evaluate the effectiveness of the League's actions to save the redwoods many questions of pivotal importance remain. Only four or five percent of the original ancient forest remains. If the redwood forest of the future is to be anchored by ancient trees, we have no margin for error. We need credible solid answers to pressing questions:

1. Are the reserves that are now protected adequate to sustain themselves? Are they large enough? Are they linked appropriately?

2. Is the design of these core reserves adequate to withstand the global changes that we are beginning to experience and that lie ahead? In these tallest of all trees, is the prospect of temperature increase likely to affect the moisture regime, so critical to the health and survival of these trees?
3. Do the reserves represent adequately the range of genetic diversity and species mix of the ecosystem?
4. What role does the second growth forest need to play to ensure the survival of the forest as a functioning unit?
5. What is the essential nature of the old growth forest: its processes and constituent elements, and what does it contribute to the health of the redwood forest as a whole?
6. As we recognize the probability that the ancient redwood forest needs more to survive, what is the role of the surrounding, connecting second growth forest?

I am sure that everyone in this room has other pressing questions.

With the ancient forest, there is no going back. Once the two thousand year old tree is cut it is gone. Once the 800 year old stand is fragmented, it can not be recreated. With time, new stands of big trees can be developed, but the ancient forest can never be re-created. It is not a renewable resource. We need policies guiding the use of the forest that are based on solid science, credible facts.

There is widespread recognition of the importance of facts, of good science, as the key element that drives policies that inspire confidence rather than contention. So long as facts are developed with integrity, free of values; free of distortion that can occur where inquiry is undertaken to rationalize a pre-conceived desired outcome, so long as science is sound, its contribution to policy is invaluable. The moment we allow the desired outcome to drive scientific inquiry, even to shape the form of the hypothesis that is studied, we risk obfuscation and distortion; we risk losing the luminous clarity and consensus that science can bring. There are widespread claims that policy decisions are based on science in an effort to inspire public confidence. Sometimes the claim is that the "best available science" supports underlies the policy. Other times it is "sound science."

The overwhelming importance of value-free science in today's world recently prompted more than 60 well-respected scientists to decry the extent to which this country's political agenda is infecting scientific inquiry. Instead of objective, credible scientific facts being the basis of policy, the facts themselves are being distorted by political values in order to justify policy.

For us at the League, the integrity of scientific inquiry is an issue of overwhelming concern and importance. Not only do we need to measure the success of our work in saving the redwoods based on sound, credible facts about the redwood forest, we need the clarity of sound credible science to guide action as we seek to restore second growth forests to complement and ultimately to function as part of the new "old forests" of the future.

Nearly two years ago, the League spearheaded the purchase of the 25,000-acre Mill Creek tract formerly owned by Stimson Lumber Company. In spite of its intensive industrial management in even-aged units, the property's strategic location linking two premier ancient redwood parks and linking the inland forest to the Pacific

Coast, together with its importance as a coho salmon refugium, made purchase a compelling opportunity. Virtually the entire watershed is now in public ownership.

There could hardly be an acquisition more distinct on the surface from the League's earliest purchase and protection of the ancient trees of Rockefeller Forest. And without question, the challenges of management are profoundly different. The League, other state funding agencies and the County are actively involved in consultation with the Department of Parks in restoration at Mill Creek. Funded by a grant to the League from the Bella Vista Foundation, working with Professor Kevin O'Hara, his graduate student, and a consulting forester, Parks has completed an initial thinning experiment in young stands designed to accelerate development of old forest characteristics. With no more than four or five percent of the ancient forest remaining, the commitment to growing more big trees and restoring forest complexity is an essential tool in permanently saving the redwoods.

As we look to the long term challenges of restoration, the opportunity to learn appears boundless. The uncertainties are magnified as we try to define appropriate human intervention to convert the industrial even-aged legacy of timber operations to the complexity of an ancient forest with fully functioning natural processes that support diverse flora and fauna. We are excited by the opportunity to collaborate, to help focus a restoration network through work at Mill Creek.

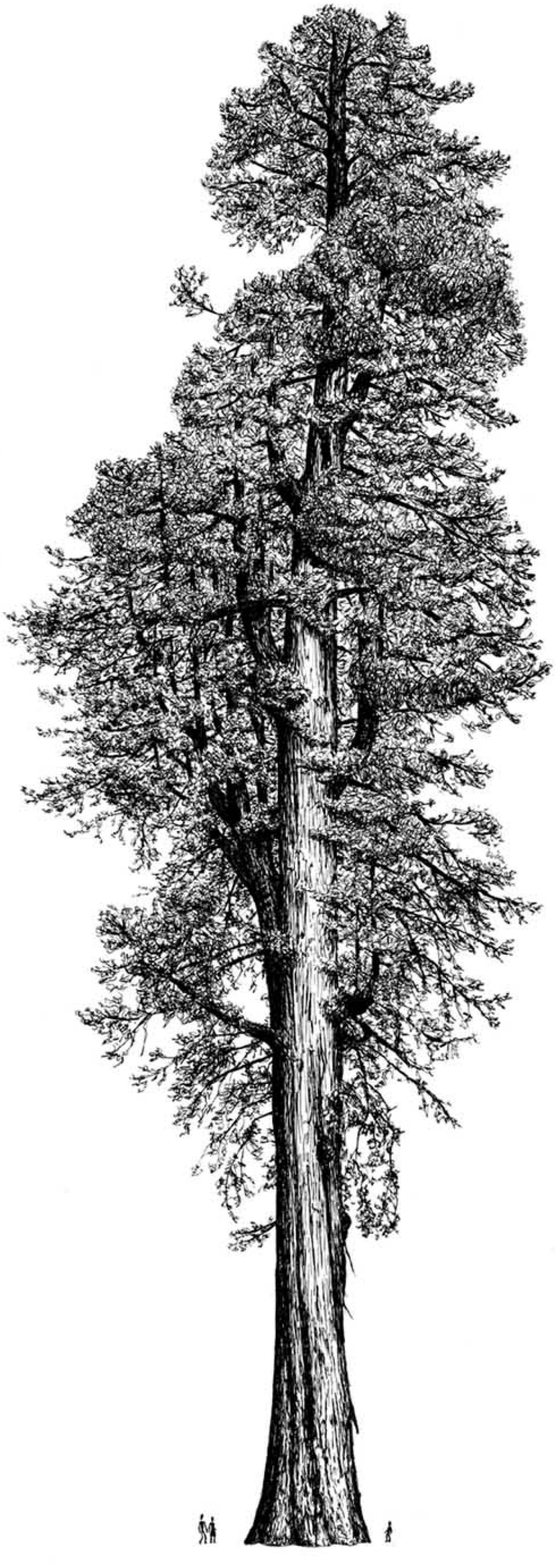
When we look today at the progress in saving the redwoods, we recognize there is much to be done and much to be learned. Critical questions must be answered: how large does a stand need to be to support the natural processes of the forest, what impact will global climate change have on the viability of the redwood forest? How do we protect the habitat within the forest of the microscopic arthropods that comprise part of the forest diversity? How important are the unidentified species of the forest to the forest's long term survival?

The questions go on and on. Some of them will be addressed in this symposium. Many have not yet even been posed with sufficient clarity to generate the hypotheses that will advance our understanding. A deeper understanding will surely inform our strategies and good public policies that balance values based on a common understanding of facts developed with scientific integrity. Whether we care about the forest for its timber value, the biodiversity it supports, the ecological services it produces or the inspiration and peace it brings, we all wish to promote the survival and health of the redwood forest.

E.O. Wilson said, "In order to care deeply about something, it is first necessary to know it." The League is happy to join all of you in these two days to develop a broader and more general understanding of the redwood forest, to know it better.

SESSION 1

Water and Watersheds I



Development of a Mechanistically Based, Basin-Scale Stream Temperature Model: Applications to Cumulative Effects Modeling[1]

Douglas Allen,[2] William Dietrich,[2] Peter Baker,[3] Frank Ligon,[3] and Bruce Orr[3]

Abstract

We describe a mechanistically-based stream model, *BasinTemp*, which assumes that direct shortwave radiation moderated by riparian and topographic shading, controls stream temperatures during the hottest part of the year. The model was developed to support a temperature TMDL for the South Fork Eel basin in Northern California and couples a GIS and a 1-D energy balance model. Spatially varying insolation is calculated in the GIS and heat and mass transfer processes are modeled using a simple steady-state scheme integrated with an optimization procedure which improves model predictions. *BasinTemp* can be applied to basins of varying sizes and requires minimal measured input data. Model predictions for three sub basins in the South Fork Eel yielded RMSE statistics ranging from 0.25 ºC to 0.30 ºC. The model also performed well using pooled data for all three sub basins, yielding an RMSE of 0.36 °C. *BasinTemp* has been used to assess local and downstream stream heating effects after modifying riparian shade. Model predictions for the three sub basins illustrate the importance of riparian shade provision on low order channels and show the shifts in the quality and quantity of potential coho habitat following different shade prescriptions.

Key words: stream temperature prediction, model, BasinTemp, riparian shade, cumulative effects

Introduction

The transformation of the Pacific Northwest and California landscape that followed the arrival of Europeans was accompanied by, and more often at the expense of, significant changes in the quality and quantity of terrestrial and aquatic biological habitat. Timber harvesting practices have been responsible for much of this change and particularly for the decline in salmonid habitat (for example, Beschta and others 1987, Lichatowich 1999). Degradation of water quality parameters, especially water temperature, has significantly reduced coldwater fish habitat (Lichatowich 1999). Coldwater fish species with physiological adaptations to cool freshwater conditions are especially vulnerable to temperature fluctuations.

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] University of California, Berkeley, Department of Earth and Planetary Science, McCone Hall, UC Berkeley, CA 94720. email: allen@eps.berkeley.edu

[3] Stillwater Sciences, 2855 Telegraph Ave., Suite 400, Berkeley, CA 94705.

Regulatory efforts to halt the decline in salmonid habitat and at the same time promote recovery and restoration, have been administered chiefly through the Endangered Species Act and the Clean Water Act (Poole and others 2001b). Towards meeting legislative requirements contained in these statutes a broad range of analytical and empirical tools have been applied to assess and quantify elevated stream temperature conditions (Deas and Lowney 2000). The model presented here, *BasinTemp* (Allen 2006), was developed to support work on a temperature TMDL (Total Maximum Daily Load) conducted for the South Fork Eel River in Northern California. The basin was listed for temperature (and sediment) under section 303(d) of the Clean Water Act primarily as a result of degradation of coho *(Oncorhynchus kisutch)* habitat. While temperature influences all coho life-stages, the summer rearing life stage is especially sensitive to change (Cafferata 1990). Coho were once abundant throughout the South Fork Eel basin, supporting a major industry well into the early 20th Century before overfishing forced its closure (U.S. Environmental Protection Agency 1999). A brief recovery was followed by a further decline in numbers in the latter half of the last century (U.S Environmental Protection Agency 1999). The TMDL objectives for the South Fork Eel (U.S. Environmental Protection Agency 1999) required locating stream reaches with elevated temperatures, identifying the source (or sources) responsible for these elevated temperatures, and developing methodologies to evaluate shade requirements necessary to meet the TMDL requirements (U.S Environmental Protection Agency 1999). Fulfilling these objectives argued for a mechanistically-based model that could be applied to large basins, that could be rapidly deployed, and required limited input data to operate. In addition, recognizing the crucial role of riparian vegetation for blocking direct insolation, we reasoned that the model needed to represent accurately the riparian shading affects, and to include the facility to modify riparian vegetation to explore the effects different shade scenarios on water temperature.

We determined that none of the temperature models available at the time fully met these criteria. Simple empirical models (for example, Mitchell 1999) and simple reach-based mechanistic models (for example, Brown 1969) were inadequate or inappropriate for large-basin applications. Fully-mechanistic temperature models also didn’t meet our needs. Reach-based, physical models [for example, SSTEMP (Bartholow 2000); HeatSource (Boyd and Kasper 2003)] were inappropriate for basin-scale application, while the spatially distributed physical models [for example, SNTEMP (Theurer and others 1984); HSPF (Bicknell and others 1997)], require substantial field-measured input data to operate. These data were unavailable and would have been prohibitively expensive to acquire. To fulfill the TMDL requirements, we developed a physically-based, large-area assessment model, *BasinTemp*, which complements rather than competes with existing fully-mechanistic physically-based models. Temperature predictions generated by *BasinTemp* may be used to guide physically-based model applications where more detailed information about the individual mechanisms responsible for stream heating (air temperature, evaporation rate, streambed conduction, and so forth) is required.

Model Development

The temperature of a water body is a function of the total heat energy contained in a discrete volume of water,

$$\frac{E}{V} = C_p \cdot \rho \cdot T_w \qquad (1)$$

where E is heat energy (calories), V is the volume (m^3) of water, Cp is specific heat capacity of water (1000·cal/kg·K), ρ is the density of water (1000·kg/m^3), and T_W is water temperature in units of degrees centigrade. Most mechanistically-based models which are designed to explicitly quantify the various mass and heat transfer mechanisms, invest the majority of effort toward quantifying the fluxes that comprise the energy term, E (Deas and Lowney 2000).

In mid-latitude regions during the hottest time of the year, direct insolation dominates the heat energy budget (Brown 1969, Ice 2001, Sansone and Lettenmaier 2001), contributing up to 80 percent of the total radiation budget (Monteith and Unsworth 1990). In forested catchments dominated by lower order channels, riparian vegetation is the primary control on the amount of direct solar radiation received at the stream surface (Lynch and others 1984, Poole and Berman 2001). In mid-latitude regions during the summer, streamflow is almost exclusively maintained by groundwater influx which enters the stream at temperatures that are reasonably well represented by the local mean annual air temperature (Beschta and others 1987). Based on this information, we determined that the essential components of summertime stream heating could be captured in a model where solar insolation dominates the fluxes that comprise the energy term, E. The model, *BasinTemp* (Allen 2006), is a mechanistically-based approach which assumes that direct shortwave radiation drives summertime stream heating and the contribution of direct shortwave radiation reaching the stream surface is controlled by shade from local topography and riparian vegetation. Heat and mass transfer processes are represented very simply in a 1-D, steady-state numerical model which assumes that water is fully mixed in the vertical and horizontal directions. An optimization routine is integrated with the 1-D heat balance model and uses locally measured stream temperature data to improve model predictions. Data preprocessing is performed in a GIS where the data required for spatially distributed insolation modeling are assembled. A hybrid topography-vegetation digital elevation model (DEM) is merged using existing digital elevation data and vegetation information converted to tree heights for the area of interest. Model simplicity is maintained by ignoring the contribution of shortwave radiation transmitted through the canopy. While important in locally-select cases, canopy-transmitted shortwave radiation is of secondary importance compared to the direct shortwave contribution (Reifsnyder and Lull 1965). A vector-based stream channel network is discretized into uniform segment lengths, where the length of each segment is scaled to match the resolution of the source elevation and vegetation data. Low-flow channel geometry is computed using a power relationship between drainage area and field-measured low-flow widths for the area of interest. Solar insolation—comprised of direct, diffuse, and reflected shortwave radiation contributions—is computed using radiative transfer routines packaged with the Image Processing Workbench (IPW) (Dubaya and others 1990, Frew 1990). The insolation model predicts daily integrated, spatially-varying insolation for every DEM grid cell for the geographic location of interest. Insolation predictions are passed to the 1-D heat balance model which computes for every stream segment, the heat energy transferred out of the reach via stream flow. This heat energy is the sum of heat transported into a reach from upstream, heat entering the reach by groundwater seepage, and heat energy supplied by shortwave radiation (in units of W/m^2). Low-flow hydrology is treated very simply—discharge leaving each reach is computed as the sum of discharges from reaches upstream and local groundwater seepage into the reach. The rate of groundwater seepage is assumed to be a fixed, linear constant whose value is calculated so that predicted discharges at reference reaches match observed low-flow discharges at gages at the location and time period of interest.

Assuming steady-state conditions, the energy balance is solved by a simple ordinary differential equation:

$$\frac{dh}{dx} = \alpha T_{GW} \frac{dq}{dx} + \left\{ K_0 I + K_1 + K_2 \left(K_3 - \frac{h}{\alpha\, q} \right) \right\} w \qquad (2)$$

h(x) is the heat flux across a surface perpendicular to the reach at a distance x from the reach head,
q(x) is water flux across this same surface (cms/km)
w is the reach width (m)
I is solar irradiance at the stream surface (W/m^2)
α is a constant which converts that energy in units of W/m^2 to calories, and mks units to cgs.
T_{GW} is the groundwater temperature parameter (°C), and
K_0, K_1, K_2, and K_3 are model parameters. In all runs K_0 was set to 1.0.

The predicted water temperature for each stream segment is then simply the ratio of heat energy calculated by *equation 2* and flow volume. An optimization routine applying a model trust region method (Dennis and Schnabel 1996) using full second derivative information is integrated with the 1-D heat balance model. The objective function minimized by the optimization routine is the simple root mean square error between predicted and observed reaches at calibration reaches. The routine fulfills the goals of a simple, rapid deployment model for large basins but at the expense of explicitly quantifying the various heat exchange mechanisms at the air-water, and water-streambed interfaces. *BasinTemp* was developed primarily for application to aquatic biology issues and thus justifying lumping these exchange processes into the fitting parameters. Furthermore, most physically-based models perform parameter tuning exercises, and where field measured data are sparse, the range of values used for these parameters (notably wind speed and evapotranspiration) often fall well beyond physically realistic values. Formally introducing an optimization scheme into the model allows for calibration and sensitivity analyses to be conducted objectively and reproducibly. Minimum source data required to run the model are shown in *table 1* which also lists the source data used for the South Fork Eel. In summary, the main features of the model are: (a) application to basins of varying size, (b) transferable to basins with very different characteristics than those where it was developed, (c) limited and flexible input data requirements, (d) water temperatures are predicted for every reach segment and then routed downstream, permitting assessment of local and cumulative downstream temperature effects, (e) modification of riparian tree height for all or part of the basin of interest, permitting the assessment of different land management scenarios on local and basin-scale water temperatures.

Table 1—*Minimum source data required to run* BasinTemp *and source data used for South Fork Eel sub basin predictions*

Data type	Minimum required	Data used for S. Fork Eel
Topography	30-meter digital elevation model	30-meter USGS DEM
Tree height	Vegetation tree height. Sources may include aerial photographs, field measured plot data; satellite imagery, and so forth.	Landsat TM imagery classified according to the California Wildlife Habitat Relations (CWHR) system (Fox and others 1997)
Stream network	Vector-based channel network resolved at a minimum scale of 1:24,000	1:24,000 USGS DLG blueline hydrography
Channel geometry	Low-flow channel width for entire stream network	Power-law relationship between drainage area and field measured low-flow width for reaches throughout the South Fork Eel.
Low-flow Discharge	Measured (daily averages) low-flow discharges for the MWAT week.	7-day mean daily discharge data from USGS gages at Weott in Bull Creek and on mainstem Elder Creek. Discharge data from nearby Tenmile Creek were used for Rattlesnake Creek.
Observed stream temperature data	Array of thermograph stations sufficient to account for basin size, drainage density, and vegetation and lithologic heterogeneities.	1996-1997 thermograph data compiled by Humboldt Country Resource Conservation District (Friedrichsen 1998, Lewis and others 2000)

Study Area

The South Fork Eel river basin is almost equally divided among Mendocino and Humboldt Counties in Northern California and drains an area of approximately 1,800 km^2 (see insert map in *fig. 1*). The climate is generally Mediterranean type and characterized by long, warm summers and cool, wet winters. Mean annual precipitation ranges from 1,500 mm to 1,800 mm most of which falls between October and April (James 1983). Summer high temperatures can exceed 31 °C (Mast and Clow 2000). Approximately 20 percent of the basin is owned by State Parks and the Bureau of Land Management and a small portion is owned by large timber industries, while the remainder is owned by small landholders, ranchers, and residential communities. Temperature modeling focused on three watersheds within the South Fork Eel Basin. The sub basins, Bull Creek, Elder Creek, and Rattlesnake Creek captured a broad range of topographic, lithologic, and vegetation characteristics, and landuse histories observed in the South Fork Eel basin. Elder Creek (drainage area 17 km^2) is a largely undisturbed basin containing one of the last remaining stands of old growth Douglas Fir in California. Rattlesnake Creek (drainage area 99 km^2) is characterized by gently rolling terrain and broad areas of grassland and chaparral vegetation. Bull Creek (drainage area 112 km^2) is almost equally comprised of old growth forest and disturbed areas.

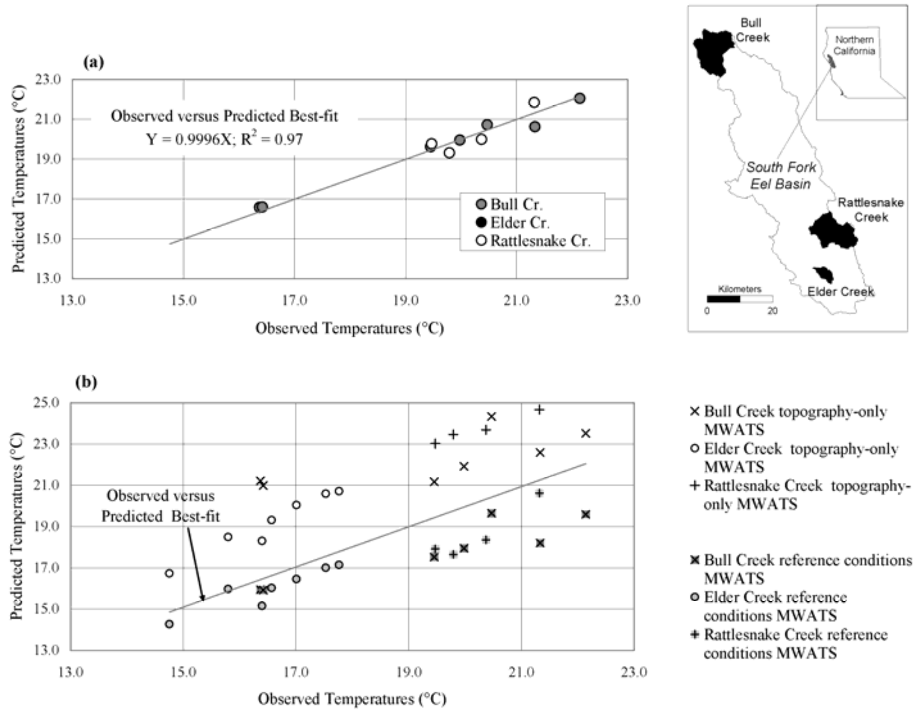


Figure 1—Simultaneous fit to observed 1996 and 1997 temperatures for Bull, Elder and Rattlesnake Creeks for, (a) current vegetation conditions, and (b) topography-only and reference vegetation conditions

Input data used for the three sub basins are shown in *table 1*. Sources for these data are identical to those used in the South Fork Eel TMDL (U.S. Environmental Protection Agency 1999), however the availability of higher resolution digital elevation data and improvements in processing capability (allowing for improved drainage area and low-flow width calculation) means that some of the results reported here differ from the results reported in the original TMDL (U.S. Environmental Protection Agency 1999). Most of the changes between the original predictions and those reported here are the result of significant changes to the 1996 thermograph data used in the calibration and optimization exercise. These data underwent several significant changes from the original 1998 report (Friedrichsen 1998) to the final report (Lewis and others 2000), which was published after work on the TMDL had been completed. The nature of these changes and effects on model predictions are discussed in detail elsewhere (Allen 2006).

The hybrid vegetation-topography DEM is comprised of 10-meter USGS (resampled to 30-meters) digital elevation data and Landsat Thematic Mapper (TM) imagery classified according to a modified California Wildlife Habitat Relations scheme (Fox and others 1997). Tree heights are computed using published diameter-at-breast-height (DBH) relations (Burns and Honkala 1990, Fowells 1965, Mayer and Laudenslayer 1988, Sawyer and Keeler-Woolf 1995, Whitney 1985). The blueline hydrography channel networks for the three sub basins were discretized into 25-meter long reaches, approximately matching the pixel resolution of the Landsat TM data. Every stream segment is attributed with a low-flow width calculated using a power relationship between drainage area and field-measured low-flow channel widths for the South Fork Eel (Stillwater Sciences 1998[4]). Shortwave radiation is computed for every grid cell for the time period of interest (week-ending July 31, 1996 and 1997). Atmospheric transmission parameters were assigned values appropriate for clear-sky, rural conditions. The groundwater seepage constant was computed for each sub basin by iteratively adjusting the rate until the predicted mean low-flow discharge for the stream segment nearest to a local USGS gage station matched the measured mean low-flow discharge for the time period of interest. No gage station exists in Rattlesnake Creek so low-flow data from a (now obsolete) gage in nearby Tenmile Creek were used to calculate the groundwater seepage rate. Thermograph data collected by the Humboldt County Resource Conservation District (Friedrichsen 1998, Lewis and others 2000) were used for calibration and optimization of model predictions. The model predicted the 7-day running Mean Weekly Average Temperature (MWAT) for every stream segment. Several researchers have shown that salmonid growth rates (Brungs and Jones 1977) and salmonid presence/absence (Welsh and others 2001) are closely correlated with the MWAT metric and the metric is also used by the National Marine Fisheries Service and the U.S. Fish and Wildlife Service (NMFS and USFW 1997).

Results

The model was initially calibrated using Bull Creek data where observed MWATs for July 31, 1996 were available for seven thermograph stations within the basin. Daily integrated insolation predictions for this period are computed for each grid cell and passed to the heat balance model. The groundwater temperature parameter, T_{GW}, is constant and set to a physically realistic temperature (11.8 °C), approximately matching the mean annual air temperature. The coefficient of insolation parameter, K_0, is fixed at 1.0 for all model runs. The parameter can be varied during the fitting process, however considerable effort is devoted to characterizing accurately the shortwave flux reaching the stream surface and the local vegetation regime that control the amount reaching the surface, and so ideally this parameter should be fixed at 1.0. Sensitivity analyses (Allen 2006) indicate that when allowed to vary, this coefficient remains at or very close to unity. The remaining three parameters, K_1, K_2, and K_3, are all used to optimize model predictions but only two are allowed to vary at any one time. The MWAT predictions generated by the model for Bull Creek, were very satisfactory (root mean square error equal to 0.25 °C and R^2 = 0.99), and appeared to validate the simplifying assumptions embedded in the model.

To test the robustness and reliability of *BasinTemp* predictions, we applied the model to Rattlesnake and Elder Creeks. Thermograph data for these basins for both 1996 and 1997 were far more limited. Neither station contained sufficiently reliable observed MWAT data for 1996 and 1997, however both basins included an adequate number of thermographs recording the seven-day running mean of the daily average (Weekly Average Temperature or WAT) for the week ending July 31, 1997, and this is the metric the model was used to predict both for Elder and Rattlesnake Creeks. The model performed well for Elder Creek, with an RMSE of 0.30 °C and an R^2 of 0.91. Results for Rattlesnake Creek were less successful yielding an RMSE of 0.93

[4] Unpublished data. Thirty-five low-flow widths were measured during July and August, 1998 from sites throughout the S.F. Eel basin.

°C and an R^2 of 0.18. WATS predictions from two thermographs were primarily responsible for the diminished model performance. When these two thermographs are eliminated from the calibration exercise, model predictions improve significantly, yielding an RMSE of 0.26 °C and an R^2 of 0.87. Both stations are located immediately downstream of tributary junctions, and thus if the combined flow from the tributary and mainstem channels are not fully mixed, the thermographs may record unrepresentative temperatures. The groundwater seepage rate used for Rattlesnake Creek computed from low-flow discharge data from the adjacent Tenmile Creek basin is almost certainly inaccurate. Furthermore, our assumption of a linear groundwater seepage rate is an oversimplification. Relief, lithologic, and structural heterogeneities, and vegetation variation across a basin guarantee that seepage rates and hence low-flow discharge must also vary spatially. An application of *BasinTemp* to Canton Creek (Allen 2006), a 164 km^2 basin which drains into the North Umpqua River in Southern Oregon has revealed the limitations of the basin-wide, linear groundwater accretion rate assumption. Spatially varying low-flow discharges across the Rattlesnake Creek basin may also explain the poorer model performance.

An additional test of model performance and reliability was conducted by generating temperature predictions using combined data for all three sub basins. The various components of this validation exercise included predicting temperatures using source data for three basins each characterized by different lithologies, vegetation coverage, and landuse histories. Groundwater seepage rates were computed from two different water years, 1996 for Bull Creek, and 1997 for Elder and Rattlesnake Creeks. The calibration data used in the fitting exercise were comprised of two different temperature metrics, 1996 MWATS for Bull Creek (for the week ending July 31), and 1997 WATS (also for the week ending July 31) for Elder and Rattlesnake Creeks. We added an additional test by permitting only two fitting parameters (K_2 and K_3) to be used during the optimization exercise, only one of which was allowed to vary for any given iteration. Individual sub basin results and the combined results are shown in *figure 1(a)*. The model performed well for the individual basin predictions and for the combined basin prediction. These results provide further support for the simple assumptions incorporated into the model and for the parsimonious model structure. The results illustrate the very different thermal characteristics of each sub basin while also having indirect implications for land management strategies. The predictions for Elder and Rattlesnake Creek occupy the lower left and upper right portions of the graph respectively with no overlap. Bull Creek model predictions bracket the entire temperature range, reflecting the broad range of physical conditions and landuse activities characteristic of that basin. The implications from these results are that improved temperature conditions (exemplified by Elder Creek results) can be achieved by improved riparian shading, and hence shifting temperatures from right-to-left on the graph.

Discussion

A critical feature built into the model is the facility to perform scenario testing by modifying the riparian shading regime. These modifications, which involve changing tree heights, can be performed for an entire basin or selected areas within a basin. The impacts on stream temperatures can be assessed at individual reach scales up to entire basin scales, providing the capability to examine local and cumulative temperature effects after modification of the riparian environment. Predictions for two scenarios which bracket the range of potential riparian shading scenarios were performed for the three sub basins. The topography-only scenario predicts temperatures where shade is supplied solely by local topography, and the reference-state scenario assumes natural, undisturbed conditions where trees are assigned representative late seral heights. The insolation predictions for these scenarios are passed to the 1-D heat balance model which uses the fitting parameters computed for current conditions predictions. *Figure 1(b)* shows results for these scenarios using eighteen thermograph calibration sites in Bull, Elder, and Rattlesnake Creeks. As expected, temperature predictions in Elder Creek change little from the current conditions to the reference state scenario. Predictions for all three sub basins for the topography-only scenario show significantly elevated temperatures above the current condition predictions, and well exceeding 4 °C for several stations. Interestingly, predictions for Rattlesnake Creek illustrate that warm temperatures predicted for current conditions are only slightly decreased for the reference state condition, suggesting that for some naturally warm basins, shade prescriptions may result in only minor temperature reductions. The results for Rattlesnake Creek also support the argument for applying basin-specific criteria to water quality regulation rather than applying a single temperature metric or a temperature threshold for entire basins.

The model has also been used to predict temperatures for sub basin scenarios in Bull Creek, following shade modification on first and second order channels. The results from these sub basin tests provide additional support for the body of work illustrating cumulative temperature effects while indirectly contributing to the debate over timber harvesting impacts on aquatic habitat quality and quantity and the effectiveness of forest practice rules for protecting habitat (SRP 1999). A wealth of research has demonstrated the deleterious impacts of timber harvesting on water temperature and aquatic habitat (for example, Beschta and others 1987, Beschta and Taylor 1988, Brown 1969, Johnson and Jones 2000, Kopperdahl 1971, and others). In some cases, recovery to pre-harvest levels can take up to two decades (Beschta 1989). While there is general agreement over the various mechanisms responsible for stream heating, a vigorous debate persists over which are the controlling mechanisms (Bartholow 2000). A few studies have suggested that a local warm environment and ambient conditions control water temperature and have also rejected the role of cumulative downstream heating (for example, Caldwell and others 1991; Larson and Larson 1996, 1997; Sullivan and others 1990; Zwienicki and Newton 1997). However, several authors have suggested that most of these studies are based on inconsistent assumptions, flawed experimental designs, or both (see Beschta 1997, Beschta and others 2003, NCASI 2001, Poole and others 2001).

Figure 2 shows results for four Bull Creek tributaries following shade modification on first and second order channels. In addition to showing current predictions, each graph shows temperature predictions for full reference vegetation shading on first and second order channels, and topography-only shading. The right ordinate on each graph displays four temperature categories originally applied in the South Fork Eel TMDL (U.S Environmental Protection Agency 1999) and which categorize salmonid habitat conditions as a function of temperature. In an effort to delineate potential coho habitat, slope thresholds for 10 percent and four percent (if present) channel gradients are also indicated. These thresholds correspond to the furthest upstream extent of the connected channel network at gradients equal to or less than four percent and 10 percent slope and provide a quantitative indicator of potential coho habitat. The four percent threshold identifies the approximate upstream limits of coho spawning habitat (Reeves and others 1989), and the 10 percent threshold is an approximate indicator for the maximum upstream extent of coho rearing habitat (Meehan and Bjornn 1991). Results for the four tributaries very clearly demonstrate elevated downstream heating for the topography-only shading scenario for all tributaries, and especially for the East-West orientated creeks (Cow Creek and Mills Creek). For the reference vegetation scenario, downstream temperatures are notably suppressed on all four tributaries.

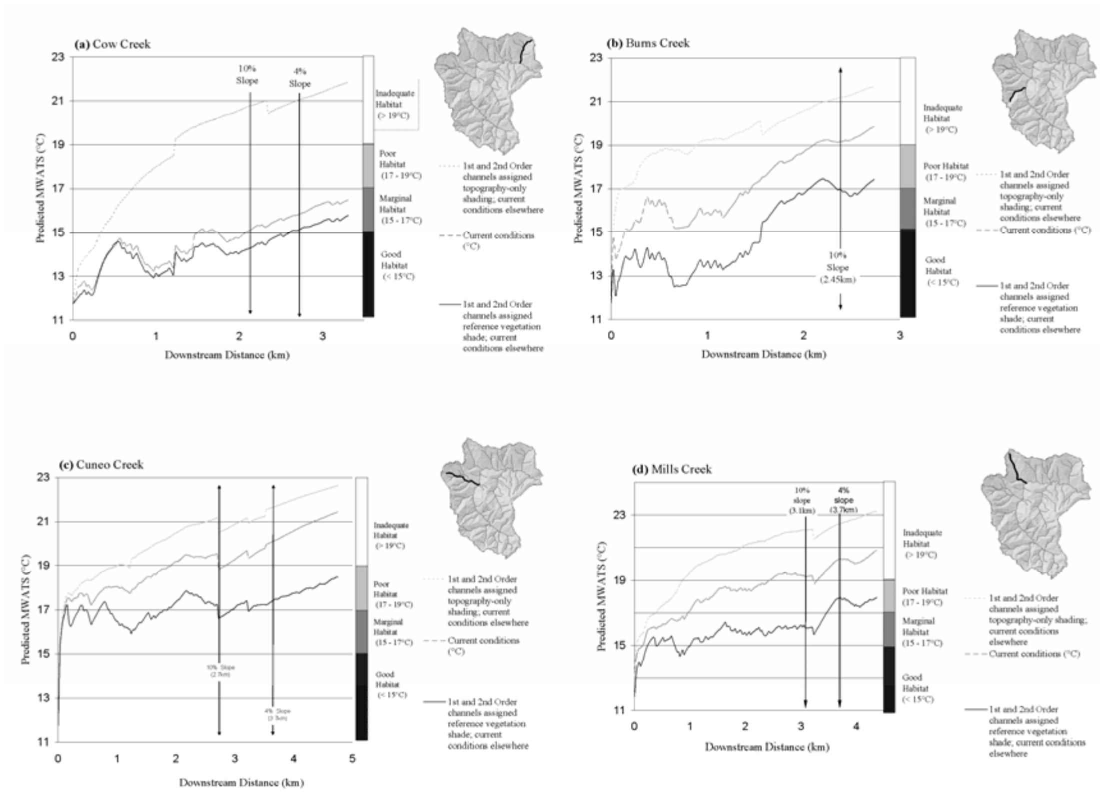


Figure 2—Downstream temperature changes following upstream shade modification on first and second order channels for four Bull Creek Tributaries.

Conclusions

We present a mechanistically-based model which assumes that direct shortwave radiation drives summertime stream temperatures and that emphasizes the importance of riparian shading for controlling insolation received at the stream surface. The model performs well for basins of varying sizes and with different vegetation, topographic, and lithologic characteristics. However, where low-flow discharges show significant spatial variability, the model performs less well. The steady-state assumption embedded in the model precludes predicting diurnal and instantaneous maximum and minimum temperatures. The model shows considerable promise, however, for temperature assessments for large watersheds and thus complements the existing physically-based models currently applied in the regulatory framework. Sub basin applications illustrate significant downstream cumulative heating effects after modifying riparian shading on low order channels, lending further support to the body of work demonstrating the role of cumulative effects (for example, Bartholow

2000, Beschta and Taylor 1988, Rowe and Taylor 1994). These results also support the arguments for providing greater protection to low order, non-fish bearing streams (Ligon and others 1999) which receive minimal protection under current California Forest Practice Rules (CDFG 1999).

References

Allen, D. McK. 2006. **Development and application of a mechanistically-based stream temperature model**. Berkeley, CA: University of California Department of Earth and Planetary Science. Ph.D. dissertation, in preparation.

Bartholow, J.M. 2000. **Estimating cumulative effects of clearcutting on stream temperatures**. Rivers 7(4): 284-297.

Beschta, R.L. 1997. **Riparian shade and stream temperature: an alternative perspective**. Rangelands 19(2): 25-28.

Beschta, R.L. 1989. **The effects of riparian vegetation on channel morphology, sediment, and temperature in streams**. In: Silvicultural Management of Riparian Areas for Multiple Resources, A COPE Workshop, December 12-13, Gleneden Beach, OR; 8 p.

Beschta, R.L.; McIntosh, B.A.; Torgersen, C.E. 2003. **Comment: perspectives on water flow and the interpretations of FLIR images**. Journal of Range Management 56: 97-99.

Beschta, R.L.; Taylor, R.L. 1988. **Stream temperature increases and land use in a forested Oregon watershed**. Water Resources Bulletin 24: 19-21.

Beschta, R.L.; Bilby, R.E.; Brown, G.W.; Holtby, L.B.; Hofstra, T.D. 1987. **Stream temperature and aquatic habitat: fisheries and forestry interactions**. In: Salo, E.O.; Cundy, T.W., eds. Streamside management: forestry and fishery interactions. Contribution No. 57. Seattle, WA: University of Washington, College of Forest Resources; 191-223.

Bicknell, B.R.; Imhoff, J.C.; Kittle, J.L.; Donigian, A.S., Jr.; Johanson, R.C. 1997. **Hydrological simulation program.** In: FORTRAN, user's manual for version 11. U.S. Environmental Protection Agency National Exposure Research Laboratory, Athens, GA. EPA/600/R-97/080.

Boyd, M.; Kasper, B. 2004. **Analytical methods for dynamic open channel heat and mass transfer: methodology for the heat source model version 7.0**. *Available at* http://www.heatsource.info

Brown, G.W. 1969. **Predicting temperatures of small streams**. Water Resources Research 5: 68-71.

Brungs, W.A.; Jones, B.R. 1977. **Temperature criteria for freshwater fish: protocol and procedures**. Duluth, MN: US Environmental Protection Agency, Environmental Research Laboratory. EPA-600/3-77-061.

Burns, R.M.; Honkala, B.H. 1990. **Silvics of North America, Vol. 1, Conifers**. Washington DC: USDA Forest Service Agriculture Handbook 654.

Cafferata, P. 1990. **Watercourse temperature evaluation guide**. JDSF Newsletter No. 39. Fort Bragg, CA: Jackson Demonstration State Forest, California Department of Forestry and Fire Protection.

Caldwell, J.; Doughty, K.; Sullivan, K. 1991. **Evaluation of downstream temperature effects of Type 4/5 waters**. Olympia, WA: Washington Department of Natural Resources.Timber/Fish/Wildlife Report TFW-WQ5-91-004.

California Department of Forestry and Fire Protection. 1999. **California forest practice rules: title 14**, California Code of Regulations, Chapters 4 and 4.5. South San Francisco, CA: Barclay Law Publishers.

Deas, M.L.; Lowney, C.L. 2001. **Water temperature modeling review**. California Water Modeling Forum, Central Valley, CA. *Available at* http://cwemf.org/Pubs/ BDMFTempReview.pdf

Dennis, J.E., Jr.; Schnabel, R.B. 1996. **Numerical methods for unconstrained optimization and nonlinear equations**. Philadelphia, PA: SIAM.

Dubayah, R.; Dozier J.; Davis, F.W. 1990. **Topographic distribution of clear-sky radiation over the Konza prairie, Kansas**. Water Resources Research 26: 679-690.

Fowells, H.A., ed. 1965. **Silvics of forest trees of the United States**. Washington, D.C.: U.S. Department of Agriculture, Agricultural Handbook 271; 203 p.

Fox, L.; Bonser, G.L.; Trehey, G.H.; Buntz, R.M.; Jacoby, C.E.; Bartson, A.P.; La Brie, D.M. 1997. **A wildlife map and database for the ORCA (Oregon-California) Klamath bioregion derived from Landsat imagery**. Arcata, CA: Klamath Bioregional Assessment Project, College of Natural Resources, Humboldt State University, CA.

Friedrichsen, G. 1998. **Eel River water quality monitoring report**. Fields Landing, CA: Humboldt County Resource Conservation District. Final report, 205(j) submitted to California State Water Quality Control Board.

Frew, J. 1990. **The image processing workbench.** Santa Barbara, CA: University of California Department of Geography. Unpublished Ph.D. dissertation.

James, S.M. 1983. **South Fork Eel watershed erosion investigation.** Northern District: California Department of Water Resources; 95 p.

Johnson, J.L.; Jones, J.A. 2000. **Stream temperature response to forest harvest and debris flows in western Cascades, Oregon.** Canadian Journal of Fisheries and Aquatic Sciences 57 (Suppl. 2): 30-39.

Ice, G. 2001. **How direct solar radiation and shade influence temperature in forest streams and relaxation of changes in stream temperature.** Cooperative Monitoring, Evaluation and Research (CMER) Workshop: Heat transfer processes in forested watersheds and their effects on surface water temperature; February 2001, Lacey, WA; 34 p.

Larson, L.L.; Larson, S.L. 1996. **Riparian shade and stream temperature: a perspective**. Rangelands 18(4): 149-152.

Larson, L.; Larson, P. 1997. **The natural heating and cooling of water**. Rangelands 19: 6-8.

Lewis, T.E.; Lamphear, D.W.; McCanne, D.R.; Webb, A.S.; Krieter, J.P.; Conroy, W.D. 2000. **Regional assessment of stream temperatures across northern California and their relationship to various landscape-level and site-specific attributes**. Forest Science Project. Arcata, CA: Humboldt State University Foundation; 420 p.

Lichatowich, J. 1999. **Salmon without rivers: a history of the Pacific salmon crisis**. Washington, DC: Island Press; 333 p.

Mast, M.A.; Clow, D.W. 2000. **Environmental characteristics and water quality of hydrologic benchmark network stations in the Western United States**. US Geological Survey Circular 1173-D; 115 p.

Mayer, K.E.; Laudenslayer, W.F., Jr., eds. 1988. **A guide to wildlife habitats of California**. Sacramento, CA: State of California, Resources Agency, Department of Fish and Game.

McCullough, D.A.; Spalding, S.; Sturdevant, D.; Hicks, M. 2001. **Summary of technical literature examining the physiological effects of temperature on salmonids**. Seattle, WA: U.S. Environmental Protection Agency. Issue Paper 5, EPA Region 10 Temperature Water Quality Criteria Guidance Development Project.

Meehan, W.R.; Bjornn, T.C. 1991. **Salmonid distributions and life histories.** In: Meehan, W.R., ed. **Influences of forest and rangeland management on salmonid fishes and their habitats**. Bethesda, MD: American Fisheries Society Special Publication 19: 47-82.

Mitchell, S. 1999. **A simple model for estimating mean monthly stream temperatures after riparian canopy removal**. Environmental Management 24(1): 77-83.

Monteith, J.L.; Unsworth, M. 1990. **Principles of environmental physics**. 2d ed. London, UK: Routledge, Chapman, and Hall; 291 p.

National Council for Air and Stream Improvement, Inc. 2001. Annotated/abstracted bibliography: **Riparian canopy cover, microclimate, and stream temperatures**. *Available at* http://www.ncasi.org/Publications/Detail.aspx?id=2623

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 1997. **Aquatic properly functioning condition matrix**. NMFS, Southwest Region, Northern California Area Office, Santa Rosa and USFWS, Arcata, CA.

Poole, G.C.; Risley, J.; Hicks, M. 2001a. **Spatial and temporal patterns of stream temperature**. Seattle, WA: U.S. Environmental Protection Agency. EPA Region X Temperature Water Quality Criteria Guidance Development Project Issue paper 3, EPA-910-D-01-003.

Poole, G.; Dunham, J.; Hicks, M.; Keenan, D.; Lockwood, J.; Materna, E.; McCullough, D.; Mebane, C.; Risley, J.; Sauter S.; Sturdevant, D. 2001b. **Scientific issues relating to temperature criteria for salmon, trout, and char native to the Pacific Northwest**. Seattle, WA: U.S. Environmental Protection Agency. Technical Report: EPA 910-R-01=007.

Reeves, G.H.; Everest, F.H.; Nickelson, T.E. 1989. **Identification of physical habitats limiting the production of coho salmon in western Oregon and Washington**. Gen. Tech. Rep. PNW-GTR-245. Portland, OR: USDS Forest Service Pacific Northwest Research Station.

Reifsnyder, W.E.; Lull, H.W. 1965. **Radiant energy in relation to forests**. Washington, DC: USDA Forest Service Technical Bulletin No. 1344; 108 p.

Rowe, L.K.; Taylor, C.H. 1994. **Hydrology and related changes after harvesting native forest catchments and establishing *Pinus radiata* plantations**. **Part 3. Stream temperatures**. Hydrological Processes 8: 299-310.

Sansone, A.L.; Lettenmaier, D.P. 2001. **A GIS-based temperature model for the prediction of maximum stream temperatures in the Cascade mountain region**. Seattle, WA: University of Washington. Water Resources Series: Technical Report No. 168.

Sawyer, J.O.; Keeler-Wolf, T. 1995. **A manual of California vegetation**. Sacramento, CA: California Native Plant Society.

Stillwater Sciences. 2002. **Stream temperature indices, thresholds, and standards used to protect coho salmon habitat: a review**. Unpublished white paper prepared for Campbell Timberland Management, Fort Bragg, CA.

Sullivan, K.; Tooley, J.; Doughty, K.; Caldwell, J.; Knudsen, P. 1990. **Evaluation of prediction models and characterization of stream temperature regimes in Washington**. Olympia, WA: Washington Department of Natural Resources. Timber/Fish/Wildlife Report No. TFW-WQ3-90-006.

Theurer, F.D.; Voos, K.A.; Miller, W.J. 1984. **Instream water temperature model**. Instream Flow Information Paper No. 16. Washington, DC: US Fish and Wildlife Service (FWS/OBS-84/15).

U.S. Environmental Protection Agency. 1999. **South Fork Eel River total maximum daily loads for sediment and temperature**. San Francisco, CA: U.S. EPA Region IX Water Division; 62 p.

Welsh, H.H., Jr.; Hodgson, G.R.; Harvey, B.C.; Roche, M.E. 2001. **Distribution of juvenile coho salmon in relation to water temperatures in tributaries of the Mattole River, California.** North American Journal of Fisheries Management 21: 464-470.

Whitney, S. 1985. **Western Forests**. National Audubon Society Nature Guides, New York: Chanticleer Press, Inc.

Zwieniecki, M.A.; Newton, M. 1999. **Influence of streamside cover and stream features on temperature trends in forested streams in western Oregon**. Western Journal of Applied Forestry 14(2): 106-113.

Ecosystem Management Decision Support (EMDS) Applied to Watershed Assessment on California's North Coast [1]

Rich Walker,[2] Chris Keithley,[3] Russ Henly,[2] Scott Downie,[4] and Steve Cannata[3]

Abstract

In 2001, the state of California initiated the North Coast Watershed Assessment Program (2003a) to assemble information on the status of coastal watersheds that have historically supported anadromous fish. The five-agency consortium explored the use of Ecosystem Management Decision Support (EMDS) (Reynolds and others 1996) as a means to help assess overall watershed and in-stream conditions for fish. EMDS is expert system software developed by the USDA Forest Service for similar efforts with the Northwest Forest Plan (2000). NCWAP developed models to help assess key watershed characteristics that contribute to shaping channel morphology and to evaluate the present stream habitat conditions in terms of suitability for anadromous salmonids. The stream condition model uses data collected during DFG stream surveys to evaluate the present stream habitat conditions for migrating, spawning and rearing anadromous fish. Factors evaluated by the model include percent of reach in moderately deep pools, pool shelter complexity, streamside canopy density, and spawning gravel embeddedness. We also developed a model addressing indirect terrestrial influences on anadromous fish in a watershed. The Potential Sediment Production Model estimates the impacts of both natural background and human-related effects on in-stream sediment delivery. NCWAP scientists learned several important lessons from using EMDS. Critical aspects of EMDS include the hierarchical structure of the model, the selection of the operators at the nodes of the networks, and the selection of breakpoints used in evaluating specific environmental data.

Key words: California salmon, ecosystem decision support, watershed assessment

Introduction

The North Coast Watershed Assessment Program (2003a) was a California state government program from 2001 to 2003. Five state agencies participated in its mandate to assess watershed conditions for salmonids on the north coast. The Departments of Fish and Game, Forestry and Fire Protection, California Geological Survey, Water Resources and the state Environmental Protection Agency (EPA) brought their expertise together to gather existing environmental data and to evaluate watersheds.

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Fire and Resource Assessment Program, California Department of Forestry and Fire Protection, Northern Region Headquarters, 135 Ridgway Ave, Santa Rosa, CA 95401. email: rich.walker@fire.ca.gov

[3] Fire and Resource Assessment Program, California Department of Forestry and Fire Protection, 1300 U Street, Sacramento, CA 95818. email: chris.keithley@fire.ca.gov, russ.henly@fire.ca.gov

[4] North Coast Watershed Improvement Center, California Department of Fish and Game, 1455 Sandy Prairie Court, Fortuna, CA 95540. email: scannata@dfg.ca.gov, sdownie@dfg.ca.gov

Through basin wide assessments of salmonid habitat, and identifying the likely causes of habitat impairment, the program's thrust was to assist prioritization of both in-stream and upland area watershed restoration and recovery efforts. The Ecosystem Management Decision Support (Reynolds 2003) expert system software, developed in connection with the Northwest Forest Plan, was selected by NCWAP to aid in the watershed evaluations. This paper reports on the successes and challenges encountered during program's use of EMDS.

Background

EMDS is a type of 'expert system' software that uses 'Linguistic' models and a formal branch of mathematics termed 'fuzzy logic' to evaluate data against specified criteria (for details see Reynolds 2003). The software can produce a synthesis from a variety of environmental data, but requires that a custom hierarchical model (termed 'knowledge base') be built to simulate the particulars of the ecosystem functioning. Models utilize data stored in a Geographic Information System (GIS – ArcView™) to perform the assessments of watershed conditions and facilitate rendering the results into maps and tables. The results can then be interpreted by professionals and the public, in terms of their implications for ecosystem management. Clearly, the results of the EMDS models will be strongly affected by the adequacy of the knowledge base in representing real ecosystem functioning, and the completeness and accuracy of the data fed into the models.

Methods

As a starting point, NCWAP referred to an EMDS knowledge base model developed by the USDA Forest Service to evaluate watershed conditions for salmon in coastal Oregon (Reeves and others 2000). The NCWAP team then constructed two knowledge base networks reflecting the best available scientific studies and information on how various environmental factors combine to affect anadromous fish in California's north coast watersheds. These networks can be graphed to resemble branching tree-like flow charts, to show the logic and types of data used in the assessment:

- The Stream Reach model *(fig. 1)* addresses conditions for salmon at the stream reach scale and is largely based on habitat data collected under the Department of Fish and Game's stream survey protocols;
- The Potential Sediment Production model *(fig. 2)*, evaluates the magnitudes of the various sediment sources in the basin according to whether they are natural or management related.

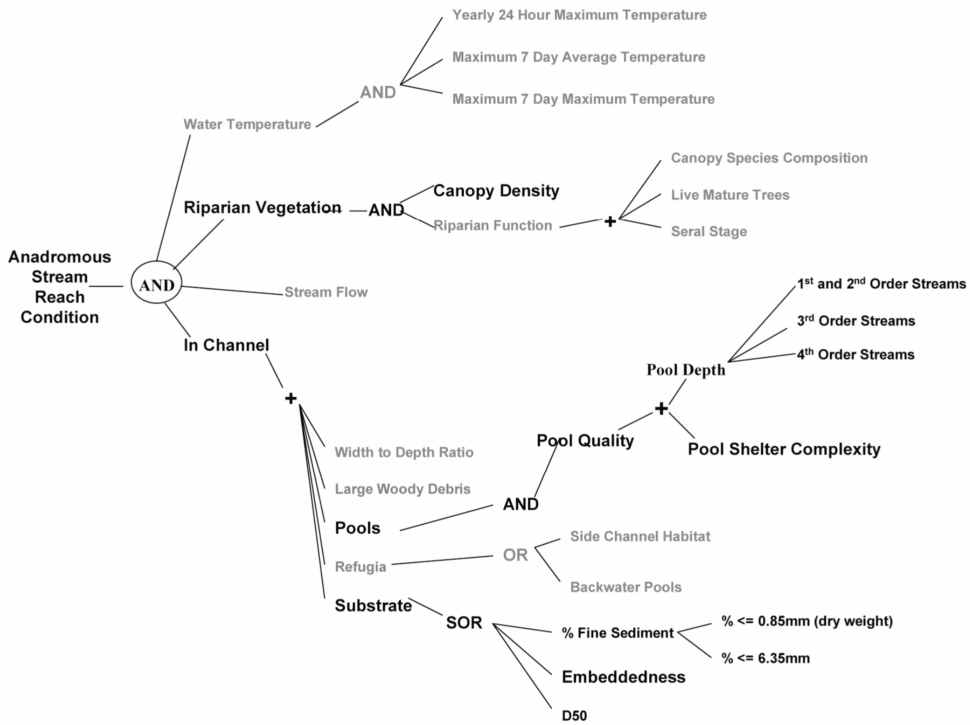


Figure 1—NCWAP EMDS anadromous reach condition model. Objects in black were populated with data and evaluated for NCWAP reports, objects in grey were considered important characteristics, but data or reference curves were not developed for the assessment. EMDS evaluates these missing data to 0 (or uncertain), which contributes to a conservative estimate of the overall stream condition.

In creating the EMDS models listed above, NCWAP scientists used what is termed a 'top-down' approach. This approach is perhaps best explained by way of example. The NCWAP Stream Reach Condition model began with the proposition: *The overall condition of the stream reach is suitable for maintaining healthy populations of native coho and Chinook salmon, and steelhead trout.* A knowledge base (network) model was then designed to evaluate the truth of that proposition, based upon data from each stream reach. The model design and contents reflect the hypotheses of the NCWAP scientists. The goal was to start with a simple model structure that addressed the proposition through evaluation of key stream habitat characteristics. Refinements, adaptations, and the addition of new habitat elements would occur as needed and as data collection protocols were developed to populate and expand the knowledge base architecture. Broad-based reference curves were developed by consensus of the science team for use in multi-basin assessments, but were not intended as thresholds or targets. They formed generally accepted points of agreement about critical habitat functions for salmonid production.

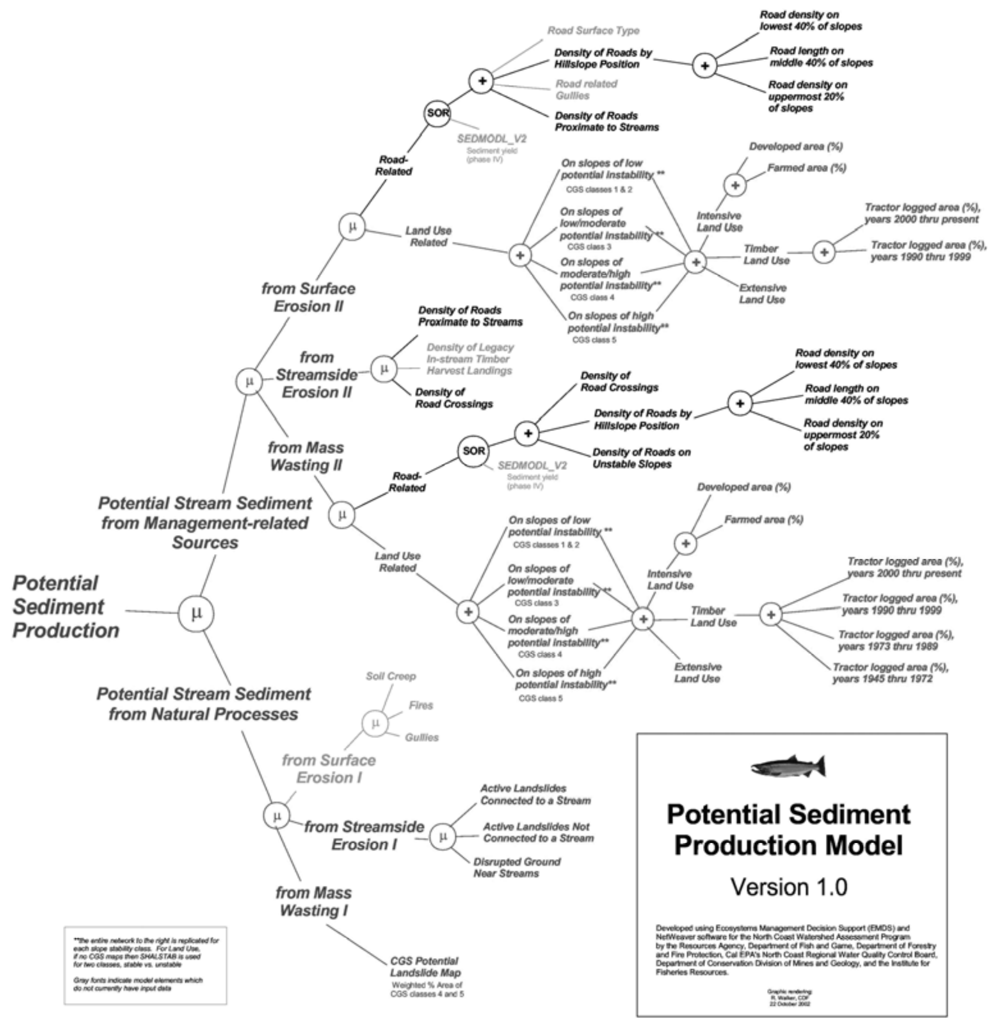


Figure 2—NCWAP EMDS potential sediment production model.

To evaluate stream reach conditions for salmonids, the conceptual model used data on several environmental factors. The first branching of the knowledge base network *(fig. 1)* shows that information on environmental conditions (stream flow, riparian vegetation, water temperature, and in-channel stream characteristics) are all used as inputs in the stream reach condition model.

In turn, each of the four branches is progressively broken into more basic data components that contribute to it. Each object in the branch is evaluated, synthesized according to the knowledge base structure, and passed forward towards final decision node. The process is repeated until the knowledge base network incorporates all information that experts believe to be important (independent of data availability) to the stream reach evaluation.

EMDS models assess the degree of truth (or falsehood) of each evaluated proposition. Simple reference curves use “fuzzy logic” to determine its degree of truth/falsehood, according to the data’s implications for salmon. *Figure 3* shows an example reference curve for the proposition *the stream temperature is suitable for salmonids.* The horizontal axis shows temperature in degrees Fahrenheit, while the vertical is labeled Truth Value and ranges from –1 to +1. The line shows what are fully unsuitable temperatures (–1), fully suitable temperatures (+1) and those that are on the continuum in-between (>–1 and <+1). A zero value means that the proposition cannot be evaluated based upon the data available. Breakpoints (where the slope of the reference curve changes) in the *figure 3* example occur at 45, 50, 60 and 68 °F. For the Stream Reach model, NCWAP fisheries biologists determined these temperatures by a review of the scientific literature and empirical studies (Armour 1991, Hines and Ambrose 2000, Klamt and others 2000, Welch and others 2000). In this way, similar numeric reference curves were developed for other propositions evaluated in the NCWAP Stream Reach Condition Model.

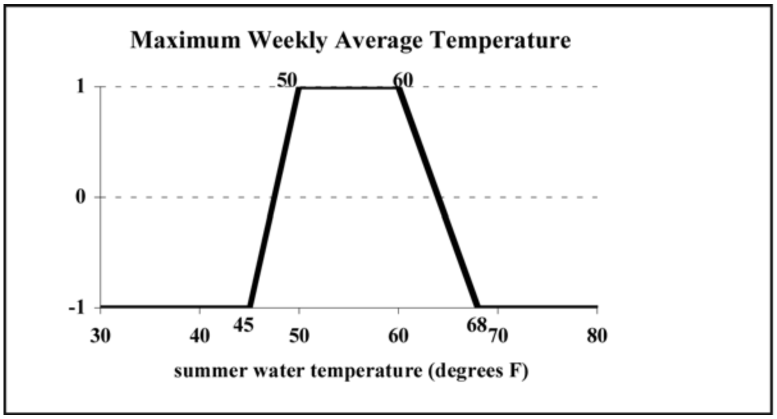


Figure 3—Generalized EMDS reference curve for summer water temperature measured by maximum weekly average temperatures (MWAT). EMDS uses this type of reference curve in conjunction with data specific to a stream reach. This example curve evaluates the proposition that the stream's water temperature is suitable for salmonids. Break points can be adapted for specific species, life stage, or season of the year.

For NCWAP parameters relating to upland geology and management activities, little scientific literature was available to assist in determining breakpoints to evaluate the watersheds. As a result, we computed the mean and standard deviation for all planning watersheds in a basin, then selected breakpoints to rank each planning watershed for that parameter, using a simple linear approximation of the standardized cumulative distribution function [with the 10th and 90th percentiles serving as the low and high breakpoints *(fig. 4)*]. (Is there some reason these breaks were used?) (Purely empirical—they where the linear approximation of the CDF where intercepts y = 0 and y = 1) The relative rankings were valid only within the basin and did not serve as an absolute measure of the suitability of a given planning watershed for salmon spawning and rearing. However, they did provide an indication of relative conditions for fish within the basin with regard to specific environmental factors.

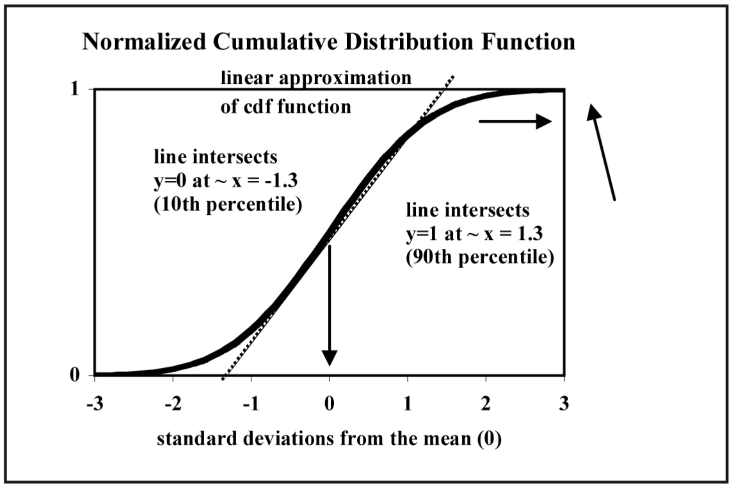


Figure 4—Normalized cumulative distribution function, and EMDS empirical breakpoints. Using the 10th and 90th percentiles as breakpoints (as with Land Use) is a linear approximation of the central part of the normalized cumulative distribution function.

Results

For each of the three initial NCWAP watersheds (Gualala, Mattole and Redwood Creek), the EMDS models produced a number of tables and maps that were examined by the team of agency experts for that basin. For example, *figure 5* shows Stream Reach model results for the middle subbasin of Redwood Creek for four factors—embeddedness, canopy density, pool shelter and pool depth. The map shows each reach of the subbasin rated for its suitability for salmonids, from fully suitable to fully unsuitable. (Is there a map which integrates all of these? Unfortunately, no.) *Figure 6* shows the results of the Natural Mass Wasting limb of the upland-oriented Potential Sediment Production model for the Gualala River basin. Similar maps were created of all evaluated EMDS parameters.

This example illustrates the graphical outputs of an EMDS run. This demonstration graphically portrays the relative amounts of potential sediment production in the Mattole Basin that comes from natural sources.

Opinions varied among the agency scientists working in the program as to whether the EMDS models performed well in synthesizing the existing data and coming up with accurate and useful results. Overall, the Stream Reach Condition Model *(fig. 1)* was viewed as a generally accurate tool to assess individual habitat elements and it provided an accurate assessment of overall stream condition. In some cases, the EMDS results required further interpretation and explanation because of data limitations in the knowledge base structure. The model would be improved if it included data input for critical habitat elements such as water temperature that were not populated with data in the knowledge base. (Due to disagreements among NCWAP scientists on the adequacy of water temperature samples, and extrapolating them to stream reaches, water temperature proved to be a difficult parameter to accurately display on EMDS spatial map outputs. On the other hand it serves here as an easily understood example for illustrating how NCWAP EMDS worked).

The potential sediment model *(fig. 2)* relied upon data generated from maps and models that were not necessarily field verified. There also was a lack of scientific literature and data sources to develop representative reference curves. This necessitated the use of a more empirically-based approach to defining reference curves, using cumulative distribution functions based upon the data. Where data quality was high, the models sometimes yielded insights into the factors most affecting a given planning watershed. In other cases the results were not regarded as accurate.

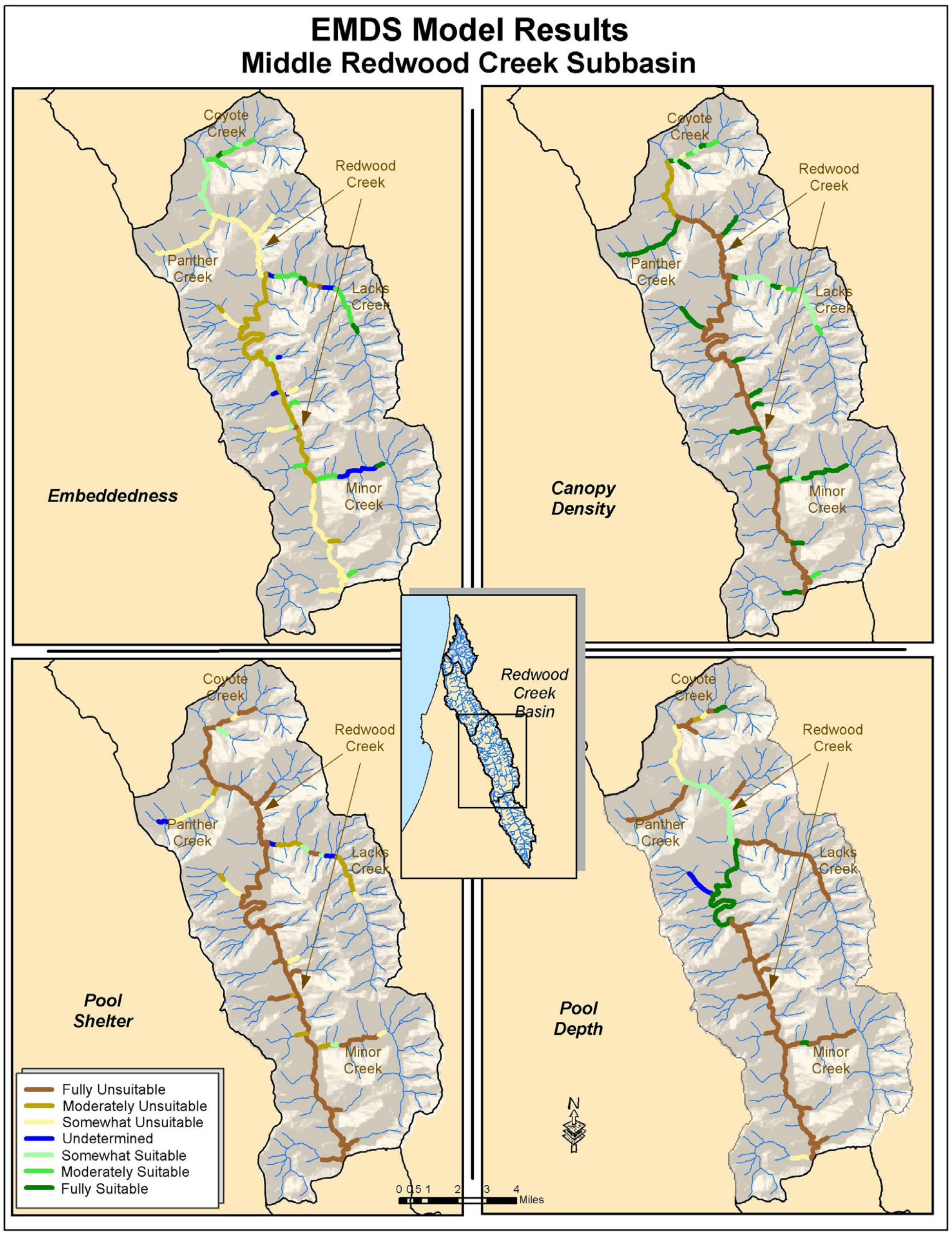


Figure 5—Map of results of part of the Stream Reach model for the middle subbasin of Redwood Creek.

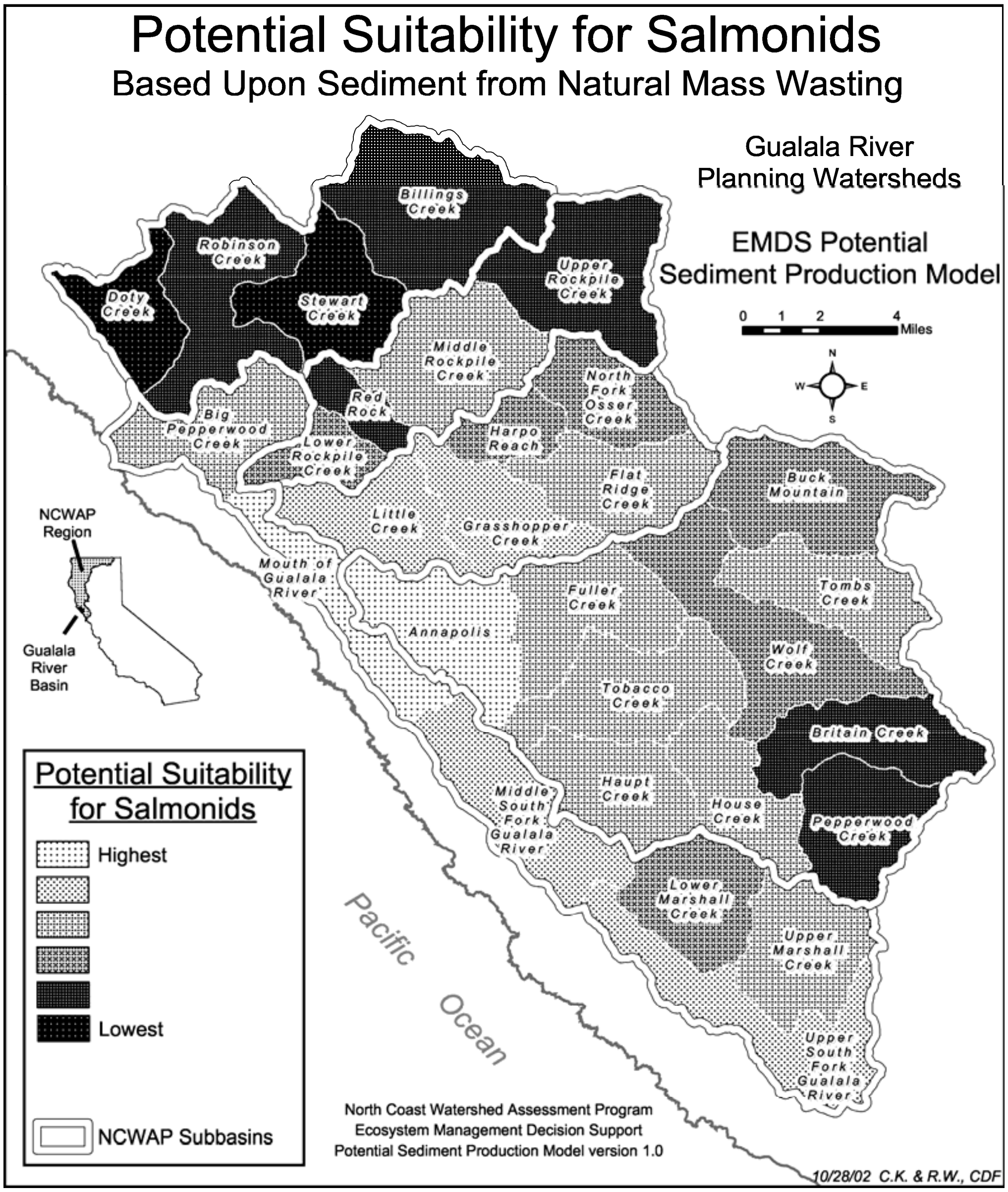


Figure 6—EMDS graphical output from Potential Sediment Production Model.

Discussion

It was a challenge to validate the two complex ‘expert system’ models. Data inconsistency (for example, missing, out-of-date, sparsely sampled) was common in the basins, affecting the model results in ways that were difficult to quantify but certainly deleterious. In some model branches, no data was available for the EMDS model. The missing data tends to move higher-level model evaluations towards the uncertain range (in other words, 0 values), thus making overall assessments conservative. EMDS provides a tool to assess the relative value of missing data, based upon the model architecture. This can help to drive data needs and priorities in future sampling.

Several core considerations emerged in the course of using EMDS in NCWAP. These included:

- Model propositions: Semantics used must be precise to define criteria
- Model architecture: Is the model an accurate reflection of the 'real world'?
- Breakpoints: Definition can be difficult, given dearth of scientific literature on many watershed characteristics, especially current and historic land use
- Parameter weightings: Relative weightings affect the influence of parameters
- Calibration and validation: Difficult to do with these non-standard models
- Data sampling: Is it representative of the entire reporting unit (stream or watershed)?

As EMDS does not produce statistical or deterministic model outputs, it was difficult to establish criteria by which to evaluate the model results. Scientific discipline would suggest that we take a sample of planning watersheds and compare the EMDS results against the best data and criteria available. However, as EMDS is intended to emulate expert opinion, which is based not only on data but other intangibles, the latter test may be inappropriately setting the bar too high. Also, a fundamental question arose in NCWAP as to whether the models' proper role was to simply confirm expert opinion (in other words, what was already known), or to provide new insights into the watershed conditions.

Conclusions

EMDS served two main roles in NCWAP. First, the model results synthesized numerous factors, and provided insights into individual watershed characteristics at several spatial scales. The stream reach condition model identified specific reaches where habitat factors were currently in good or poor condition. This can help identify the type of instream or upslope management needed to improve habitat conditions for salmonids. The sediment production model assessed landuse factors at the planning watershed scale that have already accelerated erosional process or may do so in the future (in other words, risk assessment). These data help to guide land management decisions surrounding watershed ecosystem issues. Together the models help prioritize where and when management actions should occur.

Additionally, EMDS provided a de facto second 'result'. In requiring explicit model logical structure and data 'feeds', the process of developing and reviewing the two models facilitated a number of in-depth and useful discussions of precisely how these watersheds function, and the data needed to make such evaluations. The first relates to the more standard assessment of the accuracy and utility of the numbers and watershed assessment maps produced by the models, while the second looks at the utility of EMDS from the standpoint of the program as a whole.

While EMDS-based syntheses can be important tools for watershed assessment, they cannot by themselves yield a course of action for restoration and land management. Any EMDS results require interpretation, and how they are employed depends upon other social and economic concerns. In addition to the accuracy of the EMDS model, the currency and completeness of the data available for a stream or watershed will strongly influence the confidence in the results. Where possible, the EMDS model should be validated using sensitivity analysis, independent data, expert opinion, and other information. One disadvantage of linguistically based models such as EMDS was that they do not provide results with readily quantifiable levels of error. Users of this tool need to be aware of this issue.

References

Armour, C. 1991. **Guidance for evaluating and recommending temperature regimes to protect fish.** Biological Report 90 (22). U.S. Fish and Wildlife Service. Fort Collins, CO 13p. *Available at* http://www.krisweb.com/biblio/gen_usfws_armour_1991.pdf

Hines, D.; Ambrose, J. 2000 (Draft). **Evaluation of stream temperatures based on observations of juvenile coho salmon in Northern California streams.** Campbell Timber Management, Inc., Fort Bragg, CA, and National Marine Fisheries Service, Santa Rosa, CA. *Available at* http://www.krisweb.com/biblio/gen_afs_hinesetal_xxxx.pdf

Klamt, R.; Otis, P.; Seymour, G.; Blatt, F. 2000. **Review of Russian River water quality objectives for protection of salmonids species listed under the Federal Endangered Species Act.** Regional Water Quality Control Board, North Coast Region, 5550 Skylane Boulevard, Suite A, Santa Rosa, CA 95403. *Available at* http://www.krisweb.com/krisrussian/krisdb/html/krisweb/biblio/russian_scwa_ncrwqcb_2000.pdf

North Coast Watershed Assessment Program. 2003a. California Resources Agency, Sacramento, CA. *Available at* http://www.ncwap.ca.gov

North Coast Watershed Assessment Program. 2003b. **EMDS Appendix.** California Resources Agency, Sacramento, CA.

USDA Forest Service, Pacific Northwest Region. 2000. Northwest Forest Plan. *Available at* http://www.fs.fed.us/r6/nwfp.htm

Reeves, G.; Hohler, D.; Larsen, D.; Busch, D.; Kratz, K.; Reynolds, K.; Stein, K.; Atzet, T.; Hays, P.; Tehan, M. 2000. **Aquatic and riparian effectiveness monitoring plan for the Northwest Forest Plan.** *See* http://www.reo.gov/monitoring/watershed/cover-aremp-report.htm

Reynolds, K. 2003. EMDS. Pacific Northwest Research Station, Forest Service, U.S. Department of Agriculture, Corvallis, OR. *Available at* http://www.fsl.orst.edu/emds/

Reynolds, K.; Cunningham, C.; Bednar, L.; Saunders, M.; Foster, M.; Olson, R.; Schmoldt, D.; Latham, D.; Miller, B.; Steffenson, J. 1996. **A knowledge-based information management system for watershed analysis in the Pacific Northwest US.** Artificial Intelligence Applications 10: 9-22.

Welsh, Jr., H.H.; Hodgson, G.R.; Harvey, B.C.; Roche, M. E. 2001. **Distribution of juvenile coho salmon in relation to water temperatures in tributaries of the Mattole River, California.** North American Journal of Fisheries Management 21: 464-470.

The Effects of Large Wood on Stream Channel Morphology on Three Low-Gradient Stream Reaches in the Coastal Redwood Region[1]

Scott Carroll[2] and E. George Robison[3]

Abstract

Several studies have shown that large wood has a prominent role in habitat quality, however there is little research on the role of wood on pool characteristics and other habitat components in low gradient streams (channel slopes less than one percent). Longitudinal profiles are used to analyze general residual pool characteristics of three approximately 1000-meter stream reaches with similar stream and watershed characteristics and vastly different large wood loadings (759 vs. 329 vs. 39 m^3/ha). This study devises a new classification methodology to subdivide long pools, due to different formation mechanisms and physical barriers within pools. Shallow long runs at the start or end of pools are also removed because the do not represent pool habitat. Higher large wood loading is associated with decreased individual pool lengths, maximum depths, and longitudinal residual pool areas. Greater wood loads however increase reach percent channel in pools and pool frequency. Cumulative reach longitudinal residual pool area per 100 meters and mean thalweg depths did not vary between the study reaches. Bedrock pools had greater pool lengths, maximum depths, and longitudinal residual pool areas than pools formed by large wood. Bedrock pools also had more complex bed topography than large wood pools, but had no difference in pool edge rugosity.

Key words: large wood, salmonid habitat, stream channel morphology

Introduction

The twentieth century has seen a dramatic increase in the extraction of natural resources in the western United States. Today, the impacts of these disturbances can be seen in changing habitats and decreases in species diversity and populations. The Pacific Northwest has received much attention due to two valuable resources, timber and salmonids. The role of large, downed, instream wood has been determined to play a key role in the success of salmonids (Fausch and Northcote 1992, McMahon and Hartman 1989, Quinn and Peterson 1996). Large wood (LW) is critical in

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Humboldt State University, 1706 Grizzly Mountain Drive, South Lake Tahoe, CA 96150. email: shaggycar@yahoo.com

[3] Humboldt State University, Department of Forestry and Watershed Management, 1 Harpst Street, Arcata, CA 95503. email: egr2@humboldt.edu

creating habitat complexity, and its reduction has caused stream simplification and a decrease in habitat quality (Fausch and Northcote 1992, Quinn and Peterson 1996).

Low gradient streams (less than one percent channel slope) are an essential part of salmonid habitat, yet the effect of large wood has not been studied as extensively for this particular channel type. The percent of streams in pools and the character of pools is an important component of salmonid habitat (McMahon and Hartman 1989, Moore and Gregory 1988). Yet in low gradient streams of small to moderate size (in other words, less than one percent gradient and less than approximately 20 m in bankfull width) the channel is largely made up of pools and slack water habitat and the pools can extend for dozens of meters (Robison 1988). This channel type is distinct from a classic riffle/pool sequence or step/pool systems (Leopold and others 1964, Montgomery and Buffington 1997) that have received the bulk of large wood and channel morphological study. These pool features often have multiple influences creating them over their long distances as well as distinct breaks in character that are not accounted for with traditional channel unit systems because the unit is simply called a “pool.” A study that focuses on the nuances of these types of systems is long overdue. This study represents a case study of these types of low gradient reaches for three long stream reaches that have nearly identical channel and watershed characteristics (in other words, drainage area, bankfull width, gradient, levels of confinement, climate, forest type, sediment input rates, and geology). The one thing that separates them is the degree of large wood loading in these reaches. This difference in wood loading allows for some observations to be made between these reaches in terms of stream morphology forming processes and the role wood does and does not play in forming and modifying pools. The research investigates habitat at the reach scale and the habitat unit scale, specifically pools, in order to determine complexity at various scales.

This study uses clear, repeatable, non-subjective methods in measuring physical stream parameters in order to quantify complexity. Pools are precisely defined from detailed survey data using residual depths (Bathhurst 1981, Lisle 1987, Robison 1998, Robison and Kaufmann 1994) and by scaling to channel size.

Longitudinal profiles in conjunction with three-dimensional surface surveys were used to measure pool parameters to isolate the effect of LW at both the reach and pool scale. The hypothesis at the reach scale is that LW increases the density of pools, percent channel in pools, longitudinal residual pool area (LRPA) per 100 meters, mean thalweg depth, thalweg depth variation, and percent pools formed by LW. The hypothesis at the individual pool scale is that pool parameters such as length, maximum depth, and longitudinal residual pool area decrease with higher LW loading. Additionally, at the individual pool scale, bedrock created pools tend to be longer, deeper, and larger pools than LW, but LW pools have more complex bed topography and pool edge rugosity, due to the relationship between LW and complexity.

Methods

The Mill and Prairie Creek watersheds are located on the coast in northwest California near the northern limit of coastal redwoods. Both watersheds are redwood and Douglas-fir forests with Prairie Creek almost all old growth and Mill Creek almost completely harvested at least once. Two study reaches are located in the Mill

Creek watershed. The East Fork and West Branch study reaches have drainage areas of 37 km^2 and 24 km^2. A study reach in Prairie Creek has a drainage area of 31 km^2. Bankfull widths average between 17 and 21 m on all study reaches. All three study reaches have low gradients of approximately 0.6 percent. All study reaches have unconfined channels with prominent floodplains. Valley widths on average are at least four times bankfull width. All study reaches contain bedrock in the channel. The Prairie Creek bedrock consists mostly of competent conglomerated fluvial deposits and East Fork and West Branch are mostly sandstone. Despite the difference in bedrock type, both bedrock pool types were classified the same because they create similar pools.

Data collection consisted of LW surveys, longitudinal profiles, and topographic mapping of pools. LW was defined as any piece of wood larger than 0.1 meters in diameter and two meters long. In some cases LW jams prevented the counting of individual pieces, and a total volume was obtained by measuring jam dimensions. For this study LW data is based only on pieces within the bankfull channel, as they were determined to be most important in affecting channel morphology.

Longitudinal profiles were surveyed with points taken an average of every four meters. Residual characteristics of the stream channel (Bathhurst 1981, Lisle 1987) were determined for each of the long profiles. The reaches were divided into pools and riffles based on the elevation of the downstream riffle crests (Bathhurst 1981, Lisle 1987). Residual depth removes the effect of discharge, but in low gradient streams, pools with separate formation factors and habitat features often combine to form one large pool. Simple, repeatable, guidelines were created to better characterize pools in low gradient streams. Pools were defined by the channel size (Buffington and others 2002, Montgomery and others 1995). Using Montgomery and others (1995) a minimum residual depth for a pool was determined to be 25 percent of mean bankfull depth of the reach. The minimum pool length was defined as half bankfull width. These pool dimensions represented significant habitat features and removed small, insignificant, localized scour and steps in riffles. Minimum pool depth was determined to be 0.25 m (mean bankfull depth from cross sections was approximately one meter on all reaches) and minimum pool length varied from 8.5 to 10.5 m depending on the reach.

Once pools were defined there were two factors that led to additional manipulations to pool dimensions. First, due to the low gradient of the study reaches there were often long areas of flatwater that encompassed multiple pools. These compound pools were subdivided because there were different formation factors for different areas of the pool and physical barriers within the larger pool that yielded separate smaller pools. A compound pool was divided if the depth at a high point between two pools was less than 25 percent of the maximum depth of the deeper pool and at least half the depth of the shallower pool. Secondly, shallow water at the tail or start of a pool was removed from pool dimensions if they were at least the minimum pool length and depth for that reach. These pool definitions were needed to accurately quantify pool habitat in these low gradient reaches.

The final data collection phase involved detailed mapping of pool beds using a Total Station. Eleven pools were mapped in detail by measuring point data in three dimensions and using a Geographic Information System (GIS) to create surfaces using triangulated irregular networks (TIN). Six of the pools were LW formed pools and five were bedrock pools. Pools were chosen based on formation type and ability to survey. Cross sections were placed every ~1/15 channel width and points were spaced every ~1/50 channel widths (Kail 2002). The mapping was done to compare complexity of bed topography and pool edge rugosity in LW and bedrock pools.

Results

Reach characteristics for each of the sites can be seen in *table 1*. LW loadings clearly show the old growth Prairie Creek with the most, the managed West Branch with half the old growth amount, and the managed East Fork with almost none. Although piece dimensions were similar, due to a few very large pieces on Prairie Creek the mean volume per piece is greater than twice that of the managed streams.

Table 1—*Reach, large wood, and pool characteristics of Prairie Creek, West Branch Mill Creek, and East Fork Mill Creek, 2003.*

Variable	Prairie Creek	West Branch Mill Creek	East Fork Mill Creek
Reach length (m)	1098	1051	1408
Large Wood loading (m^3/ha)	759	329	39
Number of pieces	263	244	66
Pieces/100m	24	23	5
Mean piece diameter (m)	0.6	0.4	0.4
Mean piece length (m)	7.3	6.3	7.3
Mean piece volume (m^3)	5.2	1.9	1.5
Percent channel in pools (%)	64	64	50
Number of pools	32	27	24
Pool spacing (bankfull widths)	2.0	1.8	3.2
Longitudinal residual pool area / 100 meters (m^2/100m)	27.1	30.8	28.6
Mean reach thalweg depth every 2 meters (m)	0.28	0.33	0.28
Reach depth coefficient of variation	1.07	0.97	1.37
Mean pool length (m)	22	25	29
Mean pool longitudinal residual pool area (m^2)	9.2	12.0	16.7
Mean maximum pool depth (m)	0.8	0.8	0.9
Maximum pool depth (m)	1.3	1.4	2.1
Mean pool depth (m)	0.4	0.5	0.5

Higher LW loadings led to closer pools, greater percent channel in pools, but had little effect on mean thalweg depth and LRPA per 100 m. Pool spacing, measured in bankfull widths (BFW), was almost the same on Prairie Creek (2.0 BFW) and West Branch (1.8 BFW) while greater on East Fork (3.2 BFW). Higher LW loadings increased the percent channel in pools (East Fork had just under 50 percent while West Branch and Prairie Creek had 64 percent). LW did not affect the LRPA per 100 m, as all reaches were almost the same. If LRPA is an indicator of pool size, then East Fork has longer, deeper, and larger pools than the wooded reaches. This results in East Fork having an equal amount of LRPA per 100 m despite the fact there are fewer pools. Mean thalweg depths from measurements taken every two meters showed little difference. West Branch (0.33 m) was the deepest followed closely by Prairie Creek (0.28 m) and East Fork (0.28 m). Standard deviations and coefficient of variations indicate that East Fork has greater variation in thalweg depths than Prairie Creek and West Branch.

When reaches were analyzed by pool forming mechanism, a clear separation between wooded and non-wooded reaches was determined *(fig. 1)*. Prairie Creek (62 percent) and West Branch (59 percent) had a similar percentage of LW pools, while East Fork had 79 percent bedrock and only 13 percent LW. The results that West Branch and Prairie Creek are very similar in percent pool type and different from East Fork led analysis in two directions. Individual pool parameters were first compared between the three study reaches and then between formation types, bedrock and LW.

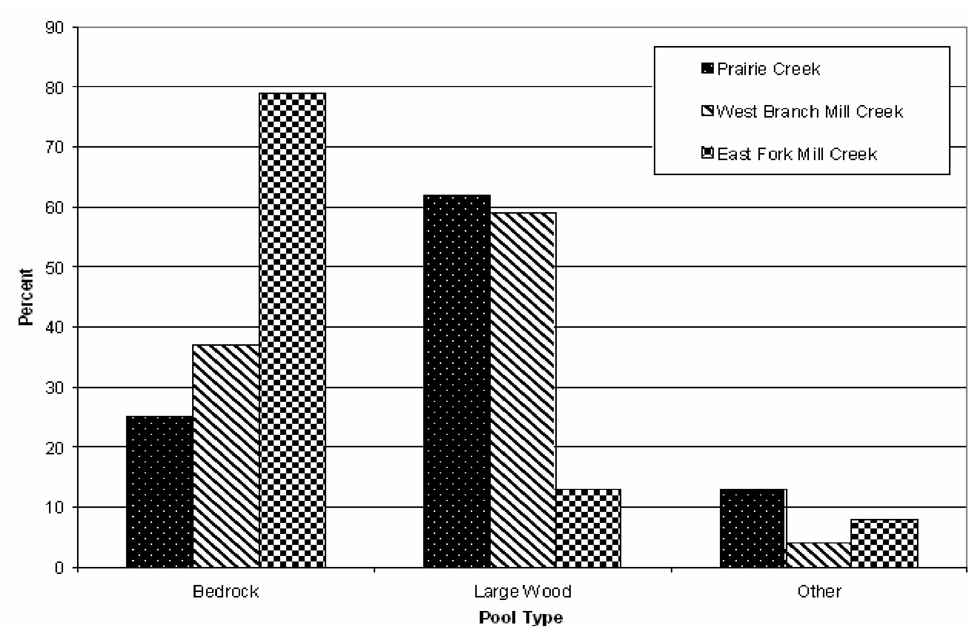


Figure 1—Distribution of pool types in Prairie Creek, West Branch Mill Creek, and East Fork Mill Creek, 2003.

All individual pool characteristics followed a similar pattern with East Fork generally having longer, deeper, and larger (based on LRPA) pools with West Branch in the middle and Prairie Creek with the shortest, shallowest, and smallest pools *(table 1)*. Frequency plots were created to show differences in pool dimensions between the reaches *(fig. 2, 3*, and *4)*. East Fork has a few pools at the smallest level in each category but as the percent increases East Fork has longer, deeper, and larger pools than the two reaches with LW. West Branch and Prairie Creek had similar pool dimension frequencies in all cases. When the smaller pools in East Fork were investigated it was observed that they were all from LW origin and that all the larger pools were bedrock. This led to analysis on the difference between bedrock and LW pools.

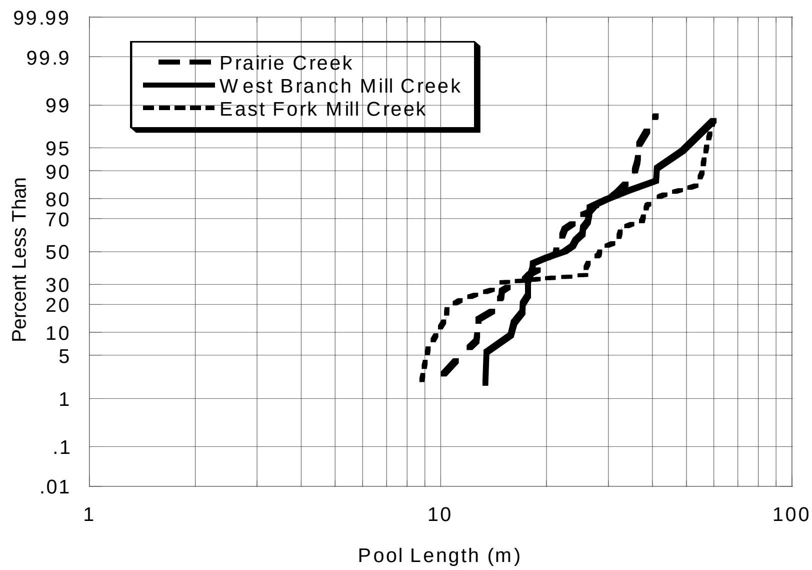


Figure 2—Frequency distribution of individual pool lengths in each study reach, 2003.

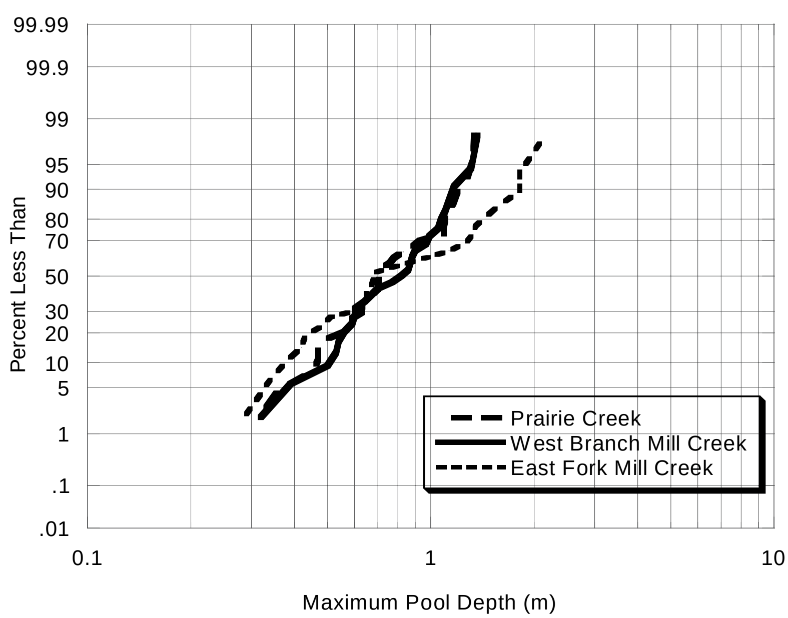


Figure 3—Frequency distribution plot of individual pool maximum depths in each study reach, 2003.

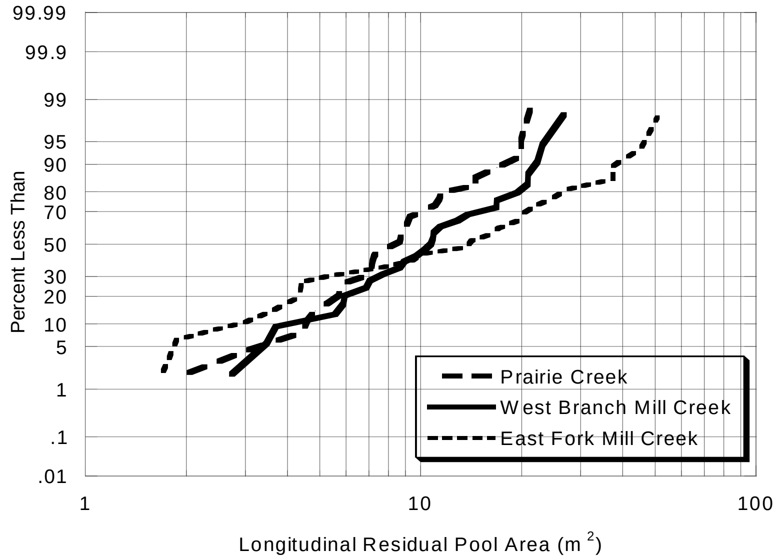


Figure 4—Frequency distribution plot of individual pool LRPA in each study reach, 2003.

Despite having different LW loadings Prairie Creek and West Branch have a similar distribution of pool types. A comparison of bedrock vs. LW pools was done to determine the differences between the pool types. Bedrock pools averaged approximately seven meters longer, almost twice the LRPA, and 0.25 deeper at its maximum depth than LW pools. These results led further into the topographic differences between the pool types. Complexity of both the bed and the edges of bedrock and LW pools were determined by mapping a subset of pools in each of the study reaches. Edge rugosity (Actual edge length / straight line edge length) was used to determine complexity of pool margins. There was no difference in mean edge rugosity (1.10) between bedrock and large wood pools. The second feature investigated was the effect of pool formation type on the bed topography. First, three-dimensional rugosity (bed surface area / pool planimetric surface area) was calculated on each of the pools. There was a difference between pool types as bedrock pools had consistently greater rugosity than large wood pools. Bedrock pools had at least a 10 percent increase in bed surface area compared to planimetric area. All large wood pools had less than a 10 percent increase in bed surface area except one. The second comparison to determine pool bed complexity was to compare the individual depths of each pool. The coefficient of variation of all individual depth values was used to remove the effect of different sized pools. Individual depths were used as a comparison because points were taken at a consistent spacing throughout the pool. A separation again occurred, with LW pools having lower coefficients of variation than bedrock pools. Both results indicate that bedrock pools have more complex bed topography than LW pools.

Discussion

Large wood affects stream channel morphology in low gradient coastal redwood streams at both the reach and individual pool scale. The results indicate that despite Prairie Creek and West Branch having different LW loadings they form a similar physical environment (pool spacing and percent channel in pools). A potential reason why there is not much difference between the reaches is that much of the large wood volume in Prairie Creek consists of a few very large pieces that only utilize a small percentage of their volume to form a pool or effect habitat. The LW character was different in West Branch, where pieces were smaller, part of jams, and typically within the low flow channel. This distribution of LW volume in these two streams may explain the similarities.

Although some reach parameters were different with various LW loadings, some characteristics were the same. LRPA per 100 meters of stream for the three reaches was not significantly different, indicating that despite East Fork having almost no LW it has a similar amount of pool area as the other reaches. Pool area was assumed to be a critical role of LW, it was assumed that increases in LW would create more scour. These results indicate LW does not affect reach pool area in these low gradient streams, rather LW affects the distribution of pool area. These results concur with Beechie and Sibley (1997) and Robison (1998) who determined that large wood in low gradient streams had less effect on pool area than in moderate and high gradient streams. In moderate and higher gradient streams (in other words, streams with gradients greater than one percent) or in larger streams with low gradients, large wood has a more active role in pool formation (Montgomery and others 1995, Robison 1998). It was also expected LW would increase mean thalweg depth, but it was almost the same for all reaches and East Fork actually had a higher thalweg depth variation. Two important results came from the reach investigation; first, LW created a clear difference between wooded (Prairie Creek and West Branch) and non-wooded reaches (East Fork) in terms of characteristics such as pool length and spacing and pool creating mechanism. Secondly, the Prairie Creek and West Branch study reaches had almost the same physical conditions, despite Prairie Creek having twice the LW loading. The similarities can be attributed to the fact West Branch had enough LW to create a similar distribution of pool types by formation mechanism to Prairie Creek. Bedrock was prominent in all the channels but when LW loading was minimal (East Fork) pool types dramatically shifted from favoring large wood to bedrock pools. The reach level results led to further investigation of individual pool parameters by reach and formation mechanism.

The hypothesis, at the individual pool scale, that higher LW loads yield shorter, shallower, and smaller pools in low gradient streams was confirmed as the pools in East Fork were significantly longer, deeper, and larger than the wooded Prairie Creek and West Branch. The difference in pool sizes is attributed to the type of pools in East Fork. The larger bedrock pools dominate in East Fork and smaller LW pools dominate in Prairie Creek and West Branch. The similar percentages of pool types found on West Branch and Prairie Creek may explain why their channel morphologies are similar. The final hypothesis was that LW pools should be more complex than bedrock pools due to the fact LW is associated with habitat complexity. When pool edge rugosity was compared there was no clear difference between the pool types. This lack of difference in pool edge rugosity may not express the margin complexity LW provides. Wooded reaches (Prairie Creek and West Branch) had more off-channel habitat than East Fork. The presence of back channels is critical in providing velocity refugia for fish during high winter flows (Bell 2001, Moore and Gregory 1988). A measure of bank rugosity rather than pool rugosity may have better shown the effect LW has on creating complex margin habitat. When pool bed surfaces were compared, bedrock pools had greater three-dimensional rugosity. Bedrock pools also had greater coefficients of variation of evenly spaced individual depths. This analysis indicates that in low gradient streams LW does not provide more complex bed topography, although it may create quality, complex habitat in other ways. LW can provide cover for fish in pools by creating shadows, covering the surface, velocity refugia, and three-dimensional partitions within the water column (Harmon and others 1986). LW creates habitat partitions at the reach scale by increasing pool distribution and increasing habitat unit diversity, while at the pool scale LW partitions habitat by creating physical dividers within the water column.

Conclusion

Management has typically revolved around setting target values of certain stream features such as LW amounts, pool dimensions, and fish populations. The problem is that quality habitat is described as complex, which cannot always be fully determined with simple measurements or surveys. This research suggests that the dimension of pools or the amount of LW in a stream may not reflect habitat quality in low gradient streams, rather the distribution and organization of LW, the cover complexity, and the distribution of pools may be most critical. This research indicates that the stream (East Fork) with the least LW has deeper, longer, and larger pools and greater thalweg depth variation. Additionally, the majority of the pools in East Fork are of bedrock origin, which has been determined to have more complex bed topography. Complexity is more than simply measured channel characteristics. LW does not create a complex environment in low gradient streams by simply creating pools and space for fish but rather it performs a number of roles throughout the length of streams.

This research has quantified stream reaches with easy to measure, non-subjective, repeatable procedures. Pools have been defined clearly, with a specific new methodology to deal with the low gradient of the study reaches. Watershed and stream characteristics were limited by choosing study sites with similar drainage area, bankfull width, gradient, levels of confinement, climate, forest type, sediment input rates, and geology. Long reaches were utilized to capture all variability in LW loading. To further this research, additional studies such as the role of LW and off channel habitat quality, the relationship between substrate and LW, velocity and LW interactions, and investigations into fish production and LW need to be done. Research also needs to be expanded from a case study to include more samples to develop statistically based results. With respect to management, this research shows that single target values might not be the best method to determine habitat quality in low gradient streams. It may not be possible to predict habitat quality by using single results such as individual pool parameters, reach characteristics, or LW loadings, rather all factors must be considered when quantifying stream habitat. The complexity that LW provides in low gradient streams is varied and diverse, and many aspects are difficult to measure and quantify. The best method involves utilizing multiple scales and taking a non-subjective, repeatable approach that encompasses the many different roles LW plays in creating complex salmonid habitat.

References

Bathhurst, J.C. 1981. **Discussion of “bar resistance of gravel-bed streams.”** Journal of the Hydraulics Division, American Society of Civil Engineers 104: 1587-1603.

Beechie, T.J.; Sibley, T.H. 1997. **Relationship between channel characteristics, woody debris, and fish habitat in northwestern Washington streams.** Transactions of the American Fisheries Society 126: 217-229.

Bell, E. 2001. **Survival, growth, and movement of juvenile coho salmon (*Oncoryhnchus kisutch*) over-wintering in alcoves, backwaters, and main channel pools in Prairie Creek, California.** Arcata, CA: Humboldt State University; 112 p. Masters thesis.

Buffington, J.M.; Lisle, T.E.; Woodsmith, R.D.; Hilton, S. 2002. **Controls of the size and occurrence of pools in coarse-grained forest rivers.** River Research and Applications 18: 507-531.

Fausch, K.D.; Northcote, T.G. 1992. **Large woody debris and salmonid habitat in a small coastal British Columbia stream.** Canadian Journal of Fisheries and Aquatic Sciences 49: 682-693.

Harmon, M.E.; Franklin, J.F.; Swanson, F.J.; Sollins, P.; Gregory, S.V.; Lattin, J.D.; Anderson, N.H.; Cline, S.P.; Aumen, N.G.; Sedell, J.R.; Lienkaemper, G.W.; Cromack, K.; Cummins, K.W. 1986. **Ecology of coarse woody debris in temperate ecosystems.** Advances in Ecological Research 15: 133-302.

Kail, J. 2002. **Influence of large woody debris on the morphology of six central European streams.** Geomorphology 51: 207-223.

Leopold, L.B.; Wolman, M.G.; Miller, J.P. 1964. **Fluvial processes in geomorphology**. San Francisco, CA: W.H. Freeman.

Lisle, T.E. 1987. **Using “residual depths” to monitor pool depth independent of discharge.** Res. Note PSW-394. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 4 p.

McMahon, T.E.; Hartman, G.H. 1989. **Influence of cover complexity and current velocity on winter habitat use by juvenile coho salmon (*Oncorhynchus kisutch*).** Canadian Journal of Fisheries and Aquatic Sciences 46: 1551-1557.

Montgomery, D.R.; Buffington, J. 1997. **Channel-reach morphology in mountain drainage basins.** Geological Society of America Bulletin 109: 596-611.

Montgomery, D.R.; Buffington, J.M.; Smith, R.D.; Schmidt, K.M.; Pess, G. 1995. **Pool spacing in forest channels.** Water Resources Research 31(4): 1097-1105.

Moore, K.M.S.; Gregory, S.V. 1988. **Summer habitat utilization and ecology of cutthroat fry (*Salmo clarki*) in Cascade mountain streams.** Canadian Journal of Fisheries and Aquatic Sciences 45: 1921-1930.

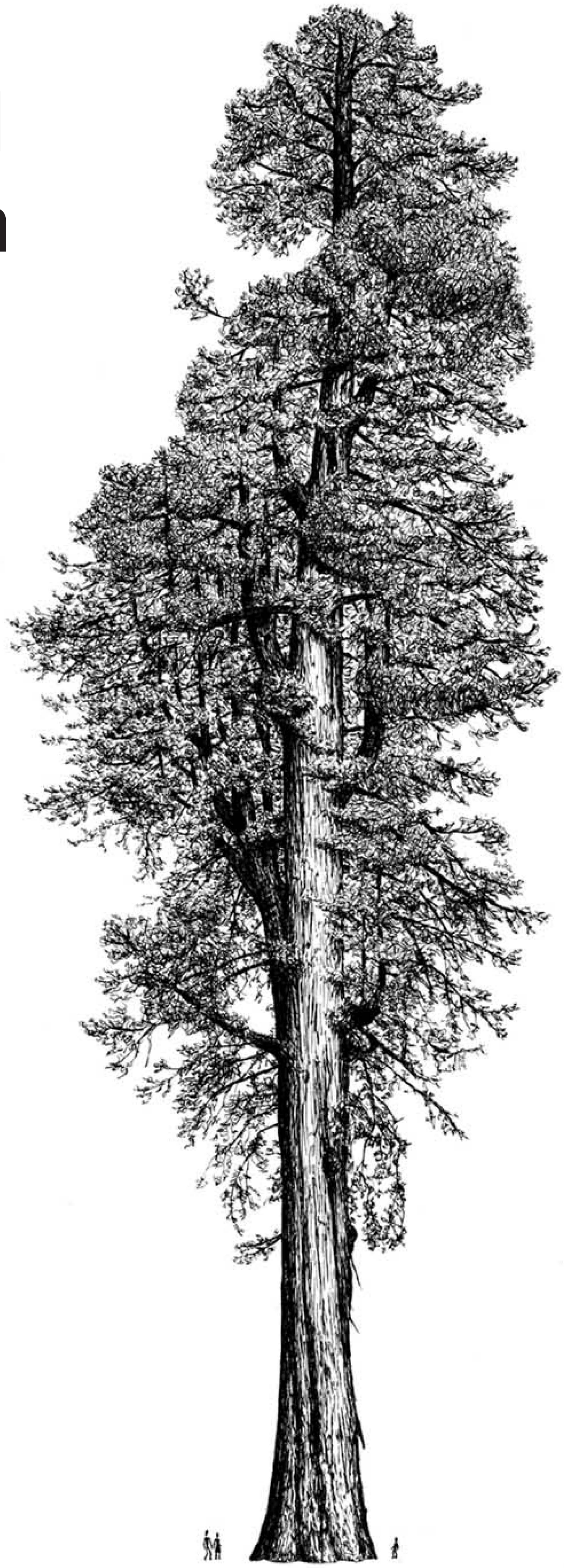
Quinn, T.P.; Peterson, N.P. 1996. **The influence of habitat complexity and fish size on over-winter survival and growth of individually marked juvenile coho salmon (*Oncorhynchus kisutch*) in Big Beef Creek, Washington.** Canadian Journal of Fisheries and Aquatic Sciences 53: 1555-1564.

Robison, E.G. 1998. **Reach scale sampling metrics and longitudinal pattern adjustments of small streams.** Corvallis, OR: Oregon State University; 254 p. Ph.D. dissertation.

Robison, E.G.; Kaufmann, P.R. 1994. **Evaluating two objective techniques to define pools in small streams.** In: Marston, R.A.; Hasfurthe, V.R., eds. Effects of human induced changes on hydrologic systems: American Water Resources Association Summer Symposium; 1994 June 26-29; Jackson Hole, WY. American Water Resources Association Technical Publication Series TPS-94-3. Heredon, VA; 659-668.

SESSION 2

Genetics and Regeneration



Applications of Redwood Genotyping by Using Microsatellite Markers[1]

Chris Brinegar,[2] Dan Bruno, Ryan Kirkbride, Steven Glavas, and Ingrid Udranszky

Abstract

A panel of polymorphic microsatellite markers have been developed in coast redwood (*Sequoia sempervirens*). Two loci in particular (*Seq*18D7-3 and *Seq*21E5) demonstrate the potential of microsatellite genotyping in the assessment of genetic diversity and inheritance in redwoods. The highly polymorphic *Seq*18D7-3 marker provided evidence for the planting of non-local redwood seedlings in a restored area of a state park and was used to genotype progeny from a controlled cross. A putative chloroplast microsatellite locus (*Seq*21E5) was shown to be paternally inherited in two controlled crosses and can therefore be used as a marker for alleles carried in pollen. Significant differences in allelic diversity were observed at the *Seq*21E5 locus in redwood populations with different ecological and management histories. Advantages of microsatellite analysis and its limitations (including the problem of null alleles) are discussed as well as its potential as a tool for forest management and conservation.

Key words: genotyping, microsatellites, redwood, Sequoia, *simple sequence repeats*

Introduction

Data from common garden experiments (Anekonda and Libby 1996), monoterpene characterization (Hall and Langenheim 1987), and allozyme analysis (Rogers 1997, 1999, 2000) show that coast redwood [*Sequoia sempervirens* (D. Don) Endl.] possesses significant genetic variation. Much of its genetic diversity might be attributed to the fact that it is a hexaploid organism. The potential origins of hexaploidy in redwood have been reviewed by Ahuja and Neale (2002). There have been limited molecular genetics studies on redwoods using DNA-based approaches. Neale and others (1989) used restriction fragment length polymorphism (RFLP) to show that both chloroplast and mitochondrial DNA are inherited paternally in redwood. Evidence for variation in the redwood chloroplast genome has also been demonstrated by RFLP analysis (Ali and others 1991). In order to understand the complexities of the redwood genome and its mechanisms of inheritance in greater detail, it will be necessary to have many DNA markers in hand that are amenable to rapid and relatively inexpensive genetic analyses. Microsatellites, also known as simple sequence repeat (SSR) markers, are non-coding, co-dominant loci with high mutation rates that have been used extensively for inheritance and population diversity studies in plants and animals (Parker and others 1998).

We have characterized a panel of five polymorphic microsatellite loci from a

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Conservation Genetics Laboratory, Department of Biological Sciences, San Jose State University, San Jose, CA 95192. email: cbrinega@email.sjsu.edu

redwood genomic library that have shown utility as genetic markers (Bruno 2002, Bruno and Brinegar 2004). This paper presents preliminary data that demonstrate how these markers can be used to assess inheritance in controlled crosses and characterize genetic diversity within and between populations. The potential of microsatellite genotyping in forest management will also be addressed.

Methods

Redwood branchlets were collected in coin envelopes and subsequently dried with desiccant at room temperature prior to DNA extraction. Redwood genomic library construction, screening, microsatellite clone characterization, primer sequences, DNA extraction, PCR conditions and gel electrophoresis are described elsewhere (Bruno and Brinegar 2004). PCR products were separated on non-denaturing, high-resolution Spreadex EL 300 or EL 500 gels (Elchrom Scientific, Cham, Switzerland). M3 DNA markers (Elchrom) were used to estimate allele sizes. Three microsatellite markers were used in this report, but sequences for all five clones [*Seq*37G5 (CT_n), *Seq*21E5 ($CTTA_n$), *Seq*20E5(CA_n), *Seq*18D7-3 (CTT_n), and *Seq*8E8 (CA_n)] are available in the GenBank database (accession numbers AY562165, AY562166, AY562167, AY562168, and AY562169, respectively).

Results and Discussion

Redwood microsatellite polymorphism

Microsatellite *Seq*18D7-3 is the most polymorphic locus in the current panel of markers with 13 alleles observed so far in several test populations. Eight alleles were observed in the genotypes of only 12 individual trees from Big Basin Redwoods State Park *(fig. 1)*. In this example, each individual had one to three alleles with the microsatellite repeats ranging in size from $(CTT)_5$ to $(CTT)_{21}$. Allele sizes differed by multiples of three base pairs (bp) as expected, and sequencing of individual alleles eluted from the gel confirmed that size polymorphisms were, in fact, due only to changes in the number of tandem CTT repeats.

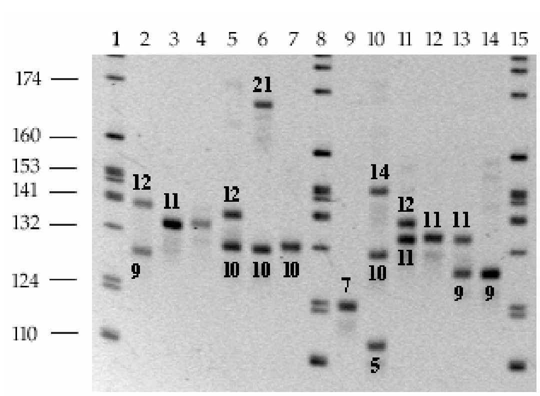


Figure 1—Size polymorphism of alleles at the *Seq*18D7-3 microsatellite locus in 12 randomly selected redwoods from Big Basin Redwoods State Park. Numbers above or below bands indicate the number of CTT repeat units in that allele. Lanes 1, 8 and 15 are DNA markers. Marker sizes (base pairs) are indicated at left.

*Seq*8E8 is also very polymorphic with 10 known alleles while *Seq*37G5 and *Seq*20E5 have three and four known alleles, respectively. However, these three markers have not been tested on as many populations as the other two and may actually have higher degrees of polymorphism. *Seq*21E5 is unique in that it shows very high polymorphism, but each individual (out of several hundred tested) possesses only one allele. This observation suggests a cytoplasmic origin, presumably from chloroplast DNA, since microsatellite sequences are common in chloroplasts (Provan and others 2001). Further data on the allelic diversity and inheritance of the *Seq*21E5 marker are provided later in this report.

In theory, since redwoods are hexaploid, the number of alleles at a nuclear locus could range from one to six per individual. Redwoods may be autoallohexaploid (Saylor and Simons 1970) with an AAAABB type of genome ancestry. If so, it is possible that some microsatellite loci may appear only in the A or B chromosome sets. This could restrict the maximum number of alleles per individual to two or four. We have yet to observe more than four alleles per locus in any of our markers. Though the true nature of redwood hexaploidy is still unresolved (Ahuja and Neale 2002), microsatellite DNA clones have potential as probes for *in situ* chromosome hybridization to help resolve this question.

Detecting non-indigenous redwoods

In a study to test the utility of our microsatellite markers, a group of closely spaced redwoods in Big Basin Redwoods State Park were genotyped to assess their genetic relationship. The trees are along a 50 m edge of a restored meadow with 12 young redwoods (0.5 to 7 m tall) growing between two older fairy ring groves to the north and south. *Figure 2A* shows the *Seq*18D7-3 genotypes of these 12 small redwoods (trees 7-18) and six redwoods each from the southern grove (trees 1-6) and northern grove (trees 19-24). Both groves had different genotypes, but all trees within a grove shared the same allelic pattern. Allozyme data (Rogers 2000) indicate that redwoods in a fairy ring may not all be clones of the parent stump, so identification of trees within these groves as clonal is tentative until more loci are genotyped.

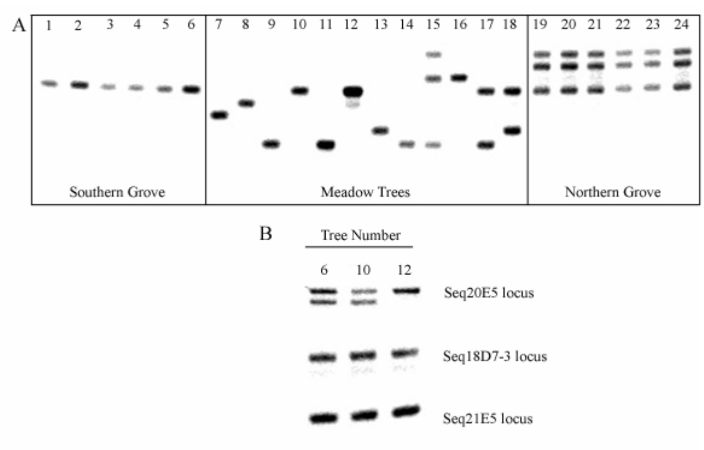


Figure 2—Microsatellite genotyping to assess relationships between redwoods in a restored area. A. *Seq*18D7-3 genotypes of redwoods in two fairy ring groves and along a meadow's edge between the groves. B. Genotypes at three loci (*Seq*21E5, *Seq*18D7-3, *Seq*20E5) of redwoods 6, 10 and 12. See text for discussion.

None of the small trees between the groves displayed the three-allele pattern of the northern grove, and only two (trees 10 and 12) had the same single-allele pattern as the southern grove. Seven of the small redwoods had no alleles in common with either of the groves. Comparison of trees 10 and 12 with tree 6 from the southern grove at two additional microsatellite loci differentiated tree 12 from the others by its extra allele at *Seq*20E5 *(fig. 2B)*.

Due to a lack of shared alleles, it is apparent that most of the small trees (7, 8, 9, 11, 13, 14 and 16) were neither clones of either grove nor derived from seed produced by either grove. Also, because of a stream on one side and a meadow on the other, there were no other seed-producing redwoods within 50 m of this site. Tree 10 appears to be the only one that could possibly be a clone of one of the groves (southern). Alternatively, it could be a highly inbred relative.

Although this limited microsatellite data suggests that the two groves are clonal and that few of the small redwoods are closely related to the groves, an analysis with more markers and sampling of other trees in the vicinity would be required for definitive proof. However, after researching the recent history of this section of Big Basin, the interpretation of the data became easier. As part of prior restoration projects, Girl Scout troops had planted seedlings from nursery stock in several areas near the park headquarters—including along the meadow between the northern and southern groves! Based on this information and the microsatellite data, there is a high probability that most of the 12 small redwoods are not indigenous to Big Basin.

The practice of planting non-local redwood seedlings in parks and private timber land could potentially lead to the "genetic pollution" of populations that are well adapted to their local environment. This potential problem has been addressed previously by Millar and Libby (1989). A thorough microsatellite analysis could be used to assess the presence of non-native trees in areas suspected of being planted with less well adapted nursery stock.

Inheritance in controlled crosses

In angiosperms chloroplasts are maternally or biparentally inherited (Provan and others 2001) whereas paternal inheritance of chloroplasts has been reported in a number of conifers, such as pine (Dyer and Sork 2001). Neale and others (1989) provided restriction fragment length polymorphism (RFLP) evidence that chloroplasts, as well as mitochondria, are paternally inherited in coast redwood. The use of chloroplast genetic markers should, therefore, allow alleles from the pollen donor to be monitored in controlled crosses. Since redwood microsatellite marker *Seq*21E5 had properties of a chloroplast locus (one allele per individual and typically higher PCR product yield than nuclear loci due to more copies per cell), we tested its possible paternal inheritance using tissue from known redwood families supplied by the UC-Berkeley Russell Plantation.

Three parents were used in two crosses with Arc154 being the female parent in one cross and the male parent in the other. In Arc154 x Arc28 *(fig. 3A)*, the female parent's allele size was 120 bp and the male parent's allele size was 140 bp. All five of the tested sibs had the 140 bp allele. All five of the S1 x Arc154 sibs inherited the 120 bp paternal allele *(fig. 3B)*. These results support the paternal inheritance of chloroplasts in redwoods, but more work is needed to determine if maternal "leakage" of chloroplasts occurs during fertilization. In the conifer *Chamaecyparis obtusa*, chloroplast DNA was inherited primarily through the pollen but a small amount (2.5 percent) of maternal leakage was detected (Shiraishi and others 2001).

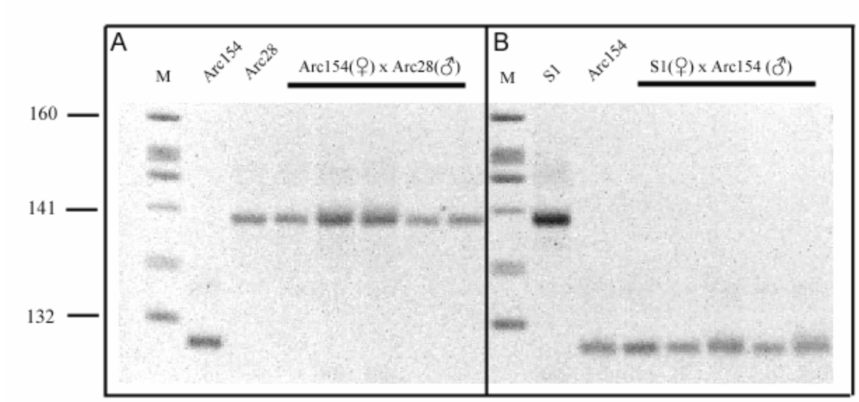


Figure 3—Paternal inheritance of the *Seq*21E5 locus in two controlled crosses. A. Genotypes of parents Arc154 and Arc28 with genotypes of five sibs. B. Genotypes of parents S1 and Arc154 with genotypes of five sibs. DNA markers are in lanes labeled M with their sizes (base pairs) indicated at left.

The *Seq*18D7-3 locus (presumably a nuclear marker) was also tested on the S1 x Arc154 family *(fig. 4)*. The female parent (S1) had one allele at 141 bp while the male parent (Arc154) had a 138 bp allele (one less repeat unit). The five sibs from this cross displayed three different genotypes: both parental alleles (sibs A and B), no visible alleles (sib C), or only the maternal allele (sibs D and E). The absence of the 138 bp allele in sibs D and E could be explained by the donation of a paternal null allele at this locus. The absence of alleles in sib C could be due to null allele contributions from both parents or a failure in the amplification reaction; however, the *Seq*21E5 locus amplified well using the same DNA extract from sib C.

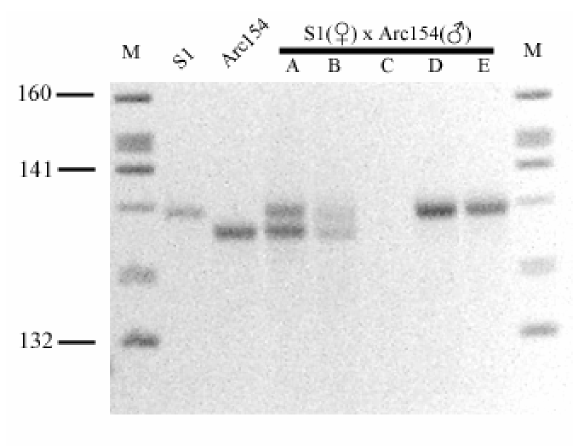


Figure 4—Inheritance of *Seq*18D7-3 alleles in a controlled cross (S1 x Arc154). DNA markers are in lanes labeled M with their sizes (base pairs) indicated at left.

Null alleles are major obstacles in the interpretation of inheritance data, especially in polyploids where there could be a higher probability of their occurrence (Weeden and Wendel 1989). Their presence can be inferred from the absence of an amplifiable allele from one parent (as above) or from a low observed population heterozygosity (H_o) at the locus. (Unfortunately, due to the difficulty in estimating allele dosage in polyploids from allele staining intensities, expected heterozygosities cannot be easily estimated from microsatellite data.) Indications of null alleles at three of our four presumptive nuclear microsatellite markers were found in a population of second-growth redwoods in the Santa Cruz Mountains. Their observed H_o values were relatively low, ranging from 0.28 to 0.44 (Bruno and Brinegar 2004). *Seq*18D7-3, however, had a higher H_o of 0.80, indicating that null allele frequency is low in that southern population. S1 and Arc154 are clones from northern populations where the null allele frequency could be higher. In a redwood allozyme study (Rogers 1997) there was no evidence of null alleles in the six enzyme systems tested. Microsatellite alleles, however, being non-coding sequences, do not impart any known selective advantages to the organism and will have no deleterious consequences if lost.

These results demonstrate that microsatellite markers can be useful in characterizing the results of controlled crosses. Libby and others (1981) provided evidence of inbreeding depression in redwoods. Microsatellite genotyping might also have potential in assessing inbreeding in redwood breeding programs.

Genetic diversity within and between populations

With highly polymorphic genetic markers, the allelic richness of populations—a measure of genetic diversity (Petit and others 1998)—can be measured and compared. To avoid the problem of null alleles, we examined the allelic frequencies at the *Seq*21E5 locus in a total of 194 trees from two northern old-growth populations (Humboldt County) and two southern second-growth populations (Santa Cruz County). Histograms of allele frequencies are shown in *figure 5*.

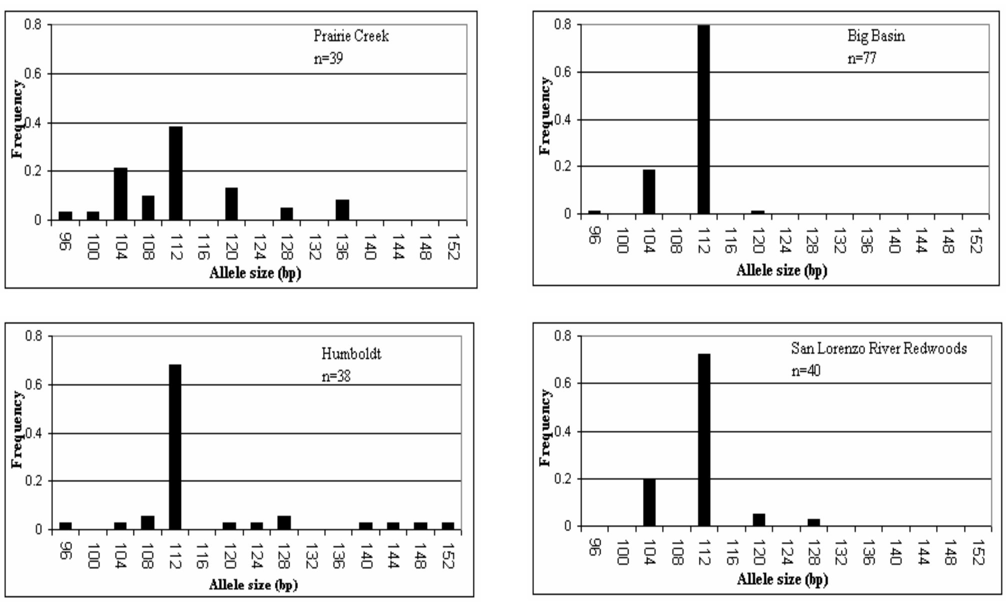


Figure 5—*Seq*21E5 allele frequencies in four redwood populations. Individuals in Prairie Creek Redwoods State Park and Humboldt Redwoods State Park were from old-growth populations. Individuals from Big Basin Redwoods State Park and the San Lorenzo River Redwoods (a private reserve near Castle Rock State Park in Santa Cruz County) were from second-growth populations.

The 112 bp allele was by far the most common, found in nearly 68 percent of all trees tested. Frequency of this allele in the four populations ranged from a low of almost 40 percent in Prairie Creek to a high of 80 percent in Big Basin. However, the allelic richness (the number of different alleles observed) in the two old-growth populations was significantly higher than that of the second-growth populations. The Prairie Creek and Humboldt Redwoods State Park old-growth populations had eight and 11 different alleles, respectively, while the Big Basin Redwoods State Park and the San Lorenzo River Redwoods second-growth populations had only four alleles each.

There are several possible explanations for the lower allele numbers in the southern populations. Collections were made to avoid the sampling of clones, but it is possible that the second-growth population samples included some trees that sprouted from the same parent, thereby reducing the chance of sampling other trees with different alleles. However, the low allelic richness of the Big Basin population, in spite of the high sample number (n = 77), argues against a significant sampling bias. It has been proposed that there is a higher percentage of clonal reproduction in the southern redwood forests due to the warmer, drier climate that makes seedling establishment more difficult (Veirs 1996). This would reduce the creation and spread of new alleles by limiting the number of trees produced through sexual reproduction. Some of the loss in diversity at this locus could also be caused by the harvesting of the old-growth forests. Powers and Wiant (1970) found that stump sprouting decreased significantly when redwoods over the age of 400 years were cut. Therefore, the primarily clonal second-growth trees that replaced the clear cut old-growth forests in the southern region might under-represent the diversity of the original forest. Finally, differences in founding populations may account for the observations.

Since *Seq*21E5 is a putative chloroplast locus, the allelic diversity it displays in various populations may not necessarily correlate with diversity at nuclear loci. Also, one should not draw too many conclusions by comparing genotypes of old-growth northern forests with second-growth southern forests. More valid comparisons would be to genotype adjacent old-growth and second-growth populations or northern and southern old-growth populations. However, the many questions that these observations create are amenable to being answered by more thorough genetic analysis using allozymes or microsatellites.

Conclusions

The preliminary data we have presented on microsatellite genotyping of redwoods demonstrates the potential of this approach for the study of redwood inheritance and diversity. Using microsatellite genotyping to detect non-local nursery stock planted in parks or timberland will ensure that native redwood populations maintain their adaptive traits. By tracking paternal inheritance with chloroplast microsatellite loci, pollination can be studied in more detail. In spite of problems caused by null alleles, genotyping progeny from controlled crosses can provide useful information to breeders.

The development and characterization of polymorphic microsatellite markers can be a long and expensive undertaking, but once available the PCR-based analysis is rapid and cost effective. Using our current system, one person can extract, amplify and genotype 40 redwoods at two loci in two days at a reagent and supply cost of approximately $70. We are working on a 96-well plate format for both DNA extraction and PCR that will increase the two-day output to at least 180 samples. Some loci might be amenable to PCR multiplexing which could eventually double or triple that output. Using a multi-capillary electrophoresis system (with fluorescent detection of labeled multiplexed PCR products) will further increase data production while reducing labor and analysis time. Such high throughput will make large-scale population studies much more feasible than they are at present. We will continue to characterize additional redwood microsatellite loci which will allow a full genetic analysis of individual trees and populations in controlled and natural environments.

Microsatellite genotyping, or other DNA-based methods, will no doubt become another routine tool for redwood geneticists, forest managers, and conservationists of the future. Decisions to conserve redwood forests could, in part, be dictated by an analysis of their genetic structure. The redwood genome is complex but it will eventually, and no doubt begrudgingly, provide us with the information we need to make good forest management decisions.

Acknowledgments

We thank the California Department of Parks and Recreation, the National Park Service, and the Sempervirens Fund for permission to collect redwood tissue. We are also grateful to Dr. William Libby and the UC-Berkeley Russell Plantation for providing tissue from controlled crosses. This work was partially funded by the Graduate Research Fellowship program in the Department of Biological Sciences, San Jose State University.

References

Ahuja, M.R.; Neale, D.B. 2002. **Origins of polyploidy in coast redwood (*Sequoia sempervirens* (D. Don) Endl.) and relationship of coast redwood to other genera of Taxodiaceae.** Silvae Genetica 51: 93-100.

Ali, I.F; Neale, D.B.; Marshall, K.A. 1991. **Chloroplast DNA restriction fragment length polymorphism in *Sequoia sempervirens* (D. Don) Endl., *Pseudotsuga menziesii* (Mirb.) Franco, *Calocedrus decurrens* (Torr.), and *Pinus taeda* L.** Theoretical and Applied Genetics 81: 83-89.

Anekonda, T.S.; Libby, W.J. 1996. **Effectiveness of nearest-neighbor data adjustment in a clonal test of redwood.** Silvae Genetica 45: 46-51.

Bruno, D.; Brinegar, C. 2004. **Microsatellite markers in coast redwood (*Sequoia sempervirens*).** Molecular Ecology Notes 4: 482-484.

Bruno, D. 2002. **Isolation and characterization of microsatellite loci in *Sequoia sempervirens*.** San Jose, CA: San Jose State University; 97 p. M.S. thesis.

Dyer, R.J.; Sork, V.L. 2001. **Pollen pool heterogeneity in shortleaf pine, *Pinus echinata* Mill.** Molecular Ecology 10: 859-866.

Hall, G.D.; Langenheim, J.H. 1987. **Geographic variation in leaf monoterpenes of *Sequoia sempervirens*.** Biochemical Systematics and Ecology 15: 31-43.

Libby, W.J.; McCutchan, B.G.; Millar, C.I. 1981. **Inbreeding depression in selfs of redwood.** Silvae Genetica 30: 15-25.

Millar, C.I.; Libby, W.J. 1989. **Restoration: Disneyland or a native ecosystem? A question of genetics.** Restoration Management Notes 7: 18-24.

Neale, D.B.; Marshall, K.A.; Sederoff, R.R. 1989. **Chloroplast and mitochondrial DNA are paternally inherited in *Sequoia sempervirens* (D. Don) Endl.** Proceedings of the National Academy of Sciences of the USA 86: 9347-9349.

Parker, P.G.; Snow, A.A.; Schug, M.D.; Booton, G.C.; Fuerst, P.A. 1998. **What molecules can tell us about populations: choosing and using a molecular marker.** Ecology 79: 361-382.

Petit, R.J.; El Mousadik, A.; Pons, O. 1998. **Identifying populations for conservation on the basis of genetic markers.** Conservation Biology 12: 844-855.

Powers, R.F.; Wiant, H.V. 1970. **Sprouting of old-growth coastal redwood stumps.** Forest Science 16: 339-341.

Provan, J.; Powell, W.; Hollingsworth, P.M. 2001. **Chloroplast microsatellites: New tools for studies in plant ecology and evolution.** Trends in Ecology and Evolution 16: 142-147.

Rogers, D.L. 1997. **Inheritance of allozymes from seed tissues of the hexaploid gymnosperm, coast redwood, *Sequoia sempervirens* (D. Don) Endl. (Coast redwood).** Heredity 78: 166-175.

Rogers, D.L. 1999. **Allozyme polymorphisms discriminate among coast redwood (*Sequoia semperviren*s) siblings.** Journal of Heredity 90: 429-433.

Rogers, D.L. 2000. **Genotypic diversity and clone size in old-growth populations of coast redwood (*Sequoia sempervirens*).** Canadian Journal of Botany 78: 1408-1419.

Saylor, L.C.: Simons, H.A. 1970. **Karyology of *Sequoia sempervirens*: Karyotype and accessory chromosomes.** Cytologia 35: 294-303.

Shiraishi, S.; Maeda, H.; Toda, T.; Seido, K.; Sasaki, Y. 2001. **Incomplete paternal inheritance of chloroplast DNA recognized in *Chamaecyparis obtusa* using an intraspecific polymorphism of the trnD-trnY intergenic spacer region.** Theoretical and Applied Genetics 102: 935-941.

Veirs, D.D. 1996. **Ecology of the coast redwood.** In: LeBlanc, J., ed. Proceedings of the conference on coast redwood forest ecology and management. University of California Cooperative Extension Forestry Publication. Arcata, CA: Humboldt State University; 9-12.

Weeden, N.F.; Wendel, J.F. 1989 **Genetics of plant isozymes.** In: Soltis, D.E.; Soltis, P.S., eds. Isozymes in plant biology, Advances in Plant Sciences Series, Vol. 4. Portland, OR: Dioscorides Press; 46-72.

Planting?—What and Where?[1]

W.J. Libby[2]

The answers to these questions depend on the goals of the organization, and who its neighbors are. For parks and other owners with long-term goals of maintaining or restoring natural ecosystems, or if their near neighbors have such goals, then two answers are preferred: (1) they probably should not plant, but should rely instead on natural regeneration; or (2) if they do plant, the plants should be from strictly local seed sources. "Strictly local" depends on the species, and it is a much smaller area for small insect-pollinated species than it is for large wind-pollinated species. For organizations with a major goal of supplying wood from the forest, then planting genetically-select tested families or clones of the species to be harvested has substantial merit. Current knowledge of the population architectures of Douglas-fir and coast redwood will be reviewed, with implications for choice of planting stocks. In short, Douglas-fir seems to allow seed-transfer only over relatively short ecological distances, with well-documented cases of serious failures when non-local populations are used for reforestation. The problems were generally not evident for several decades, but became serious before harvest age was attained. Redwood seems more broadly-adapted, but data are based on trees less than one percent of the age attained by old-growth redwood forests. The implications of genetic contamination of native forests by pollen from planted trees or associated species will also be presented.

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Zobel Forestry Associates, 28 Valencia Rd., Orinda, CA 94563. email: wlibby2@aol.com

Spatial Genetic Patterns in Four Old-Growth Populations of Coast Redwood[1]

Deborah L. Rogers[2] and Robert D. Westfall[3]

Key words: canonical surface analyses, demographic structure, spatial autocorrelation

Introduction

The mating systems of plant species and their patterns of pollen flow and seed dispersal create a pattern in the distribution of genetic diversity which is referred to as spatial genetic structure (Wright 1946). Over time, that pattern may be modified by other influences such as natural selection. In studying the spatial genetic structures of plant species directly, we can infer the past events of gene flow that contributed to those structures. Elucidating the dynamics of gene flow within a species improves our understanding of the nature of local populations—the units within which the majority of gene sharing occurs, and the units that are often the focus of conservation or management.

The capacity of coast redwood (*Sequoia sempervirens* D. Don) to reproduce sexually to produce genetically distinct seedlings or by sprouting to produce clonal replicates provides for many possibilities for its spatial genetic structure. Results of previous common-garden studies of morphological and physiological traits (Anekonda and Criddle 1993, Anekonda and others 1994, Millar and others 1985) and of allozyme diversity (Rogers 2000) have revealed high levels of genetic diversity in coast redwood. This diversity provides the potential for detecting local spatial genetic structure. Furthermore, results from the first genetic study of clonal structure in the species showed that clone sizes were generally small (in other words, few ramets per clone), indicating that many of the studied redwoods were genetically distinct (Rogers 2000). In addition, the increasing array of available genetic markers and statistical methods provide more opportunities and analytical power for illuminating genetic structure. For example, whereas estimates of gene flow based on Wright's *F*-statistics represent evolutionary averages (for example, Bossart and Prowell 1998), recently developed statistical methods performed on hypervariable markers such as microsatellites allow the potential to detect contemporary patterns of dispersal (Smouse and Peakall 1999). Multivariate statistical methods commonly employed to study spatial genetic structure include principal components analysis, canonical variate analysis, canonical correlation analysis and spatial autocorrelation methods. Most recently, multilocus spatial autocorrelation methods have been

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Assistant Research Geneticist, Genetic Resources Conservation Program, University of California, One Shields Avenue, Davis, CA 95616. email: debrogers@ucdavis.edu

[3] Biological Statistician (Quantitative Geneticist), Sierra Nevada Research Center, USDA Forest Service, PSW Research Station, POB 245, Berkeley, CA 94701. email: bwestfall@fs.fed.us

developed and applied to studies of spatial genetic structure. This multiallelic approach (as opposed to a locus-by-locus, allele-by-allele analysis) may be particular appropriate for fine-scale spatial genetic structure studies, strengthening the spatial signal by reducing stochastic noise (Peakall and others 2003, Smouse and Peakall 1999).

Methods

In this study, we employ multilocus spatial autocorrelation procedures and canonical correlation analysis to describe fine-scale spatial genetic structure in old-growth populations of coast redwood. We compare two geographically proximate but contrasting redwood habitats to illustrate that microgeographic selection and population history, not just gene flow, may play a role in spatial genetic patterns for this species.

Four sites were selected from old-growth populations of coast redwood in Humboldt Redwoods State Park in northern California. Two were located in upland areas with (relatively) dry microclimates, mixed species canopy conditions, and with historically frequent fire disturbance. Veirs (1980a, 1980b) studied fire disturbance patterns in the general vicinity and determined that fire frequency was sometimes ten times greater on the xeric sites (comparable to the upland sites of this study) than on the more mesic sites (lowland sites). Fire influences a variety of reproductive and early growth conditions relevant to coast redwood including soil sterilization and mineralization (McBride and Jacobs 1977), vegetative sprouting, and removal of overstory (Jacobs 1987). Two sites were selected from another typical old-growth condition—that of lowland, flood-influenced areas where redwood was dominant.

Foliage was collected from 121 to 160 contiguous trees from each of the four sites, mapping the location of each tree. Diameter (at breast height) was measured on each sampled tree. See Rogers (2000) for additional detail on site conditions and sampling methods. Foliar tissue was assayed for allozyme loci. According to previously established methods, we resolved four alleles at each of seven loci (Rogers 1997, 1999). Because coast redwood is a hexaploid and staining intensities from the protocols used in this study do not accurately or consistently reflect allele frequencies (Rogers 1997), individual isozyme phenotypes were scored for the presence or absence of each allele. Consequently, multilocus allozyme phenotypes rather than allele frequencies were analyzed.

To examine within-site spatial patterns of individual phenotypes, we regressed a second-order model of the x and y coordinates of each individual against its multilocus phenotype by canonical trend surface analysis. In the lowland and upland sites, the dominant pattern was in the x-direction, which was roughly coincident along the Eel river in the former and with elevation in the latter. In all sites, the direction of least change was in southeast to southwest sector. To further examine spatial relationships we generated directional spatial correlograms at each site. Though some differences are apparent, the general pattern of highest correlations was in a similar direction on all sites, consistent with the trend surface results, possibly reflecting the dominant wind pattern within the region. Spatial autocorrelations were positive and significant within the first 100-foot distance interval (in other words, in pairwise comparisons of all trees included in the study) on all four sites. However, the values were generally larger on the two upland sites, and continued to be positive at greater distances (relative to the two lowland sites). Inclusion of putative clones greatly increases autocorrelation in the first two distance intervals, especially on the upland sites.

Discussion

Some features of coast redwood are traditional indicators of long-distance gene flow and absence of fine-scale spatial genetic structure. The tall stature of mature trees, wind dispersal of pollen and seed, high rates of outcrossing, and high levels of inbreeding depression are traits that are generally correlated with low expectations of genetic structure (Hamrick and Godt 1990). Yet we found considerable fine-scale genetic structure. Some of this is related to clonal replicates, although even when this effect is removed, significant and positive spatial autocorrelations remain within one hundred feet on all sites, and at greater distances and higher values on the upland sites.

Differences in elevation and disturbance events between the upland and lowland sites would seem to provide different selection regimes that could be expected to result in some degree of genetic differentiation between site types. The higher and more widespread autocorrelations on upland sites might be related to several factors. The redwoods that persist on these fire-influenced sites compete with other tree species such as Douglas-fir *(Pseudotsuga menziesii)*, tanoak *(Lithocarpus densiflora)*, madrone *(Arbutus menziesii)*, and Pacific yew *(Taxus brevifolia)*. Consequently, following a fire disturbance, microsites would quickly become colonized and are more likely to reflect recruits from the dominant seed source at a specific time, rather than gradual recruitment of redwoods from multiple parents. The age structure, too, may differ between the upland and lowland sites. Coast redwood is a very difficult species from which to infer age based on diameter—because of factors including false rings and missing rings (Brown and Swetnam 1994). However, the size distributions for the four sites showed a much higher proportion of very large trees (for example, above 60 inches diameter) on the two lowland sites (20 to 25 percent) relative to the two upland sites (five to 15 percent). Generalizing that very large trees are older than much smaller trees, it seems probable that a disturbance that was sufficiently severe and comprehensive to be considered ‘stand replacing’ probably has occurred more recently (and more frequently, in the long term) on the upland sites relative to the lowland sites. Also, the upland sites contained some younger (seedling and sapling) trees of nonclonal origin. The lowland sites, with much higher densities of coast redwood, contained almost no young trees that weren’t sprouts from pre-existing trees. Studies of genetic structure in other plant species have suggested that spatial structure may also vary with life stage and age class (for example, Kalisz and others 2001).

Although the study is limited to old-growth conditions in one particular region of the species’ range, and covers only a modest geographic distance, the results suggest that the majority of gene flow in coast redwood may be more local than might be expected on the basis of generalizations concerning open-pollinated conifers. The seeds of redwood—although light, small, and borne at often-great heights in the canopy—are, nevertheless, wingless, and may settle relatively close to the parent trees. However, in communities that have persisted for long periods, as is likely in the lowland sites, gene flow will diminish spatial structuring across the site.

Conclusion

In conclusion, coast redwood in old-growth context can have considerable fine-scale spatial genetic structure. This is a reflection of both (albeit limited) clonal reproduction and patterns of seed and pollen dispersal, possibly reinforced by local natural selection. The local structure is stronger on upland (relative to lowland) sites and consistent with low effective population size and low dispersal distance conditions simulated in Epperson and others (1999), reflecting younger and sparser redwood populations that colonize quickly, and in competition with other species, following a fire disturbance. Although upland and lowland sites are distinguished in their autocorrelation values and distances, the direction of correlation is similar on all sites. This may reflect a common direction for the predominant winds in this region during the months of most abundant pollen and seed dispersal, generally December through January (Boe 1961, Metcalf 1924). The generally small clone size previously reported for coast redwood (Rogers 2000), coupled with the fine-scale genetic structure reported in this study, suggest that in areas that are managed to conserve natural genetic structure and ecosystem processes of coast redwood, attention should be paid to maintaining local populations that are probably smaller than suggested by the tree's stature. In particular, any restoration activities that involve seeding or planting of coast redwoods in natural or conserved areas, should consider using genetic material from the immediate vicinity unless there are higher-priority concerns that would over-ride genetic considerations, or additional information that indicates that the local populations for the target area are different from those in this study.

References

Anekonda, T.S.; Criddle, R.S.; Libby, W.J.; Hansen, L.D. 1993. **Spatial and temporal relationships between growth traits and metabolic heat rates in coast redwood**. Canadian Journal of Forest Research 23: 1793-1798.

Anekonda, T.S.; Criddle, R.S.; Libby, W.J. 1994. **Calorimetric evidence for site-adapted biosynthetic metabolism in coast redwood (*Sequoia sempervirens*).** Canadian Journal of Forest Research 24: 380-389.

Boe, K.N. 1961. **Redwood seed dispersion in old-growth cutovers**. Res. Note 177. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture.

Bossart, J.L.; Powell, D.P. 1998. **Genetic estimates of population structure and gene flow: limitations, lessons and new directions.** Trends in Ecology and Evolution 13: 202-205.

Brown, P.M.; Swetnam, T.W. 1994. **A cross-dated fire history from coast redwood near Redwood National Park, California.** Canadian Journal of Forest Research 24: 21-31.

Epperson, B.K., Huang, Z.; Li, T.Q. 1999. **Measures of spatial structure in samples of genotypes for multiallelic loci.** Genetical Research 73: 251-261.

Hamrick, J.L.; Godt, M.J.W. 1990. **Allozyme diversity in plant species.** In: Brown, A.H.D.; Clegg, M.T.; Kahler, A.L.; Weir, B.S., eds. Plant population genetics, breeding and genetic resources. Sunderland, MA: Sinauer; 43–63.

Jacobs, D.F. 1987. **The ecology of redwood (*Sequoia sempervirens* [D. Don] Endl.) seedling establishment.** Berkeley, CA: University of California; Ph.D. dissertation.

Kalisz, S.; Nason, J.D.; Hanzawa, F.M.; Tonsor, S.J. 2001. **Spatial population genetic structure in *Trillium grandiflorum*: the roles of dispersal, mating, history, and selection.** Evolution 55: 1560-1568.

McBride, J.; Jacobs, D. 1977. **The ecology of redwood [*Sequoia sempervirens* (D. Don) Endl.] and the impact of man's use of the redwood forest as a site for recreational activities.** Berkeley, CA: University of California Dept. of Forestry and Conservation; 39 p.

Metcalf, W. 1924. **Artificial reproduction of redwood**. Journal of Forestry 22: 873-893.

Millar, C.I.; Dunlap, J.M.; Walker, N.K. 1985. **Analysis of growth and specific gravity in a 20-year-old provenance test of *Sequoia sempervirens*.** Rep. No. 59. Berkeley, CA; California Agriculture and Experiment Station, Forest Service, U.S. Department of Agriculture.

Peakall, R.; Ruibal, M.; Lindenmayer, D.B. 2003. **Spatial autocorrelation analysis offers new insights into gene flow in the Australian bush rat, *Rattus fuscipes***. Evolution 57: 1182-1195.

Rogers, D.L. 1997. **Inheritance of allozymes from seed tissues of the hexaploid gymnosperm, *Sequoia sempervirens* (D. Don) Endl. (Coast redwood).** Heredity 78(2): 166–175.

Rogers, D.L. 1999. **Allozyme polymorphisms discriminate among coast redwood (*Sequoia sempervirens*) siblings.** Journal of Heredity 90: 429–433.

Rogers, D.L. 2000. **Genotypic diversity and clone size in old-growth populations of coast redwood (*Sequoia sempervirens*).** Canadian Journal of Botany 78: 1408-1419.

Smouse, P.E.; Peakall, R. 1999. **Spatial autocorrelation analysis of individual multiallele and multilocus genetic structure.** Heredity 82: 561-573.

Veirs, S.D., Jr. 1980a. **The role of fire in northern coast redwood forest vegetation dynamics**. In: Proceedings of the second conference sci. res. natl. parks; 1979 Nov. 26-30; San Francisco, CA. Washington, DC: National Park Service; 190–209.

Veirs, S.D., Jr. 1980b. **The influence of fire in coast redwood forest**. In: Proceedings of the fire history workshop. Gen. Tech. Rep. RM-81. Fort Collins, CO: Rocky Mountain Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 93–95.

Wright, S. 1946. **Isolation by distance under diverse systems of mating**. Genetics 31: 39-59.

Clonal Spread in Second-Growth Stands of Coast Redwood, *Sequoia sempervirens*[1]

Vladimir Douhovnikoff[2] and Richard S. Dodd[3]

Abstract

Coast redwood *(Sequoia sempervirens)* is one of the rare conifers to reproduce successfully through clonal spread. The importance of this mode of reproduction in stand development is largely unknown. Understanding the importance of clonal spread and the spatial structure of clones is crucial for stand management strategies that would aim to maximize genetic diversity. We have developed genetic markers to identify the clonal structure of nine second-growth redwood stands in Jackson Demonstration State Forest, California. Clonal spread was found to be important in the development of these stands, with an average of 6.7 stems being assigned as ramets of the same genet. Although fairy ring structures were commonly identified, we also detected a range of other spatial structures in these second-growth redwood stands. We also detected mixed genets within perceived fairy rings. The extent of clones and their spatial structure may have important evolutionary implications and will also have important consequences on effective population sizes for management purposes. Management and conservation of redwoods will benefit from a better understanding of the dynamics and structure of clonal spread in these forests.

Key words: AFLPs, clonal spread, genet, molecular markers, ramet, Sequoia sempervirens*, spatial structure, woody plant*

Introduction

Clonal spread is the vegetative establishment of genetically identical stems (ramets), which allows a genetic individual (genet) to replicate itself independent of sexual reproduction. Depending on the extent and the structure it takes there are many important ecological advantages (high productivity, competitive advantage for site colonization and persistence, survival of disturbance) and evolutionary implications (population genetic structure, effective population size, evolutionary potential) to clonal spread.

Coast redwood is exceptional for a gymnosperm in its ability to readily resprout and spread clonally after disturbances (Rydelius and Libby 1993; Rogers 1994, 2000), however, very little is known about the spatial structure of redwood clones, the relative importance of clonal vs. nonclonal reproduction, or the variables that influence clonal spread. In a study of old-growth stands using allozyme markers, Rogers (2000) found that a significant number of redwood stems were not genetically

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004 Rohnert Park, California.

[2] Simmons College, Park Science Center, Room S-259, 300 The Fenway, Boston, MA 02115. email: vladimir.douhovnikoff@simmons.edu

[3] University of California, Berkeley, Department of Environmental Science, Policy and Management, 201 Wellman Hall, Berkeley, CA 94720. email: dodd@nature.berkeley.edu

distinct individuals, but were ramets of larger genets, and thus resulted from cloning and not from reproduction by seed. There are no similar studies of second-growth redwood forests.

If we are to properly understand redwood ecology and evolutionary potential, it is essential to further explore the structure, dynamics, and significance of clonal spread in these forests. In the context of management and conservation, for example, this would result in more accurate estimates of minimum viable population size. This is the first major study of clonal spread in redwoods using molecular genetic techniques.

In this research our goals were to evaluate the importance of clonal spread in second-growth redwood stands, and to explore the local spatial structure of redwood clones.

Materials and Methods

The study was conducted in the Jackson Demonstration State Forest *(fig. 1)* because of its combination of historic intensive management and the availability of mature second-growth stands. A total of nine sites were randomly selected *(table 1)*. Sites were widely distributed across the forest, at least 500 m from each other and were pure stands of redwood estimated to have been last cut 50 to 70 years ago.

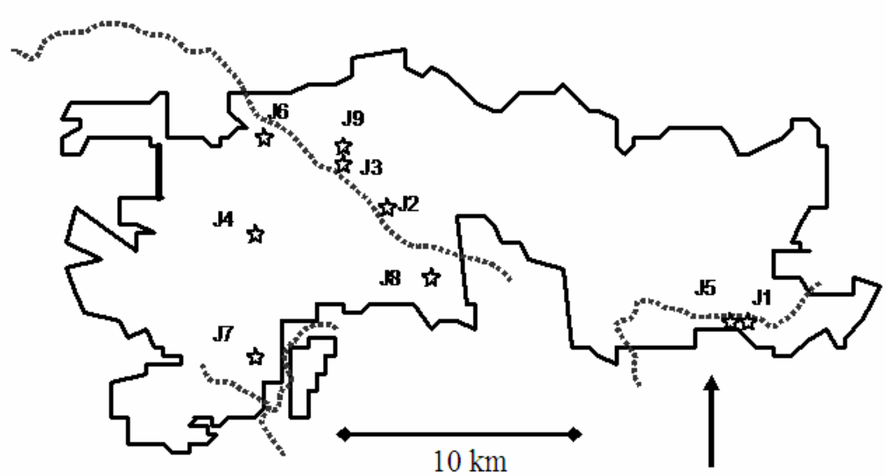


Figure 1—Sample sites of *Sequoia sempervirens* in the Jackson Demonstration State Forest, north coast of California. Sites are represented by stars and rivers are marked as dotted lines.

Table 1—*Data for nine sites across three* Sequoia sempervirens *site types, in Jackson Demonstration State Forest, north coast of California.*

Site	Stems sampled	Genets	Clones	SSG (#, percent stem total)	PD value	Mantel
J1	37	9	6	3 (8)	0.24	-0.41
J2	45	17	6	11 (24)	0.38	-0.42
J3	42	21	2	19 (45)	0.50	-0.11
J4	42	25	3	22 (52)	0.60	-0.14
J5	31	11	3	8 (25)	0.35	-0.45
J6	43	26	3	23 (53)	0.60	-0.34
J7	38	10	4	6 (16)	0.26	-0.63
J8	37	7	4	3 (8)	0.19	-0.59
J9	41	22	4	18 (54)	0.54	-0.22
Totals	356	148	35	113 (32)	NA	NA

On each site, up to 45 adjacent trees over 10 cm in diameter at breast height (dbh) were mapped, tagged, measured for dbh, and sampled for DNA. Sampling included clumped and forked stems that met the size requirement at forking point. With a target of up to 45 trees sampled per site, different stand densities resulted in a variation in total site areas. In a few instances we were unable to collect leaf material due to the extreme height or canopy structure of a tree.

Genetic Fingerprinting

DNA was extracted according to the Cullings (1992) modification of Doyle & Doyle (1987). Genetic AFLP fingerprints were generated and analyzed for each sampled stem according to the protocols described by Douhovnikoff and others (2004). Clones were then identified using a calibrated threshold methodology (Douhovnikoff and Dodd 2003).

Clonal Importance

To assess the relative importance of clonal spread on each site, sample percentage distinguishable (PD) values were calculated according to Ellstrand and Roose (1987). PD values are calculated as the number of genets identified, divided by the total number of stems genetically sampled. This metric of clonal diversity measures the percentage of genetically distinct sample trees.

The size of genets was measured in two ways. First, the total number of ramets per genet was tabulated to evaluate unit size. Second, aggregate basal areas per genet were calculated to serve as a surrogate for biomass. A limit of 45 sampled trees per site does not necessarily capture all ramets in each genet measured and thus these are minimum genet values.

Clone Spatial Structure

To test for spatial clustering of clones, we compared the clonal structure based on our threshold approach with the spatial pattern of samples at each of the field sites by means of Mantel tests (Manly 1991; Dodd and others 1998, 2000). For each of the field sites, Mantel R correlations were computed between Euclidean distance matrices calculated from the point coordinates for all sampled individuals and binary matrices, in which pairs of individuals assigned to the same clone were given the value one and all non-clone pairs were given the value zero. Significance of the correlations was tested by 1000 random permutations. Mantel R correlation coefficients were Z transformed to produce a parameter with an approximate normal distribution (Sokal and Rohlf 1995) and homogeneity among the Z-transformed Mantel R correlations for all site types was tested using a chi-square distribution.

Results

Clone Size

Out of 356 redwood stems genetically sampled over all nine sites, 113 were identified as single stem genets (SSGs). The remaining 243 were ramets from 35 multistem genets (clones). Clones ranged in size from two to 20 ramets with a mean of 6.7 ramets per clone (SE = 0.75). Ramets per genet are minimum estimates due to the 45 stem sampling limitation.

PD values are an inverse measure of the importance of clonal spread on a site. The smaller the value, the greater is the proportion of stems that are replicate ramets of clones. In this study, PD values were as low as 0.19, with an all-site mean of 0.43 (SE = 0.05).

On a whole genet basis, mean clone basal area per site 1.36 m^2 (SE = 0.26) was more than six times greater than mean SSG basal area per site (0.20 m^2, SE = 0.03). SSGs represented an average of 31 percent (SE = 0.05) of total basal area of all trees sampled *(table 2)*.

Clone Structure

Generally, clones formed partial "fairy-ring" structures surrounding large stumps. In several cases, however, genetically unique individuals were detected within the fairy-ring clones *(fig. 3d)*. Also, a wide range of additional structures were detected, including figure eights or chains of multiple overlapping rings as found on site J8 *(fig. 2a)*; concentric rings within larger rings as found on site J7 *(fig. 2b)*; disjunct clones with difficult to characterize patterns *(fig. 2c)*; and distantly disjunct clones with satellite clusters at distances as great as 40 m from the larger ring as found on site J5 *(fig. 2d)*.

Mantel correlations between spatial distances among stems on a site and assignment to clones ranged from –0.11 to –0.63. All correlations were negative, indicating a tendency for stems of the same genet to be clustered.

Table 2—*Percent of site total basal area represented by each clone and all single stem genets (singles).*

Site	Number of clones	Clone 1	Clone 2	Clone 3	Clone 4	Clone 5	Clone 6	Single stem genets (total percent)
		--*percent*---------------------------------------						
J1	6	37	15	14	10	8	2	14
J2	6	18	17	11	10	9	6	29
J3	2	62	4	na	na	na	na	34
J4	3	30	18	8	na	na	na	44
J5	3	31	21	18	na	na	na	30
J6	3	39	14	2	na	na	na	45
J7	4	32	24	15	9	na	na	20
J8	4	58	16	10	7	na	na	9
J9	4	13	12	10	8	na	na	57

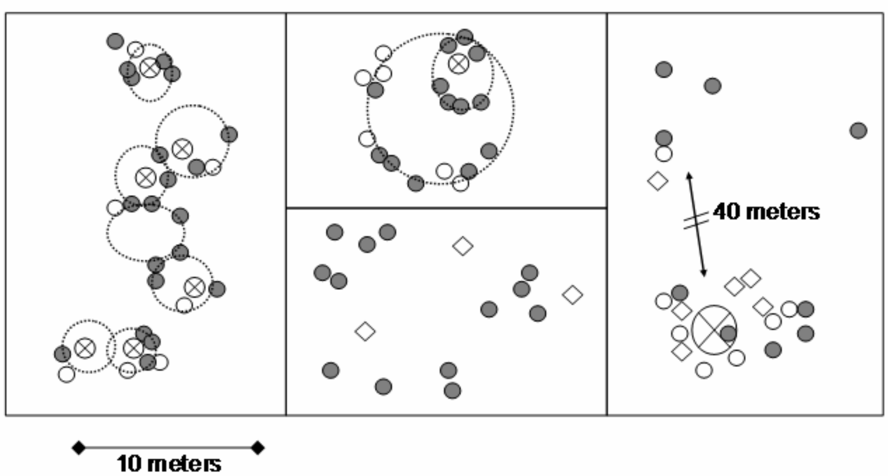


Figure 2—A selection of four genets that demonstrate the range of *Sequoia sempervirens* clonal structures found in the Jackson Demonstration State Forest, north coast of California, including (a) chains of overlapping rings on site J8, (b) concentric rings on site J7, (c) disjunct stems on site J1, and (d) distantly disjunct stems on site J5. Shaded circles represent ramets from a single clone. Open circles represent trees not sampled, open triangles (◊) represent single stem genets, crossed circles represent stumps, and dashed circles outline perceived fairy-rings.

Discussion

Clonal Spread and Clone Structure

Clonal spread plays an important part in regeneration of redwood stands. We found that on average 59 percent of the stems at a site resulted from clonal spread, with a high of 81 percent and a low of 40 percent. (These are minimum values, and it is possible that 100 percent of the stems are clonal copies of predating stems that no longer exist.) The potential for high numbers of ramets (genetic copies) is commonly overlooked in ecological studies, but our results emphasize the need to consider this mode of regeneration in clonal species such as redwood.

Genet sizes are skewed by several SSGs and by a few disproportionately large clones with as many as 20 stems. While the second growth sites we sampled ranged in size from 1000 m^2 to 3000 m^2 (measured by the outer drip zone), two to six genets contained the great majority of stems, and distinct genets were only about one sixth of the total number of stems. The skew in ramet number per genet is likely a result of the unevenness of disturbance events that open a window of opportunity for sprout release. For example, a tree that has experienced disturbance (windfall, burned, cut, etc.) may clear an opening and promote sprouting at its base and possibly along its fallen stem resulting in a large numbers of ramets in a particular clone relative to undisturbed genets. Regardless of the cause of this imbalance it is most significant that a few genets monopolize a site as large clones. This has the following interesting ecological and evolutionary implications:

(1) Resource use— If all ramets draw upon resources equally, larger clones with more ramets occupy more area and use a greater proportion of site resources. It is still unknown, but if redwood ramets also remain physically integrated, within genet resource sharing is possible. Therefore, large redwood clones not only have a proportionately larger footprint in terms of resource use, but they may also be taking advantage of economies of scale and greater access to resources.

(2) Site persistence—Large redwood clones may have important long-term advantages in maintaining site representation and dominance. For example: (i) Clones are genetic replicates of locally successful genotypes giving them a competitive edge over newcomers that are subject to the trial-and-error involved in sexual reproduction. (ii) Little seasonality is associated with sprouting, allowing for a quick response to resource availability. Clones have access to an already established root system, grow much faster than seedlings, and are not susceptible to damping off (Olsen and others 1990). (iii) The risk of mortality for a genet can be spread among its many ramets (Cook 1983, Eriksson and Jerling 1990). (iv) Ramet clumping diversifies stem location and can produce local characteristics that buffer some ramets from damage (Peterson and Jones 1997). (v) With no known biological limit to the number of ramet generations over time that can be produced vegetatively, a successful redwood genotype is locally persistent and theoretically immortal.

(3) Greater genetic representation—Large redwood genets have greater genetic representation on a site with more stems, pollen, and seeds—all are important factors driving evolutionary potential. When regeneration by seed is possible, larger clones have a greater likelihood of being a genetic contributor to the cohort.

Our results show that redwoods both sprout on a local scale and can spread horizontally through lateral ramet multiplication. For example ramets from the same clone were found to be up to 40 m apart from each other. While in many cases fairy-rings were a standard unit of clonal structure, on a larger scale clones showed a wide range of shapes. Little is known about how redwood clonal spread occurs beyond local basal sprouting. While fairy-rings, concentric circles, and figure eights or longer chains can be explained by repeated basal sprouting of new ramets, it is still unclear how disjunct structures come about. It is possible that falling trees bury branches that then sprout (Rogers 2000), or that soil disturbance, such as in the use of harvesting equipment, may expose and promote sprouting in roots (Weber 1990, Lavertu and others 1994). An example of this may be the disjunct and somewhat chaotic pattern found on site J1 *(fig. 2c)*. This is an important area for further research.

Management Implications

The implications of these results can be important and wide reaching in redwood forest management and conservation. Some examples include: i) With significant clonal growth, seedling establishment may be a rare event and, in the short term, may contribute little to site regeneration (Rogers 2000). However, in the long-term, rare seedling recruitment may be sufficient to maintain high levels of genetic diversity (Bond and Midgley 2001). ii) Resprouting and the fact that the genotype is often represented by more than a single stem, can minimize the impact of harvesting on a population's gene pool. This is contingent on the maintenance of conditions that promote sprouting and adequate sprout survival. iii) Redwood ramets may be biologically and physically integrated and manipulation of a ramet may have an effect on its sister ramets. For example there may be a minimum number of sprouts required to sustain the root system associated with a stump at a particular point in clone development (Olsen 1990). Ramet thinning is known to strongly promote height growth in dominant ramets and is strongly correlated with residual stand density (Oliver and others 1994), and ramet removal can decrease physical stability of remaining ramets (Peterson and Jones 1997).

Remarkably little is known of the importance and structure of coast redwood clonal growth at the stand scale. This, combined with earlier technical limitations of accurately measuring clones has resulted in many researchers and forest managers having to use a one stem or one fairy ring per genet assumption in their work. By using molecular methods for identification of clone size, we have shown that such an assumption can result in overestimates of up to 500 percent in genet numbers on a stand scale, and in some cases underestimates diversity within fairy-rings. This work demonstrates that a redwood tree as a genetic organism has a significant horizontal component that is overlooked when the single vertical stem is used as the working unit. In order to better understand stand structure and dynamics and to assess the impact of management practices on redwood stands, this horizontal dimension should be taken into consideration.

References

Bond, W.J.; Midgley, J.J. 2001. **Ecology of sprouting in woody plants: the persistence niche**. Trends in Ecology and Evolution 16(1): 45–51.

Cook, R.E. 1983. **Clonal plant populations**. American Scientist 71: 244-253.

Cullings, K. 1992. **Design and testing of a plant-specific PCR primer for ecological and evolutionary studies**. Molecular Ecology 1: 233-240.

Dodd, R.S.; Afzal-Rafii, Z.; Bousquet-Mélou, A. 2000. **Evolutionary divergence in the pan-Atlantic mangrove *Avicennia germinans***. New Phytologist 145: 115-125.

Dodd, R.S.; Afzal-Rafii, Z.; Power, A.B. 1998. **Ecotypic adaptation in *Austrocedrus chilensis* in cuticular hydrocarbon composition**. New Phytologist 138: 699-708.

Douhovnikoff, V; Cheng, A.M.; Dodd, R.S. 2004. **Incidence, size and spatial structure of clones in second growth stands of coast redwood, *Sequoia sempervirens* (Cupressaceae)**. American Journal of Botany 91: 1140-1146.

Douhovnikoff, V.; Dodd, R.S. 2003. **Intra-clonal variation and a similarity threshold for identification of clones: application to** *Salix exigua* **using AFLP molecular markers**. Theoretical and Applied Genetics 106: 1307-1315.

Doyle, J.J.; Doyle, J.L. 1987. **A rapid DNA isolation procedure for small quantities of fresh leaf tissue**. Phytochemical Bulletin 19: 11-15.

Ellstrand, N.C.; Roose, M.L. 1987. **Patterns of genotypic diversity in clonal plant species**. American Journal of Botany 74: 123-131.

Eriksson, O.; Jerling, J. 1990. **Hierarchical selection and risk spreading in clonal plants.** In: de Kroon, H.; van Groenendael, J., eds. Clonal growth in plants. The Hague, Netherlands: SPB Academic Publishing; 79-94.

Hepting, G.H. 1971. **Diseases of forest and shade trees of the United States**. Agric. Handb. 386. Washington, DC: U.S. Department of Agriculture; 658 p.

Lavertu, D.; Mauffette, Y.; Bergeron, Y. 1994. **Effects of stand age and litter removal on the regeneration of *Populus tremuloides***. Journal of Vegetation Science 5: 561-568.

Manly, B.F.J. 1991. **Randomization and Monte Carlo methods in biology**. London: Chapman & Hall.

Oliver, W.W.; Lindquist, J.L.; Strothmann, R.O. 1994. **Young-growth redwood stands respond well to various thinning intensities**. Western Journal of Applied Forestry 9: 106-112.

Olsen, D.F.; Roy, D.F.; Walters, G.A. 1990. ***Sequoia sempervirens* (D. Don) Endl. redwoods**. In: Burns, R.M.; Honkala, B.H., eds. Silvics of North America. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture; 541-551.

Peterson, C.J.; Jones, R.H. 1997. **Clonality in woody plants: a review and comparison with clonal herbs**. In: de Kroon, H.; van Groenendael, J., eds. The ecology and evolution of clonal plants. Leiden, Netherlands: Backhuys Publishers; 263–290.

Rogers, D.L. 1994. **Spatial patterns of allozyme variation and clonal structure in coast redwoods (*Sequoia sempervirens*)**. Berkeley, CA: University of California. Ph.D. dissertation.

Rogers, D.L. 2000. **Genotypic diversity and clone size in old-growth populations of coast redwood (*Sequoia sempervirens*)**. Canadian Journal of Botany 78: 1408-1419.

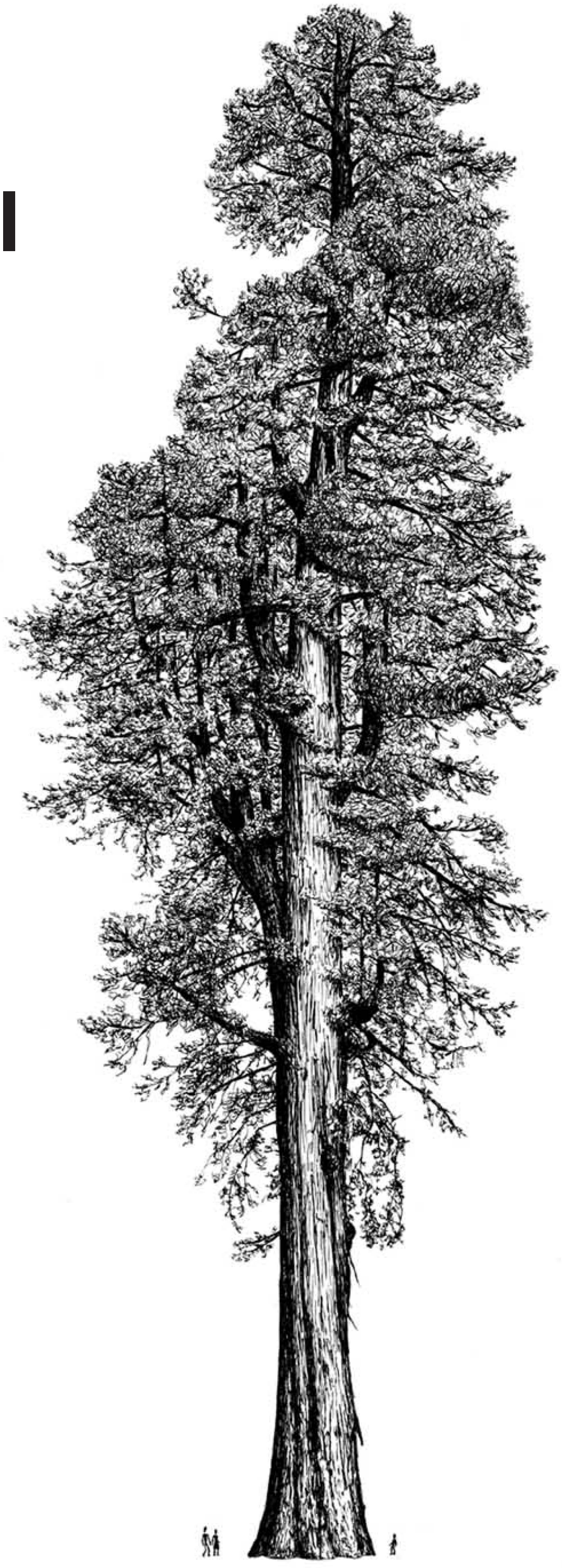
Rydelius, J.A.; Libby, W.L. 1993. **Arguments for redwood clonal forestry**. In: Ahuja, M.R.; Libby, W.J., eds. Clonal forestry II: conservation and application. Berlin, Germany: Springer; 158-168.

Sokal, R.R.; Rohlf, F.J. 1995. **Biometry**. New York: W. H. Freeman and Co.

Weber, M.G. 1990. **Response of immature aspen ecosystems to cutting and burning in relation to vernal leaf-flush**. Forest Ecology & Management 31: 15-33.

SESSION 3

Water and Watersheds II



A Preliminary Study of Streamside Air Temperatures Within the Coast Redwood Zone 2001 to 2003[1]

Tracie Nelson,[2] Richard Macedo, and Bradley E. Valentine

Abstract

Timber harvest practices must address potential impacts to aquatic and riparian habitats. Stream shading and cool water temperature regimes are important to protect stream-dwelling organisms. We are examining riparian temperature regimes within the coastal redwood area of Mendocino County. Summer temperature gradients are being characterized along fifteen transects set perpendicular to watercourses within four different watersheds. Data loggers record temperature along each transect at pre-determined spatial intervals from the watercourse midpoint. With the exception of a control transect, all watercourses in this study are subject to near-future timber operations. This study is part of a larger program to detect changes in riparian microclimate as a result of timber harvest. Here we examine the relationship between temperature and physiography, and explore different methods of analysis in order to characterize pre-harvest temperature gradients. As with similar studies, we find upslope temperatures increase with distance from the watercourse. The greatest temperature increase per unit distance occurred between the watercourse mid-point station and the 25ft stations on either side. Our analysis indicates the calculated maximum weekly average temperatures (MWAT) may be the best graphical indicator of overall riparian temperature regimes, while the mean maximum weekly temperatures (MMWT) appear more sensitive to physiographic characteristics.

Key words: microclimate, riparian, temperature, timber harvest

Introduction

Few studies have focused on riparian microclimate gradients in California's coastal redwood region (Ledwith 1996). Timber harvesting near watercourses, resulting in changes to canopy structure and composition has the potential to alter water temperature and microclimate regimes of adjacent stream-side areas, especially during warm summer periods (Beechie and others 2000; Brosofske and others 1997; Brown and Krygier 1970; Chen 1991, Chen and others 1992, 1993a, 1993b, 1995; Corner and others 1996, Erman and others 2000, Independent Multidisciplinary Science Team 1999, O'Connell and others 2000, Spence and others 1996). Reduced shade leads to increased water temperatures, which can impair important life requisites and reduce survival rate of salmonids during adult upstream migration, juvenile rearing, and downstream migration of smolts (NMFS 2000). Nearly all of California's salmonid species inhabiting the coastal redwood zone are Federally and/or State listed as threatened or endangered (DFG 2004), thus requiring protective

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Environmental Scientist, California Department of Fish and Game, 7329 Silverado Trail, Napa, CA 94558. email: tnelson@dfg.ca.gov

measure to be implemented to mitigate the effects of timber harvest related impacts. While changes in biological variables resulting from intensive timber harvest are often difficult to document, changes in air temperature, relative humidity and wind speed appears highly responsive (Chen 1990). Our long term plans are to examine the effects of timber harvest on summer stream temperature and riparian microclimate regimes. In this study, we focus on the first three years of data, documenting temperature gradients across riparian zones prior to timber harvest in coastal Mendocino County. The secondary goal of this research is to analyze the best method of characterizing summer air temperature gradients within and upslope of riparian buffer zones, using the different calculated values of the maximum weekly average temperature (MWAT) and mean maximum weekly temperature (MMWT).

Study Sites

Watercourses selected for this study are within western Mendocino County and discharge into the Pacific Ocean between the towns of Fort Bragg and Mendocino *(fig. 1)*. With the exception of a control transect on Russian Gulch within the Russian Gulch State Park, all watercourses in this study are within the Jackson Demonstration State Forest (JDSF). We collected data from the following watersheds: North Fork of the South Fork Noyo River (NFSFN) including a tributary to Brandon Gulch, upper Parlin Creek (tributary to the South Fork Noyo River), Caspar Creek, and Russian Gulch. Stream channel elevations ranged from approximately 356 m (840 ft: Parlin Creek; Transect F) to approximately 49 m (160 ft: Casper Creek; Transect #2). Distance to ocean ranges from 19 km (12 miles: Parlin Creek - Transect F), to four km (2.5 miles; Russian Gulch). Orientation of specific stream reaches varied.

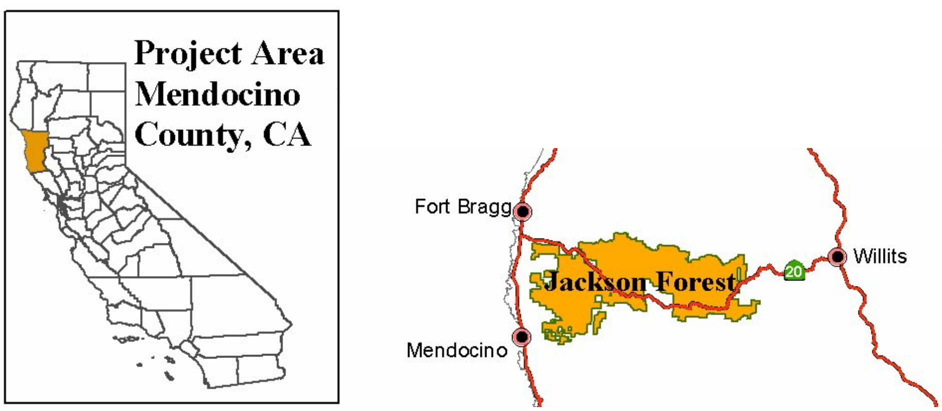


Figure 1—Overview of Jackson Demonstration State Forest (JDSF). Study area is within JDSF, with one control transect located north of Mendocino at Russian Gulch State Park.

Most precipitation in this region occurs in winter. Summer periods are usually warmer and without rainfall. Summer fog commonly intrudes into some study sites and precipitation may occur in the form of fog-drip. Mean annual precipitation at Caspar Creek from 1962 to 1997 was 1,190 mm, with a range of 305 to 2,007 mm. (Henry 1998). Average winter air temperatures are approximately 6.5 °C with minimums reaching 4.5 °C. Average summer air temperatures are approximately 15.5 °C, with average maximums reaching 22.5 °C. Summer discharges at study streams vary from dry to less than 0.5 ft^3/second.

Overstory vegetation consists of mature second growth coast redwood *(Sequoia sempervirens)*, Douglas-fir *(Pseudotsuga menziesii)*, western hemlock *(Tsuga heterophylla)*, and grand fir *(Abies grandis)*. Other notable overstory species include tanoak *(Lithocarpus densiflorus)*, and red alder *(Alnus rubra)*. Understory vegetation includes evergreen and red huckleberry *(Vaccinium ovatum* and *V. parvifolium)*, Pacific rhododendron *(Rhododendron macrophyllum)*, sword fern *(Polystichum munitum)*, and poison oak *(Toxicodendron diversilobum)*. In-stream animal species occupying study streams include Pacific giant salamander *(Dicamptodon ensatus)*, coho salmon *(Oncorhynchus kisutch)*, steelhead *(O. mykiss)*, sculpin *(Cottus sp.)*, and threespine stickleback *(Gasterosteus aculeatus)*. In addition, numerous aquatic invertebrate species are dependant upon streams and adjacent riparian zones.

Methods

We are monitoring summer (May through September) temperature gradients along fifteen transects oriented perpendicular to stream channels. Depending on the watershed, transect location was used to document microclimate conditions along a given reach or to assess differences between units planned for timber harvest and units planned for timber retention. Along each transect, we established up to eleven sampling stations; one at mid-channel with a data logger suspended above the stream channel and another submersed underwater on or near the transect, and the other stations running up each adjacent hillslope (two at each distance) at 7.6 m (25 ft), 22.9 m (75 ft), 45.7 m (150 ft), and up to 91.4 m (300 ft) *(fig. 2)*. For sites where timber has been marked for future harvesting, we established additional stations in areas where trees will be removed and in areas where groups of trees will be retained following harvest.

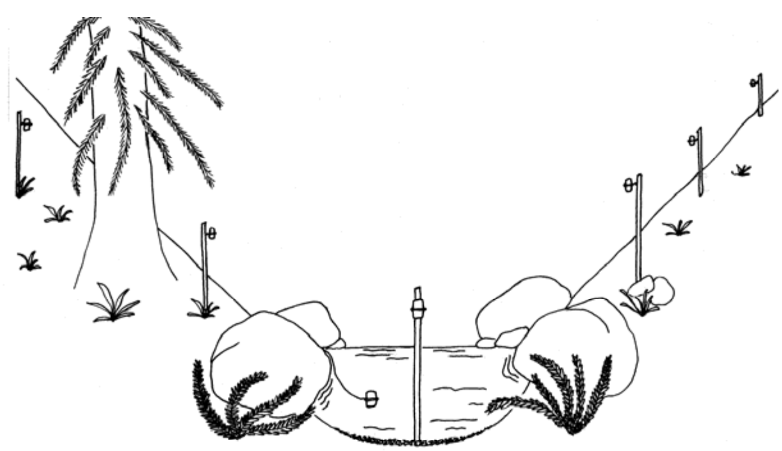


Figure 2—Schematic depicting a transect with temperature data loggers set at mid-channel, 25 ft, 75 ft, and 150 ft on either side of the watercourse. A submersed instream unit is also shown. (Figure by Sheryl Greene 2002)

Of the fifteen transects, eight were established in upper reaches of Parlin Creek, two in the NFSF Noyo drainage, and four in the SF Casper area. A control transect, where no timber harvesting will occur, was established at Russian Gulch within Russian Gulch State Park. Study sites were chosen based on the type and timeframe of timber operations planned for those areas.

For the NFSF Noyo and Parlin Creek sites, we used HOBO® H8 data loggers from Onset Computer Company (OCC) to measure and record temperature at each station. The manufacture's specified accuracy is ± 0.7 °C. These were programmed to record temperature at 1 hour intervals from late spring to mid-autumn. All H8 data loggers were housed in OCC's white submersible cases. Terrestrially-located loggers were wired to single wooden stakes measuring 91.4 cm long (three ft), 3.8 cm wide (1.5 inches), and 0.8 cm thick (5/16 or 0.3 inches). Loggers were positioned so case bottoms were 61 cm (two ft) above the ground or water surface. Styrofoam cups, with cut-out ventilation holes, were placed over the casings to insulate them from direct solar radiation.

For the Caspar Creek and Russian Gulch sites, we used HOBO®Pro temperature/relative humidity loggers from OCC to measure air temperature and relative humidity at each station. Humidity data was not analyzed for this paper. The manufacture's specified accuracy is ± 0.2 °C. These were programmed to record conditions at 1 hour intervals over 24 hours per day. Pro loggers were housed in OCC's white HOBO® solar radiation shields. Each logger was wired to two wooden stakes each measuring 122 cm (four ft) long, 3.8 cm wide (1.5 inches), and 1.6 cm thick (10/16 or 0.6 inches). Pro loggers were positioned so shield bottoms were 75 cm (2.5 ft) above the ground or water surface.

Two metrics of temperature were examined for each station along the transect for each year. The maximum weekly average temperatures (MWAT) was calculated as a moving average temperature for a 168 hour period (seven days) one-hr reading using that interval's value, the preceding 83 values, and the subsequent 84 values (3.5 days). The maximum value of these seven-day moving averages is the MWAT. The mean maximum weekly temperature (MMWT) was calculated using daily maximum temperature values. The MMWT was calculated by averaging each day's value with those of the preceding and subsequent three days (seven values). For each station, the maximum of the seven-day running average maximum temperatures is the MMWT. To assess temperature gradient differentials between mid-channel and upland stations, we used the mathematical differences between the mid-channel and the upland MWAT and MMWT within the same transect in the same year.

Shade, overstory canopy, and aspect were measured at each station to determine their influence on temperature measurements. Shade, or the percentage of available solar energy blocked at each station by canopy and topography, was measured. A Solar Pathfinder® using the sun arc for August, the month stream temperatures in the area commonly peak, was employed for this purpose. Overstory canopy was also measured by using a vertical sighting tube at 49 points situated at and around each station along four 24-meter transects centered on the data logger. At each point, canopy was scored as present when the sighting tube encountered canopy. Thus, the canopy at each data logger is indexed to a possible range from zero (no canopy) to 49 (full canopy). The resulting data is useful in mapping "holes" in the canopy. Aspect was measured with a handheld compass at each station.

All transects were established in 2001 after the May 1 summer season began.

Thus, some of the data represents only two of the three years of data, where we are not confident we captured the periods of highest temperature intensity in 2001.

Results

On average, temperatures gradually increased with increasing upslope distance from the watercourse midpoint. MWAT and MMWT values generally escalated with increasing distance from the transect's mid-channel station. At 7.6 m (25 ft), MWAT values averaged 0.8 °C higher than adjacent channel midpoint MWAT values for the same year. The MMWT values averaged 2.5 °C higher than the mid-channel MMWT for the same transect. At the 22.9 m (75 ft) stations, MWATs averaged 1.2 °C, while MMWTs averaged 2.9 °C above mid-station values. Between the watercourse and the 45.7 m (150 ft) stations, MWATs averaged 1.5 °C and MMWTs 3.5 °C higher. For the transects that had stations between 70.1 m (230 ft) and 91.4 m (300 ft), MWATs averaged 2.0 °C higher, and MMWTs averaged 4.1 °C than transect mid-channel values *(fig. 3)*.

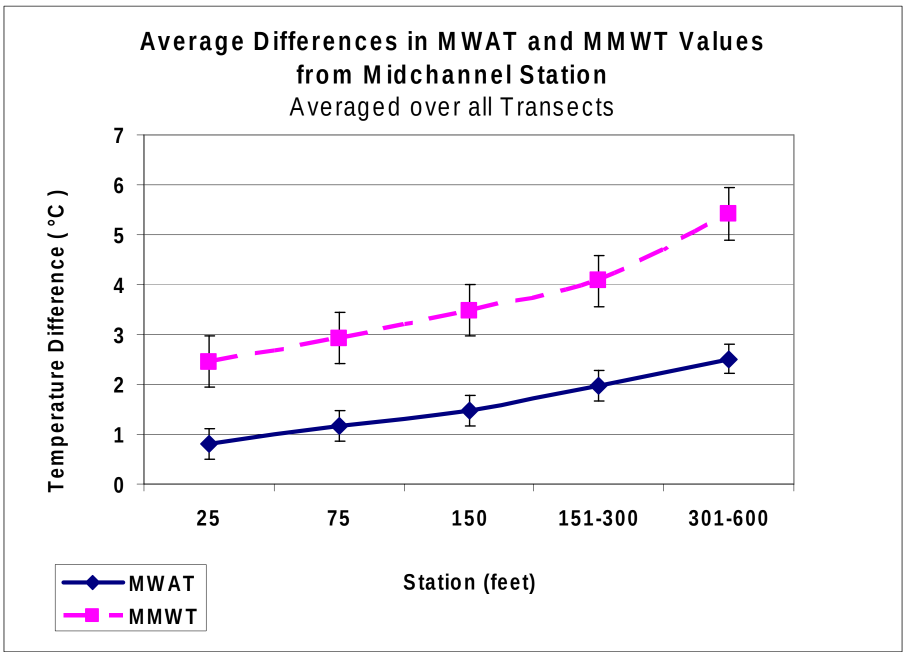


Figure 3—Average differences in the station's MWAT and MMWT values minus the MWAT and MMWT values of the same transects' mid-channel station, respectively.

The greatest temperature increase per unit of distance occurred between the mid-channel station and the 25 ft stations on either side. Average MWAT value at the 7.6 m (25 ft) station was 0.81 °C higher than the mid-channel station, giving the slope of the temperature gradient a (0.81/7.6) = 0.11 °C/m value. Subsequent temperature gradients between stations decrease. From the 7.6 m (25ft) unit to the 22.9 m (75ft) unit, the MWAT gradient averaged (1.17-0.81)/(22.9-7.6) = 0.02 °C/m. The 22.9 m (75 ft) to the 45.7 m (150 ft) gradient has an average slope of 0.01 °C/m. Not every transect included a unit above 45.7 m, thus these slope values were not calculated. MMWT temperature gradient values were 0.32 °C/m, 0.03 °C/m and 0.02 °C/m for 7.6 m, 22.9 m and 45.7 m respectively.

The MWAT and MMWT values were calculated for each station each year. Data used in our analysis represent the highest MWAT and MMWT values achieved for that station in that year. Mathematical differences between MWATs and MMWTs of the upslope stations and the MWATs and MMWTs of the mid-channel station of the same transect in the same year were calculated *(fig. 4)*.

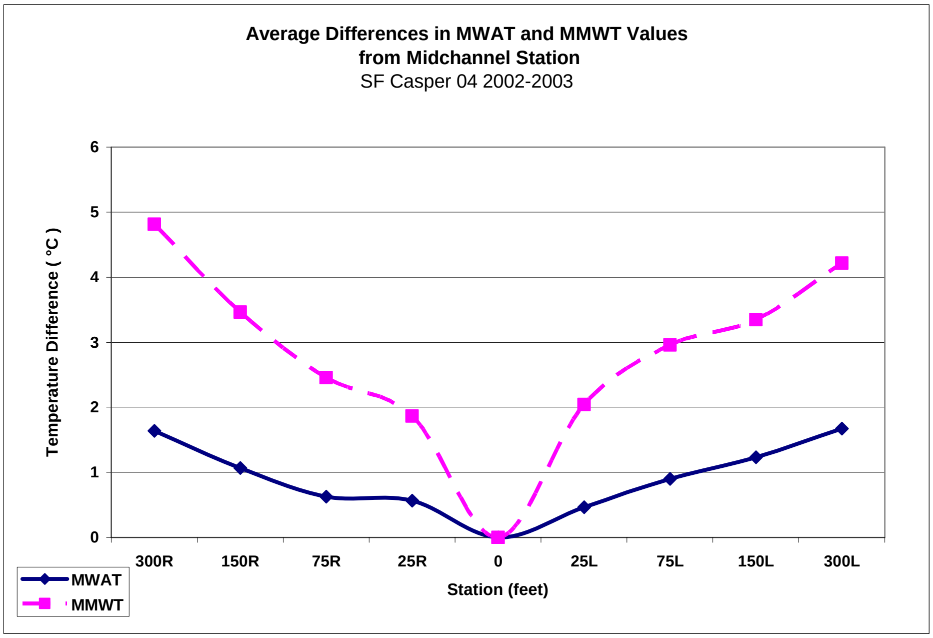


Figure 4—Two seasons of pre-harvest data for one transect. Graphed values are the MWAT and MMWT values at that station, minus the MWAT and MMWT values at the mid-channel (0 ft) station. These differences are then averaged over 2 seasons. Station numbers represent distances from the mid-channel station as well as station orientation [right side (“R”) or left side (“L”), facing downstream].

Observing individual transect graphs, MWAT values generally increased with distance from the watercourse mid-point station. However, in six of fourteen transects, this pattern was slightly altered, with one downslope station measuring slightly higher than its adjacent upslope neighbor. All six MWAT values are within the specified accuracy ranges of the temperature data loggers. MMWT values also generally increased with distance from the mid-channel station, but in eleven of fourteen transects, the gradually increasing pattern was disrupted. On one or both sides of the transect, one downslope unit measured higher temperatures than its upslope adjacent neighbor(s) *(fig. 5)*. These are not all within the accuracy ratings for the units. Instead, these ‘atypical’ patterns may be linked to physiography. Canopy, aspect, or environmental placement (such as in a hollow stump or near a road) may influence the temperatures measured at these stations. MMWT values appear to be more sensitive to these physiological characteristics than do MWAT values.

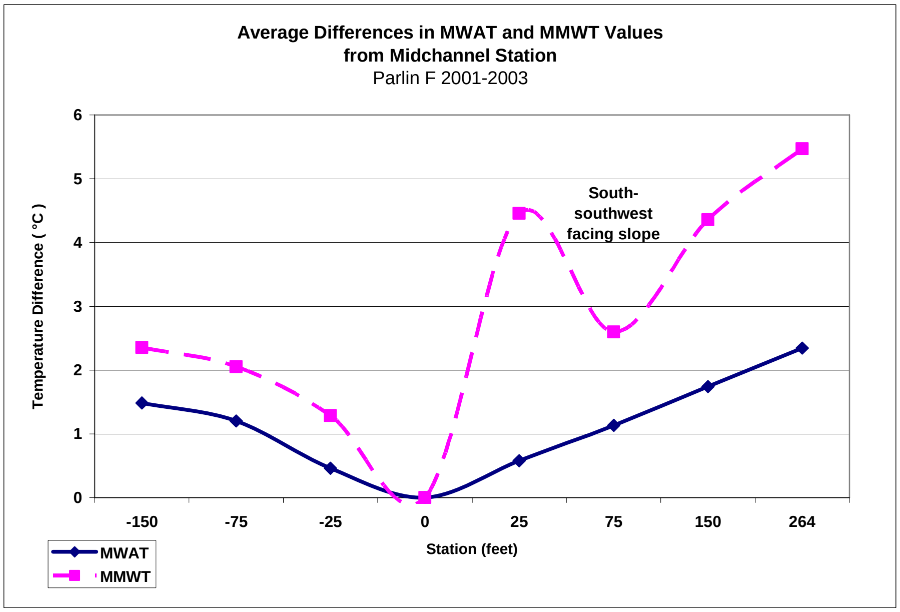


Figure 5—Three seasons of pre-harvest data for one transect. Graphed values are the MWAT and MMWT at that station, minus the MWAT and MMWT values at the mid-channel (0 ft) station. These differences are then averaged over 3 seasons. The station number refers to the data logger's distance from the mid-channel station, while + designates a station in a (future) harvest area and – designates a station in a non-harvest area.

The fourteen transects were comprised of a graphed total of 116 units. Of these, 19 downslope units measured higher temperatures, and thus had a greater difference in their MMWT values from their midpoint stations, when compared to their upslope adjacent neighbors. This represents approximately 16 percent of the units. Of the19 units, 13 were on south-facing slopes. This represents approximately 68 percent of the units with atypical relationships. Also 13 of the 19 units had a canopy index equal to or less than 40 (out of 49), and/or a shade measurement of 90 percent or less. Canopy or shade measurements at or below these values usually indicate there is nearby opening or "hole" in the canopy. These are not necessarily the same 13 units as those on south facing slopes, although there is much overlap. Four of the 19 stations receive some measure of direct solar exposure due to being on or at the edge of a road or forest opening. One of the 19 units consistently measuring lower MMWT values than its downslope neighbor arbitrarily resides in a hollow stump, sheltered from direct exposure to wind, heat vectors and solar radiation.

Discussion

To achieve aquatic resource conservation, timber harvest activities must address potential (significant) impacts including reduced canopy, increased summer air temperatures, and reduced humidity. For aquatic habitats, management should address stream shading and conservation of cool water temperature regimes (Welsh and others 2001). Increases in air temperature may promote attendant increases in evapotranspiration rates of riparian vegetation, which could reduce streamflows and extend the dry period for intermittent streams (Ledwith 1996).

As our data suggests, pre-harvest air temperatures rise as distance from mid-channel to upslope areas increase. This is in agreement with similar studies. In Sierra Nevada basins with undisturbed canopy, near stream air temperatures were significantly lower than adjacent uplands (Erman and others 2000). Before timber harvest, stream and riparian areas in Washington are generally characterized by cool air and soil temperatures, high humidity, and low wind speed relative to upland forest interior conditions (Chen and others 1999). Ledwith (1996) found that air temperatures at streams increased with decreasing buffer width in even-aged managed forests in Northwestern California.

The decision to undertake this study was primarily based on the scarcity of information specific to the North Coast redwood region. The region is highly influenced by ocean air currents and seasonal fog conditions, unlike conditions in central Washington or the Sierra Nevada range. The long term goal of this study will be to track potential changes in microclimate in response to un-even aged management practices. Timber harvest practices vary by forest type and are regulated differently by each state. The California Forest Practice Rules (CFPRs) regulate timber harvest activities in California. Several sections of the CFPRs are dedicated to protect aquatic and riparian areas from impacts caused by timber harvest. Despite this, there is significant debate over the size and structure of stream-side riparian zones for buffering harvest related impacts. Key elements defining riparian zones include topography, surface water, soils, vegetation and microclimate (O'Connell and others 2000). To determine the optimum buffer width (one that protects the integrity of the riparian ecosystem while allowing profitable timber harvest), researchers have examined riparian processes as a function of distance from stream channels (Ledwith 1996). Our long term efforts are to examine the microclimate of riparian buffer zones in the coast redwood region, and to characterize changes in microclimate and water temperature due to uneven-aged harvest management as practiced under the CFPRs.

Conclusions

Our preliminary study focused on air temperature regimes across riparian and upslope areas of mature second growth coast redwood forests. In agreement with similar studies, we found that upslope temperatures increased with increasing distance from the watercourse. We also found that physiography can influence these regimes. Temperature gradients are steepest between the watercourse and the 7.6 m (25 ft) stations, and decrease as distance upslope increases. MWAT values are less sensitive to physical characteristics than are MMWT values. MWAT values appear to reflect overall temperature gradients, while MMWT values reflect temperature gradients relative to physiography. The sensitivity of MMWT values may make them more valuable in determining changes in microclimate resulting from timber harvest practices. We plan on continuing to collect summer microclimate data in these areas, both before and after timber is harvested. Humidity monitoring and analysis will be added to the long term study. We also hope to determine the influences of canopy, solar radiation, aspect, and slope on temperature and humidity gradients. Changes in temperature and humidity regimes, as well as changes in canopy and forest structure, will be analyzed to document the magnitude of changes due to un-even aged timber harvest practices in a coastal-influenced area.

References

Beechie, T.J.; Pess, G.; Kennard, P.; Bilby, R.; Bolton, S. 2000. **Modeling recovery rates and pathways for woody debris recruitment in northwestern Washington streams.** North American Journal of Fisheries Management 20: 436-452.

Brosofske, K.D.; Chen, J.; Naiman, R.J.; Franklin, J.F. 1997. **Harvesting effects on microclimate gradients from small streams to uplands in western Washington**. Ecological Applications 7: 1188-1200.

Brown, G.W.; Krygier, J.T. 1970. **Effects of clear-cutting on stream temperature.** Water Resources Research 6 (4): 1133-1139.

California Department of Fish & Game (DFG). 2004. **Federal and State Endangered Species Act Status for California Anadrouous Fish as of August 6, 2004.** California. *See* http://www.dfg.ca.gov/nafwb/index.html

Chen, J. 1990. **Microclimatic pattern and basic biological responses at the clearcut edges of old-growth Douglas-fir stands.** The Northwest Environmental Journal 6(2): 423-425.

Chen, J. 1991. **Edge effects: microclimate pattern and biological responses in old-growth Douglas-fir forests.** Seattle, WA: University of Washington; Ph.D. thesis.

Chen, J.; Franklin, J.F.; Spies, T.A. 1991. **Vegetation responses to edge environments in old-growth Douglas-fir forests.** Ecological Applications 2(4): 387-396.

Chen, J.; Franklin, J.F.; Spies, T.A. 1993a. **An empirical model for predicting diurnal air-temperature gradients from edge into old-growth Douglas-fir forest.** Ecological Modelling 67: 179-198.

Chen, J.; Franklin, J.F.; Spies, T.A. 1993b. **Contrasting microclimates among clearcut, edge, and interior of old growth Douglas-fir forest.** Agriculture and Forest Meteorology 63: 219-237.

Chen, J.; Franklin, J.F.; Spies, T.A. 1995. **Growing-season microclimatic gradients from clearcut edges into old-growth Douglas-fir forests.** Ecological Applications 5(1): 74-86.

Chen, J.; Saunders, S.C.; Crow, T.R.; Naiman, R.J.; Brosofske, K.D.; Mroz, G.D.; Brookshire, B.L.; Franklin, J.F. 1999. **Microclimate in Forest Ecosystem and Landscape Ecology.** BioScience 49(4): 286-297.

Corner, R.A.; Bassman, J.H.; Moore, B.C. 1996. **Monitoring timber harvest impacts on stream sedimentation; instream vs. upslope methods.** Western Journal of Applied Forestry 11(1): 25-32.

Erman, D.C; Erman, N.A.; Drury, I.; Greathouse, E.A; Ransom, B.O. 2000. **Testing variability of riparian temperatures in Sierra Nevada stream basins.** Tahoe National Forest Challenge Cost Share Agreement CCS-05-97-17-019.

Henry, N. 1998. **Overview of the Caspar Creek Watershed Study.** In: Ziemer, R.S., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek Story; 1998 May 6 1998; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 149 p.

Independent Multidisciplinary Science Team. 1999. **Recovery of wild salmonids in western Oregon forests: Oregon Forest Practices Act Rules and the measures in the Oregon Plan for Salmon and Watersheds.** Technical Report 1999-1 to the Oregon Plan for Salmon and Watersheds, Governor’s Natural Resources Office, Salem, OR. 92 p.

Ledwith, T.S. 1996. **The effects of buffer strip width on air temperature and relative humidity in a stream riparian zone.** Humboldt, CA: Humboldt State University; 88 p. M.S. thesis.

National Marine Fisheries Service, National Oceanic and Atmospheric Administration (NMFS). 2000. **Salmonid Guidelines for Forestry Practices in California.** California. *Available at* http://www.krisweb.com/biblio/gen_nmfs_nmfs_2000_sgfpc.htm

O'Connel, M.A.; Hallett, J.G.; West, S.D.; Kelsey, K.A.; Manuwal, D.A.; Pearson, S.F. 2000. **Effectiveness of riparian management zones in providing habitat for wildlife.** Final Report to the Washington Timber, Fish and Wildlife Program; TFW-LWAG-00-001.

Spence, B.C.; Lomnicky, G.A.; Hughes, R.M.; Novitzki, R.P. 1996. **An ecosystem approach to salmonid conservation.** ManTech Environmental Research Services Crop, Corvallis, OR. Available from the National Marine Fisheries Service. Portland, OR; TR-4501-96-6057.

Welsh, H.H.; Hodgson, G.R.; Harvey, B.C. 2001. **Distribution of juvenile coho salmon in relation to water temperature in tributaries of the Mattole River, California.** North American Journal of Fisheries Management 21(3): 464-470.

Effects of Timber Harvest on Fog Drip and Streamflow, Caspar Creek Experimental Watersheds, Mendocino County, California[1]

Elizabeth Keppeler[2]

Abstract

Within the second-growth redwood forest of the Caspar Creek watershed, fog drip was measured in 1998 at 12 sites where heavy fog drip was expected. The following year, two one-ha plots were each instrumented with six randomly sited 1.35 m^2 fog-drip collectors and one additional collector in a nearby clearcut. Fog-drip totals were highly variable, ranging from zero to 18 mm per event (mean = eight mm) and 0.2 to 99 mm (mean = 39 mm) during the 1999 season. Within the one-ha plots, fog drip under the canopy was only three mm greater than accumulations in the clearcut at one site and not significantly different at the other. Given the annual average precipitation of 1170 mm, fog drip does not appear to strongly influence groundwater recharge and baseflow processes at Caspar Creek. These results are consistent with streamflow measurements. From 1971 to 1973, 65 percent of the timber volume in the South Fork was selectively cut, and from 1985 to 1992, 50 percent of the North Fork basin was clearcut. Annual water yield and summer flows increased following both timber harvests, indicating that the effects of reduced rainfall interception and transpiration exceed that from the loss of fog drip.

Key words: fog drip, hydrologic processes, streamflow, timber harvest

Introduction

The summer fog that blankets California's north coast is an ecologically important characteristic of the region as it moderates temperatures and brings water to the plants and animals occupying this seasonally arid landscape. The coast redwood is limited to a narrow belt between central California and southern Oregon in large part because of the prevalence of summer fog. Advection fog forms when westerly winds move warm moist Pacific air over cooler coastal waters, causing air temperatures to drop and condensation to occur. This stratus layer may travel onshore and far inland, depending on the strength and direction of the prevailing breeze and local topography. Eventually, fog dissipates due to the warmer temperatures of the drier inland air mass. Foliage intercepts and absorbs some of this fog water while some droplets coalesce and are delivered to the ground as fog drip. This process is not unique to the north coast. Perhaps the earliest fog drip accounts date back to the legendary magic "rain tree" of the Canary Islands, an endemic laurel uprooted in 1610 that has inspired both legend and popular tradition (Gioda and others 1995). Scientists have measured fog drip at a variety of locations around the world,

1 This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

2 Hydrologist, USDA Forest Service, Pacific Southwest Research Station, Redwood Sciences Lab, 802 N. Main St., Fort Bragg, California, USA 95437. email: ekeppeler@fs.fed.us

including both coasts of South America, Mexico, Canada's maritime coast, northern Kenya, Australia, and northwestern United States (Azevedo and Morgan 1974; Dawson 1996, 1998; Harr 1982; Huntley and others 1997; Ingraham and Matthews 1988, 1995; Jacobs and others 1985; Kittredge 1948; Oberlander 1956; O'Connell 1975; Vogelman 1973; Yin 1994).

In the redwood region, published values dating back more than 50 years vary greatly *(table 1)*. More recently, researchers have used isotopic analyses of rain and fog water in comparison to groundwater and plant water to quantify the relative contributions of each source (Dawson 1996, 1998; Ingraham and Matthews 1995). According to Dawson (1998), fog inputs may account for as much as 10 to 45 percent of total water input to California's coastal redwood forest. Ingraham and Matthews (1995) found evidence of groundwater recharge due to fog drip at their most coastal site on the Point Reyes Peninsula. Other studies suggest fog is hydrologically significant, as well. Reduced summer streamflows following timber harvest in western Oregon have been attributed to the loss of fog drip (Harr 1982, Hicks and others 1991, Ingwerson 1985). On the basis of such studies, a Sonoma County judge recently banned logging on a 13-acre parcel because the removal of the tall redwoods might reduce groundwater recharge, thereby impacting domestic well water yields (Benfell 2003).

Table 1—*Fog-drip measurements in the redwood region.*

Region	Fog drip (mm/day)	Study duration (days)	Source
Coastal Oregon	2.3	126	Kittredge 1948
San Francisco Peninsula	1.2 to 37.4	40	Oberlander 1956
Orick, Humboldt Co.	0 to 2	70	Freeman 1971
Eel River Valley	0.1 to 9.2	46	Azevedo and Morgan 1974
Muir Woods	0 to 1	56	Jacobs and others 1985
Klamath River	5*	1095	Dawson 1998

* 3-yr average annual fog drip of 447 mm occurring on average 88.2 fog-days per year.

In view of the potential influence of fog drip on the water budget, a study was instituted at the Caspar Creek Experimental Watersheds on the Jackson Demonstration State Forest, Mendocino County, to assess the importance of the process in this watershed.

Methods

The Caspar Creek Experimental Watersheds are located approximately seven km from the Pacific Ocean and about 10 km south of Fort Bragg at 39º21'N 123º44'W *(fig. 1)*. Elevation ranges from 37 to 320 m. Climate is wet and mild, with an average annual precipitation of 1170 mm and mean monthly air temperatures ranging from 6.7 to 15.6 ºC. Stratus fog frequently inundates the watersheds during the summer when rainfall is scarce.

In 1998, a pilot study was initiated to determine if fog drip was measurable beneath the forest canopy of the Caspar Creek watersheds. Sampling sites were established where field observations suggested fog drip was likely at locations 4 to 9 km from the ocean and across a range of topographic conditions. Eleven sites were selected under second-growth trees and one under an old-growth redwood. Between four and seven four-inch diameter one-liter cylindrical collectors were mounted on stakes approximately 18 inches above the ground surface at each site. Five sites were located along exposed ridges, four were near-ridge, and the remaining three were at lower elevations near the channel *(fig. 1)*. No attempt was made to randomly sample the entire topography. During the months of June through September, fog-drip accumulations were measured volumetrically on those days when night or morning fog was observed along the coast. Measurements were delayed until the first workday following weekends and holidays. A recording rain gauge documented the onset and duration of fog drip at a single location.

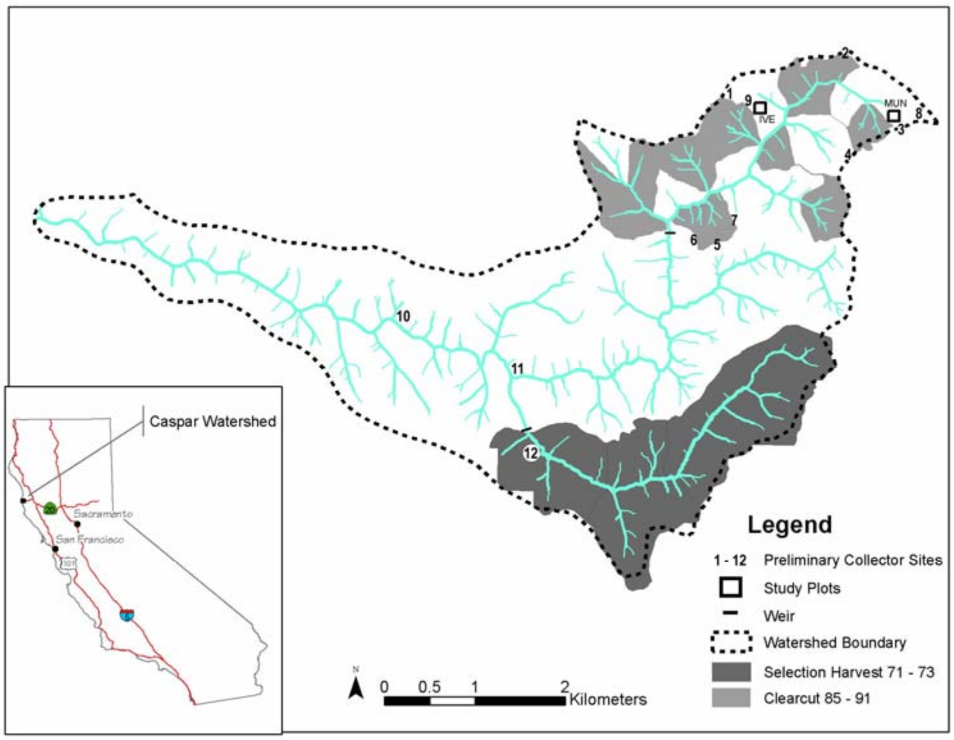


Figure 1—Caspar Creek watersheds, harvest areas, and fog-drip sampling sites.

In early August, 12 of these collectors were replaced with square platform collectors constructed of painted plywood. These were installed at a 25^{o} angle to facilitate drainage, producing an effective area of 1.35m^2. Yields from these platforms were collected in a covered bucket. Two additional platform collectors were installed in nearby clearcuts.

After measurable fog drip was documented and the efficiency of the platform collectors confirmed during 1998, a sampling plan for determining the within-stand variation in fog drip was designed and implemented at two one-ha plots as part of a larger study to assess the role of foliage interception within the North Fork of the Caspar Creek watershed (Reid and Lewis 2006). The plots were established within 120-yr-old second-growth stands dominated by coast redwood *(Sequoia sempervirens* (D. Don) Endl.) and Douglas-fir *(Pseudotsuga menziesii* (Mirb.) *Franco*). Sites were selected to span the range of stand class and density conditions within the North Fork while facilitating the instrumentation of a sample site in an adjacent clearcut. The IVE plot is located on a southeast-facing slope at an elevation of 220 m and supports a forest stand having a basal area of 240 m^2/ha. The other, MUN, is on a north-facing slope located 1.5 km to the east at an elevation of 270 m and has a basal area of 268 m^2/ha *(fig. 2)*.

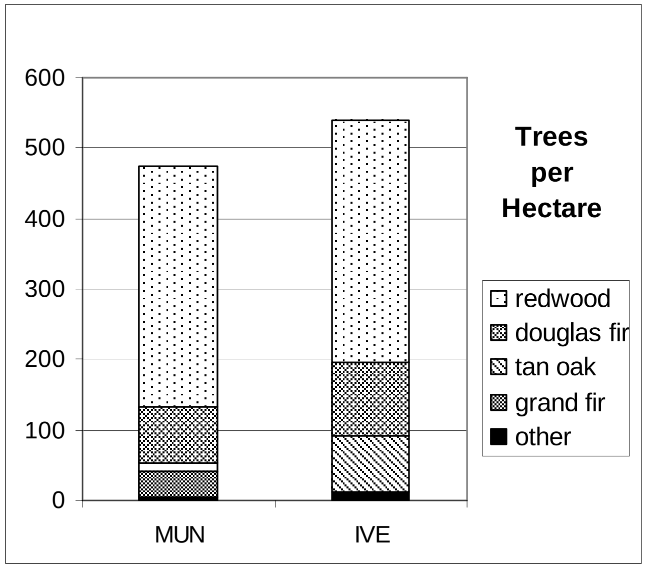


Figure 2—Stand composition in two North Fork sampling plots.

Six platform collectors were randomly located in each plot while two additional collectors (each paired with a standard eight-inch tipping bucket rain gauge) were situated in a nearby clearcut. Drainage from each platform collector was routed to a plastic barrel suspended from a load cell wired to a data logger recording at five-minute intervals. In addition, manual measurements were continued at five of the 1998 fog-drip sites and, on a less frequent schedule, at the platform collectors. Twenty-four-hour totals (noon to noon) were calculated for each collector and compared to rain gauge records. Equipment problems with two load cells eliminated one canopy collector at both IVE and MUN from this analysis. The Students *t*-test ($p < 0.05$) was used to detect a difference in fog-drip event yields between the open collectors and the mean of the five under-canopy collectors at each plot.

Results

Climatic observations indicate the presence of fog during one third of days during June through September of 1998 and 1999. After eliminating events for which rainfall was detected at the three standard rain gauge sites, 24 events were determined to have produced fog drip in 1998 and 29 in 1999. Typically, fog drip began after 1800 pst, peaking around 0600 pst and ending before noon *(fig. 3)*. In 1998, site-specific fog-drip yields varied from zero to 13 mm per single 24-hr event and were as high as 91 mm for the season. Similarly, in 1999 the maximum yields were 18 mm for a single event and 99 mm for the season. Differences among gauges suggest that fog drip is greatest at ridge locations most exposed to fog-laden westerly winds *(fig. 4)*.

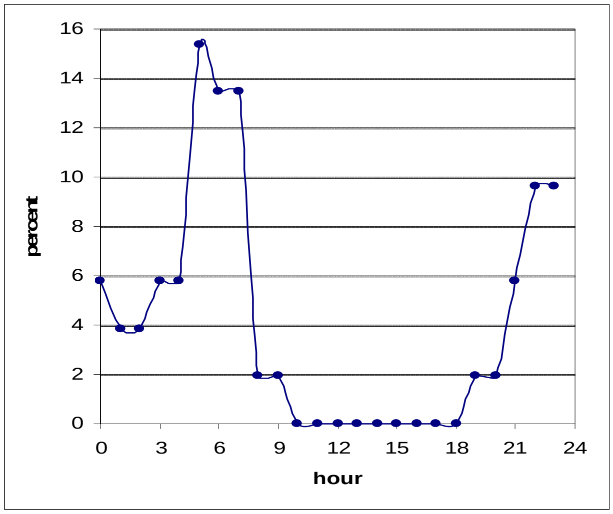


Figure 3—Temporal distribution of fog drip at one site during a three-week period in August 1998.

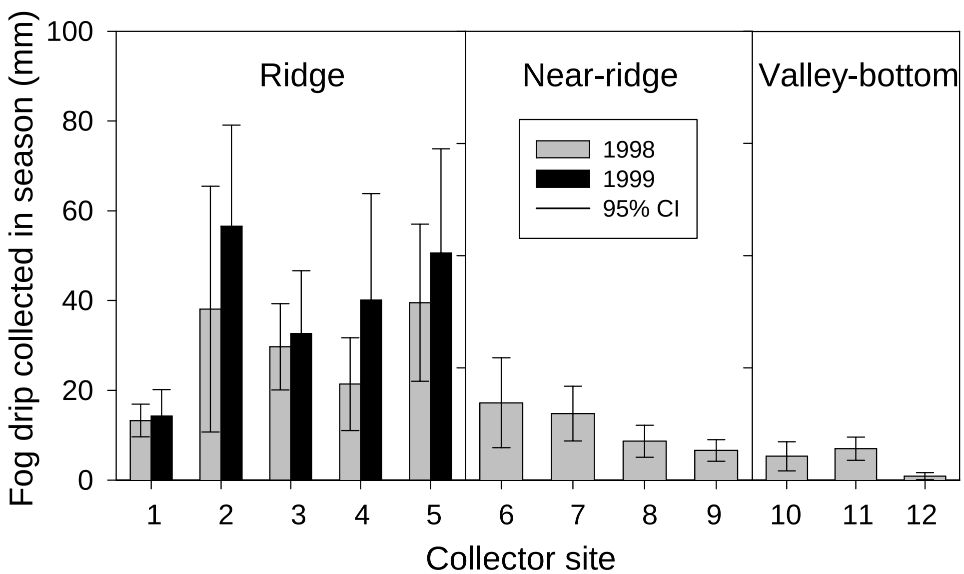


Figure 4—Seasonal fog-drip yields from 12 sites (61 sampling points). The mean value and 95 percent confidence interval for each site is shown.

Plot data from MUN and IVE produced far less dramatic results. Comparing the mean accumulation under the canopy to that in the nearby opening (*fig. 5*), fog-drip yields were greater under the canopy than in the opening at MUN, but not significantly different at IVE ($p < 0.05$; Students t-test). At MUN, fog drip enhanced moisture input by about 3 mm during the 29 fog-drip events of 1999. When the four days of summer rain are included in the analysis, no significant difference between water inputs in the forest and the open were detected ($p < 0.05$; Students *t*-test).

Discussion

Clearly, results from ridge-top fog collectors and casual observations of heavy drip trickling down the forest ridge roads suggest that fog drip can be appreciable in certain locations under certain conditions. Just as clearly, the absence of significant drip at other locations—including the two heavily-instrumented plots—indicates that other portions of the watershed are not strongly influenced by the process. These results confirm the occurrence and variability of fog drip in the Caspar Creek drainage and contrast to Dawson's data documenting much greater fog-drip accumulations directly under large redwoods along the much foggier Klamath coast. In addition to differences in site proximity to the ocean, collector type and stand versus tree sampling are important factors affecting these disparate results.

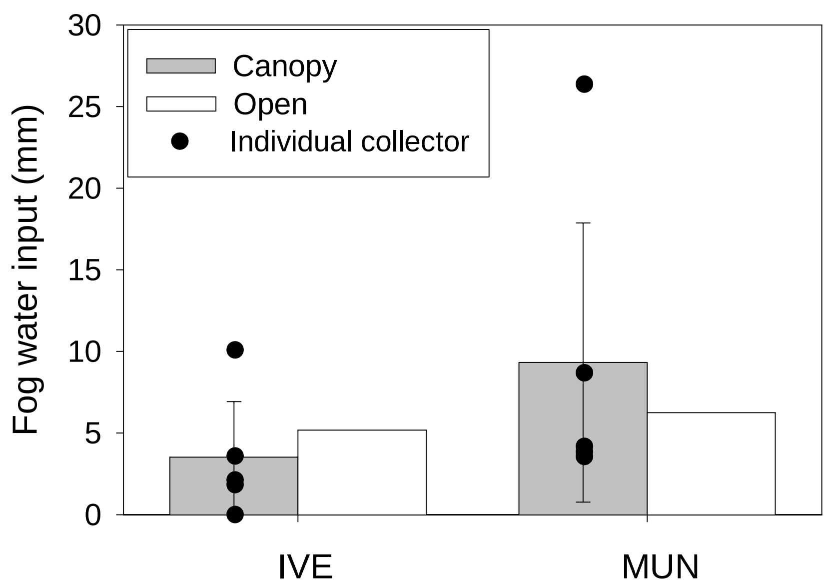


Figure 5—Fog water inputs (mm) for June to September 1999 beneath the canopy and in the open. The mean (shaded bar), individual collector values, and 95 percent confidence limits are shown.

The ridge-top gauges provide a non-statistical indication of probable maximum fog-drip rates for the experimental watershed. The mean values for the five ridge-top sites averaged 39 mm per year during the four month sampling period. This value is only three percent of the mean annual precipitation (1170 mm) at Caspar Creek, and so would represent only a minor influence on the annual water budget. However, only five percent of the annual rainfall (62 mm) occurs between May and September, so the average fog drip at ridge-top sites would augment dry-season precipitation by 63 percent. Water budget calculations described by Reid and Lewis (2006) suggest that transpiration over this period averages 162 mm under forested conditions, while runoff accounts for 27 mm. Even on ridge-top sites, where measured fog-drip rates are highest, a decrease in summer water input of 39 mm due to loss of fog drip would be over-balanced by a decrease in transpiration of 162 mm. At sites farther from the ridge, the relative influence of decreased transpiration would be even greater

At Caspar Creek, research focuses primarily on investigating the effects of timber harvest on streamflow processes. Data from long-term streamflow monitoring at Caspar Creek can be used to test the validity of these results. Two major investigations of logging effects on Caspar Creek streamflow have been completed in the past several decades. Both of these conclude that streamflow increases after logging. The first evaluated changes in streamflow following selection harvest of 65 percent of the South Fork watershed timber volume. Annual yield increases averaged 838 m^3/ha (1973 to 1982) and minimum flows increased 38 percent (1972 to 1978). The second evaluated the effects of clearcutting 50 percent of the North Fork basin. Again, annual yields increased 730 m^3/ha (1993 to 1997) and minimum flows by 148 percent (1990 to 1997). Additional instrumentation in the North Fork further indicated increased soil pipe flow, increased groundwater levels, and reduced soil moisture tensions within the cut units during the summer season (Keppeler 1998).

Streamflow increases are explained by reduced transpiration and interception in the logged areas. Reid and Lewis (2006) suggest clearcut logging in the North Fork watershed resulted in a 28.7 percent increase in effective precipitation at the forest floor (in other words, the forest would have intercepted 22.3 percent of the rainfall recorded at the rain gauge) and transpiration reductions estimated at 325 mm per year. Using these estimates, and assuming that all slash is burned and regrowth is not yet significant, one can calculate the expected average annual change in water availability to the North Fork watershed in the years immediately following logging:

$$\Delta W = L[0.223(PPT) + ET - FOG]$$

where

ΔW = additional available water (mm) for a year,

L = watershed area clearcut (percent),

PPT = annual precipitation,

ET = annual evapotranspiration from a mature second-growth stand,

FOG = annual fog water input to a mature second-growth stand.

Using the average annual rainfall from two North Fork gauges between 1993 and 1997 (1314 mm), a percentage cut of 37 percent, assuming no recovery of vegetation, and not including the loss of fog drip, the annual water balance of the North Fork was theoretically enhanced by as much as 229 mm (or 17 percent of average annual precipitation) during this five-year post-harvest period. In comparison, monitoring data indicate that increases in North Fork annual yield averaged 73 mm for five years after partial clearcutting. This equates to a *net* increase of 197 mm (73/.37) per year in streamflow from the clearcuts during this five-year post-harvest period—the net effect of reduced transpiration, interception, fog drip, and changes in soil moisture and groundwater storage. Differences between expected and observed changes are attributable to changes in soil moisture and groundwater storage, progressive hydrologic recovery from revegetation over the five-year period, annual variations in weather, and, especially near the ridges, some decrease in fog drip.

Reid and Lewis (in preparation) estimate post-drip canopy storage to be about one mm. A similar value is expected for potential storage of condensed fog before drip is significant. The effect of this wetting of the canopy is important in ameliorating moisture stress by direct foliar absorption and reduced transpiration losses. In 1999, fog drip occurred during 29 days, but solar radiation data collected at Caspar Creek indicate the presence of fog or cloud cover during more than a third of days in the study period. Although the canopy may not have been saturated to the point of drip by every fog event, enhanced humidity and reduced insolation and air temperatures moderated actual evapotranspiration rates.

Conclusions

Direct measurements of fog-drip inputs to the Caspar Creek watersheds indicate it is a highly variable, but minor component of the annual water balance at this site. Although this result contrasts with a few studies that have shown fog water to be a significant portion of the water balance, it does not contradict other results. The climatic and topographic conditions that influence the distribution and frequency of fog along the California coast vary considerably, so fog drip is expected to be more significant at some sites than others. Certainly, fog has a pervasive effect on redwood ecology, but the indirect effects of reduced evapotranspiration are probably larger than the direct effects of fog-drip inputs to soil moisture and groundwater at all but the most coastal locations. Only by a more comprehensive research effort might the spatial and temporal variations of fog water inputs in this region be more fully defined.

References

Azevedo, J.; Morgan, D.L. 1974. **Fog precipitation in coastal California forests**. Ecology 55(5): 1135-1141.

Benfell, C. 2003. **Fog drip plays role in halting logging.** The Press Democrat. November 30; Sec. B: 1 (col. 1).

Dawson, T.E. 1996. **The use of fog precipitation by plants in coastal redwood forests.** In: Proceedings of the conference on coast redwood forest ecology and management; 1996 June 18-20; Arcata, CA. Berkeley, CA: University of California Cooperative Extension; 90-93.

Dawson, T.E. 1998. **Fog in the California redwood forest: ecosystem inputs and use by plants**. Oecologia 117: 476-485.

Freeman, G.J. 1971. **Summer fog drip in the coastal redwood forest**. Arcata, CA: Humboldt State University; 101 p. M.S. thesis.

Gioda, A.; Hernandez, Z; Gonzalez, E.; Espejo, R. 1995. **Fountain trees in the Canary Islands: legend and reality.** Advances in Horticultural Science 9(3): 112-118.

Harr, R.D. 1982. **Fog drip in the Bull Run municipal watershed, Oregon**. Water Resources Bulletin 18(5): 785-788.

Hicks, B.J.; Beschta, R.L.; Harr, R.D. 1991. **Long-term changes in streamflow following logging in western Oregon and associated fisheries implications**. Water Resources Bulletin 27(2): 217-226.

Huntley, L.B.; Doley, D.; Yates, D.J.; Boonsaner, A. 1997. **Water balance of an Australian subtropical rainforest: the ecological and physiological significance of intercepted cloud and fog.** Australian Journal of Botany 45(2): 311-329.

Ingraham, N.L.; Matthews, R.A. 1988. **Fog drip as a source of groundwater recharge in northern Kenya.** American Geophysical Union 24(8): 1406-1410.

Ingraham, N.L.; Matthews, R.A. 1995. **The importance of fog-drip water to vegetation: Point Reyes Peninsula, California.** Journal of Hydrology 164(1995): 269-285.

Ingwersen, J.B. 1985. **Fog drip, water yield, and timber harvesting in the Bull Run Municipal Watershed, Oregon.** Water Resources Bulletin 21(3): 469-473.

Jacobs, D.F.; Cole, D.W.; McBride, J.R. 1985. **Fire history and perpetuation of natural coast redwood ecosystems.** Journal of Forestry 83(8): 494-497.

Keppeler, E.T. 1998. **The summer flow and water yield response to timber harvest.** In: Proceedings of the conference on coastal watershed: the Caspar Creek story; Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 35-44.

Kittredge, J. 1948. **Forest influences: the effects of woody vegetation on climate, water, and soil.** New York: Dover Publications; 115-119.

Oberlander, G.T. 1956. **Summer fog precipitation on the San Francisco peninsula.** Ecology 37: 851-852.

O'Connell, M.J. 1975. **The Wallaby Creek fog drip study.** No. MMBW-W-0004. Victoria, Australia: Melbourne Metropolitan Board of Works; 66 p.

Reid, L.M.; Lewis, J. 2006. **Rates and implications of rainfall interception in a coastal redwood forest.** In: Standiford, R.B.; Giusti, G.A.; Valachovic, Y.; Zielinkski, W.J., technical coordinators. Proceedings of the Redwood Region Forest Science Symposium. 2004 March 15-17; Rohnert Park, CA. Gen. Tech. Rep. PSW-GTR-194. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture [this volume].

Vogelman, H.W. 1973. **Fog precipitation in the cloud forests of eastern Mexico.** BioScience 23(2): 96-100.

Yin, Xiwei; Arp, P.A. 1994. **Fog contributions to the water budget of forested watersheds in the Canadian maritime provinces.** Atmosphere Ocean 32(3): 553-556.

Trends in Streamflow and Suspended Sediment After Logging, North Fork Caspar Creek[1]

Jack Lewis[2] and Elizabeth T. Keppeler[3]

Abstract

Streamflow and suspended sediment were intensively monitored at fourteen gaging stations before and after logging a second-growth redwood (*Sequoia sempervirens*) forest. About 50 percent of the watershed was harvested, primarily by clear-cutting with skyline-cable systems. New road construction and tractor skidding were restricted to gently-sloping ridge top locations, and watercourse protections were enforced.

Storm peak flows increased as much as 300 percent in clear-cut watersheds, but as antecedent wetness increased, percentage increases declined. In the first five to seven years after logging, the average two-year peak flow increased 27 percent in clear-cut watersheds and 15 percent in partially clear-cut watersheds. Changes in flows are attributable to reduced canopy interception and transpiration. Peak flows and flow volumes had recovered to near-pretreatment levels by about 10 years after logging, when renewed increases occurred from precommercial thinning.

Annual suspended sediment loads in the years following logging increased 123 to 238 percent in four of the five clear-cut watersheds. Loads did not change significantly at most downstream sites as sediment was deposited in the main stem. Channel erosion and changes in storage appear to be important mechanisms for explaining suspended sediment trends at Caspar Creek. Ten years after logging, storm-event sediment yields at one clear-cut tributary were near pretreatment levels, but were elevated again in year 12. At another, yields have remained well above pretreatment levels in the 12 years since harvest.

Key words: clear-cutting, logging effects, peak flow, streamflow, suspended sediment,

Introduction

In 1985, a multiple-basin watershed study was initiated in the North Fork of the Caspar Creek Experimental Watershed, in north coastal California. The study is a cooperative effort by the USDA Forest Service, Pacific Southwest Research Station and the California Department of Forestry and Fire Protection to investigate the impacts of harvesting second-growth redwoods under the Z'Berg-Nejedly Forest Practices Act of 1973. Although the logging included large clear cuts (maximum clear-cut size has since been reduced under California rules from 32 to 12 ha), erosional impacts were limited by careful road design and greatly restricted use of

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Mathematical Statistician, USDA Forest Service, Pacific Southwest Research Station, 1700 Bayview Drive, Arcata, CA 95521. email: jlewis01@fs.fed.us

[3] Hydrologist, USDA Forest Service, Pacific Southwest Research Station, 802 N. Main St., Fort Bragg, California, USA 95437. email: ekeppeler@fs.fed.us

tractors. The Proceedings of the Conference on Coastal Watersheds: The Caspar Creek Story (Ziemer, 1998) and Lewis and others (2001) reported results of the North Fork study through HY1996 (hydrologic year from August 1, 1995 to July 31, 1996). This paper extends the results through HY2003.

Methods

Study location

The Caspar Creek Experimental Watersheds are located about seven km from the Pacific Ocean and about 10 km south of the town of Fort Bragg in northwestern California. Elevation ranges from 37 to 320 m. Soils in the basin are well-drained clay loams derived from Franciscan sandstone and weathered coarse-grained shale of Cretaceous age.

The climate is typical of low-elevation coastal watersheds of the Pacific Northwest. Winters are mild and wet, characterized by periods of low-intensity (maximum 2.6 cm/hr) rainfall. Snow is rare. Average annual precipitation is 1170 mm. Typically, 95 percent falls during the months of October through April. Summers are moderately warm and dry with maximum temperatures moderated by frequent coastal fog. Mean annual runoff is 650 mm.

Like most of California's north coast, the watersheds were clear-cut and broadcast burned largely prior to 1900. By 1985, the North Fork watershed supported a 100-year-old second-growth forest composed of coast redwood (*Sequoia sempervirens* (D.Don) Endl.), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), western hemlock (*Tsuga heterophylla* (Raf.).Sarg.), and grand fir (*Abies grandis* (Dougl. ex D.Don) Lindl.).

Measurements

The North and South Forks of Caspar Creek (draining 473 ha and 424-ha, respectively) have been gaged continuously since 1962 using 120° V-notch weirs widening to concrete rectangular sections for high discharges. In 1985, three rated sections were constructed on the main stem upstream of the North Fork weir, and 10 Parshall flumes were installed on North Fork subwatersheds with drainage areas of 10 to 77 ha. Two of the original redwood Parshall flumes were replaced with fiberglass Montana flumes in HY1999 and 2001.

Since HY1986, stream discharge has been recorded at all gaging stations using electronic data loggers equipped with pressure transducers. From HY1986 to HY1995, suspended sediment was automatically sampled using real-time stage measurements to control a pumping sampler (Thomas 1989). Since HY1996, turbidity is recorded along with stage, and the sampling logic has been altered to use real-time turbidity (Lewis and Eads 2001).

Treatments

Ten areas were designated for harvest in compliance with the California Forest Practice Rules in effect in the late 1980s *(fig. 1)*. Two of these areas (13 percent of the North Fork watershed) were harvested in 1985 and 1986 with the intent of excluding them from the study. However, this harvest affects all subsequent analyses of North Fork weir data. After a calibration period between 1985 and 1989, clear-cut logging began elsewhere in the North Fork in May 1989 and was completed in January 1992. These clear-cuts occupied 30 to 99 percent of treated watersheds and totaled 162 ha. Between 1985 and 1992, 46 percent of the North Fork watershed was clear-cut, 1.5 percent was thinned, and two percent was cleared for road right-of-way. Of the fourteen gaged watersheds in the North Fork, five were clear-cut, three were left as unlogged controls, and six included mixtures of clear-cut and unlogged areas. In HY1996, stream gaging was discontinued at all but two of the clear-cut watersheds, two of the controls, and three of the partially clear-cut watersheds.

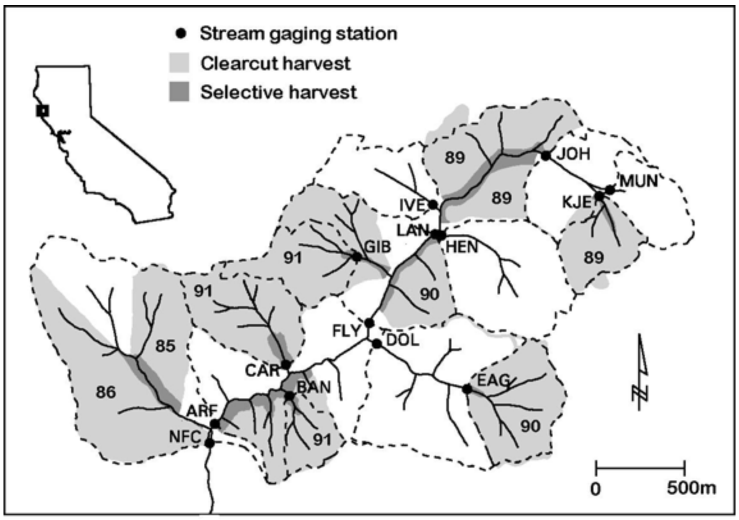


Figure 1—North Fork Caspar Creek gaging stations and harvest units.

Harvest was conducted under stream-buffer rules that mandated equipment exclusion and 50 percent canopy retention within 15 to 46 m of watercourses providing aquatic habitat or having fish present. Most of the yarding (81 percent of the clear-cut area) was accomplished using skyline-cable systems. Yarders were situated on upslope landings constructed well away from the stream network. New road construction and tractor skidding was restricted to ridgetop locations with slopes generally less than 20 percent. Four harvest blocks, 92 ha total, were broadcast burned and later treated with herbicide. Pre-commercial thinning in 1995, 1998, and 2001 eliminated much of the dense revegetation and reduced basal area in treated units by about 75 percent.

Results

Storm peaks

Lewis and others (2001) analyzed peak flow response to clear-cutting in the North Fork using 526 observations from HY1986 to HY1996, representing 59 storms on 10 treated watersheds. After logging, eight of the 10 tributary watersheds experienced increased storm peaks ($p < .005$) relative to those predicted on the basis of the controls for an uncut condition. In clear-cut units, individual storm peaks increased as much as 300 percent, but most increases were less than 100 percent. The largest increases occurred during early season storms. As basin wetness increased, percentage peak flow increases declined *(fig. 4)*. In the larger, partially clear-cut

North Fork watersheds, smaller peak flow increases were observed. Under the wettest antecedent moisture conditions of the study, increases over the first five to seven years after logging averaged 23 percent in clear-cut watersheds and 3 percent in partially clear-cut watersheds. The average increase in storm peak with a two-year return period was 27 percent in the clear-cut watersheds and 15 percent in the partially clear-cut watersheds (Ziemer 1998) for this five to seven year period. While variability is great, ongoing measurements clearly show a recovery to near pre-treatment flow conditions 10 years post-harvest and the suggestion of a renewed response to the pre-commercial thinning *(fig. 2)*.

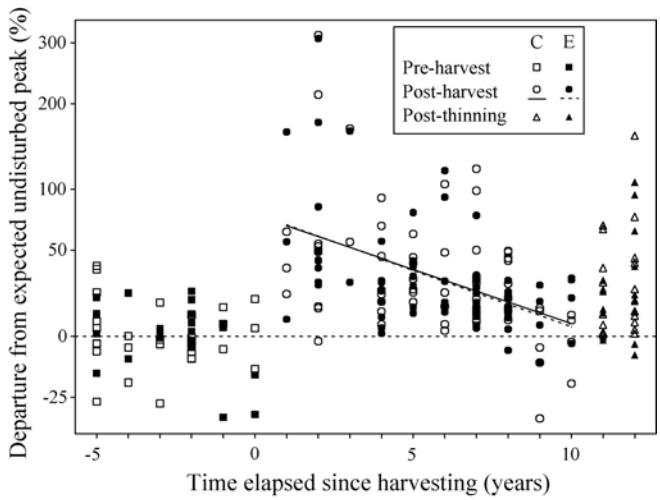


Figure 2—Peak flows observed in North Fork clear-cut units C and E from HY1986 through HY2003. Expected undisturbed peak is based on log-log regressions of pre-harvest peak flows at CAR and EAG on the mean of the corresponding peak flows at control watersheds HEN and IVE.

Wetter soils resulting from reduced transpiration in logged units explain some of the observed increases in streamflow. In addition, because of reduced canopy interception, 28 percent more precipitation is delivered to the forest floor after clear-cut logging in these second-growth redwood stands (Reid and Lewis 2006). Under forested conditions, canopy interception is significant even during the wettest mid-season storms. Loss of interception is therefore expected to maintain wetter soil conditions in logged terrain throughout the rainy season.

Lewis and others (2001) fit an empirical model expressing the HY1986-1996 North Fork peak flows as a function of peaks in the control watersheds, antecedent wetness, proportion of area logged, and time since logging. In this follow-up, a slightly simplified version of that model was refit, using generalized non-linear least squares, to all peak flows before pre-commercial thinning (HY1986-2001).

$$\ln(y_{ij}) = \beta_{0i} + \beta_{1i}\ln(y_{Cj}) + \left[\left(1-\beta_2(t_{ij}-1)\right)c_{ij} + \beta_3 c'_{ij}\right]\left[\beta_4 + \beta_5\ln(y_{Cj}) + \beta_6\ln(w_j)\right] + \varepsilon_{ij} \quad (1)$$

where

y_{ij} = unit area peak flow at treated watershed *i*, storm *j*,

y_{Cj} = mean of unit area peak flows at control watersheds HEN and IVE in storm *j*,

t_{ij} = area-weighted mean cutting age (number of summers passed) in watershed *i* for areas logged in water years preceding that of storm *j*,

c_{ij} = proportion of watershed *i* logged in water years prior to that of storm *j*,

c'_{ij} = proportion of watershed *i* logged in the fall prior to storm *j* (in the same water year)

w_j = wetness index at start of storm *j*, computed from daily streamflow (30-day half-life) at South Fork weir

ε_{ij} = independent normally distributed errors with variance inversely proportional to a power function of watershed area

β_{0i} and β_{1i} are "location" parameters to be estimated for each watershed *i*, and

β_2, β_3, β_4, β_5, and β_6 are parameters describing the effects of the explanatory variables

The first two terms in the model predict the peak flow in the absence of disturbance. The first bracketed term represents vegetation removal and regrowth, and the terms in the second set of brackets are the main effect of vegetation change (β_4) and interactions of vegetation change with storm size and antecedent wetness. The coefficient estimates and their standard errors are given in *table 1*. This model fits the data well (r^2 = 0.95) and residuals are normally distributed with standard error equivalent to 25 percent of the predicted peak.

Table 1—*Parameter estimates for storm peaks and flow volume models.*

		----------Storm peak----------			-------Storm flow volume-------		
Parameter	**Effect**	**Estimate**	**Std error**	**p**	**Estimate**	**Std error**	**p**
β_2	Recovery	0.101	0.0063	<0.0001	0.110	0.0059	<0.0001
β_3	Fall logging	0.447	0.0965	<0.0001	0.876	0.0926	<0.0001
β_4	Vegetation reduction	1.290	0.2596	<0.0001	2.824	0.2287	<0.0001
β_5	Storm size interaction	-0.110	0.0363	0.0025	-0.140	0.0392	0.0004
β_6	Wetness interaction	-0.278	0.0177	<0.0001	-0.298	0.0178	<0.0001

The fitted value of 0.101 for the coefficient β_2 implies recovery of peak flows to pretreatment conditions after 11 growing seasons, in concordance with *figure 2*. A 95 percent confidence interval for β_2 implies recovery in 10 to 12 years. The fitted value of 0.447 for β_3 suggests that the effect on peak flows during the first winter was reduced by about 55 percent because much of the harvest occurred late in the growing season, after substantial transpiration had occurred. The storm size interaction indicates that the proportional increase in peak flows was smaller for larger events, and the wetness interaction indicates that increases in peak flows are greatest during low antecedent wetness conditions.

Model (1) was used to predict peak flows without accounting for the change in cover following thinning. *Figure 3* shows the departures from peak flows predicted by this model for the two clear-cut watersheds, CAR and EAG, that are still being monitored. Departures, e_{ij}, are converted to percentage of predicted peak through the transformation 100exp(e_{ij}). The recovery trend depicted in *figure 2* is not visible in *figure 3* because the model accounts for the recovery. However, the mean post-thinning departure from the predicted peak is 26 percent (the 95 percent confidence interval is 16 to 37 percent). These departures are greatest when antecedent wetness is greatest *(fig. 4)*, suggesting that mechanisms similar to those responsible for increasing peaks after clear-cutting are involved in changing peaks after thinning.

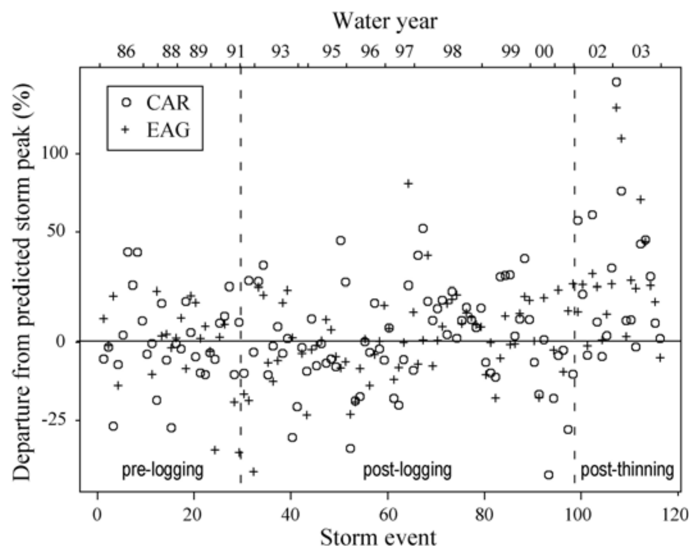


Figure 3—Departures from model (1) predictions of storm peak.

The thinning in watersheds CAR and EAG removed 68 and 84 percent of the crown volumes, respectively. The peaks model permits a test of whether these treatments were equivalent to clear-cutting the same percentage of the watersheds. For the calculations of *figures 3* and *4*, the variable t_{ij} was coded as 10 and 11 years, respectively, for CAR and EAG in HY2002, the winter following thinning. However, if we treat the disturbance as if 68 and 84 percent of the areas were clear-cut in the beginning of HY2002, the area-weighted mean cutting ages t_{ij} should be coded 3.2 for CAR and 1.8 for EAG in HY2002; and the ages in HY2003 should be 4.2 and 2.8 years. Based on this recoding of t_{ij}, the model predicts an average increase of 52 percent in peak flows, suggesting that thinning had half the impact on peak flows of an equivalent harvest by clear-cutting. Such a result is expected if evaporation and transpiration rates are elevated in a thinned stand because of lower aerodynamic resistance to the transport of water vapor as suggested by Calder (1990).

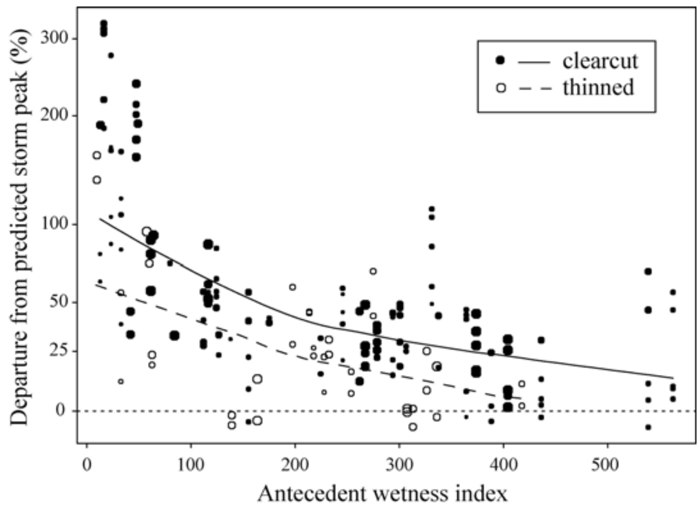


Figure 4—Relation to antecedent basin wetness of (a) clear-cut departures from pretreatment regressions (BAN, CAR, EAG, GIB, and KJE through HY1996), and (b) post-thinning departures from model (1) (CAR and EAG, HY2002-2003). Symbol sizes denote relative storm sizes.

Storm flow volumes

Storm flow volumes were analyzed using the same methods as for peak flows. The results through HY1996, reported by Lewis and others (2001) were similar to peak flow results. In clear-cut units, storm flows increased as much as 400 percent, but most increases were less than 100 percent. The largest increases occurred during early-season storms. As basin wetness increased, percentage increases declined. Under the wettest antecedent moisture conditions of the study, increases averaged 27 percent in clear-cut watersheds and 16 percent in partially clear-cut watersheds over the five to seven year period following harvest. Annual storm runoff volume (sum of storms) increased an average of 58 percent in clear-cut watersheds and 23 percent in partly clear-cut watersheds (the mean percentage harvested was 38 percent). As with peak flows, ongoing measurements show a return to pre-treatment flow volumes approximately 10 years post-harvest, followed by a response to the pre-commercial thinning *(fig. 5)*.

Model (1) also the fits the flow data well (r^2 = 0.94) with normally-distributed residuals and standard error equivalent to 21 percent of the predicted flow volume. The estimated recovery coefficient (*table 1*) suggests return of storm flows to pretreatment condition 10 years after logging, and is consistent with *figure 5*.

The flow model enabled quantification of the impact of pre-commercial thinning at CAR and EAG for 18 events in the two post-thinning years. The mean post-thinning departure from predicted flow volume was 26 percent (the 95 percent confidence interval is 15 to 38 percent) and the total storm flow volume was 19 percent greater than predicted by the model.

When the variable t_{ij} was recoded (as described above for peaks) to represent thinning as an equivalent harvest by clear-cut, the model predicts a mean increase of 53 percent and total increase of 44 percent in storm flow. Compared to an equivalent clear-cut, the mean effect of thinning on storm flows was about half (26/53) and the total effect on storm flows was 43 percent (19/44).

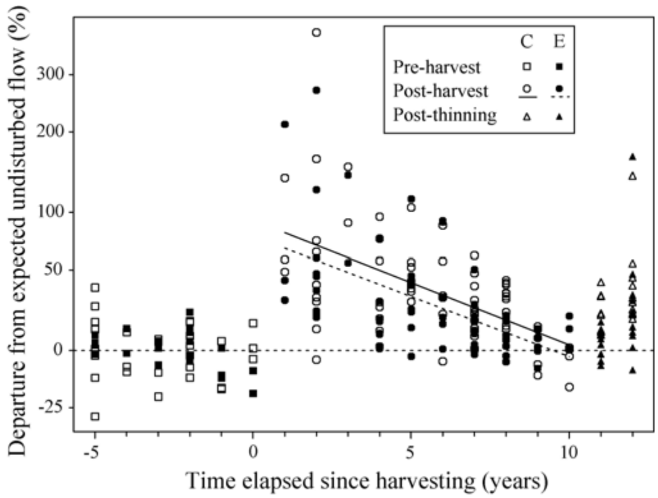


Figure 5—Storm flow volumes observed in North Fork clear-cut units C and E from HY1986 through HY2003. Expected undisturbed flow is based on log-log regressions of pre-harvest flows at CAR and EAG on the mean of the corresponding storm flows at control watersheds HEN and IVE.

Sediment Loads

Suspended sediment loads, summed over post-logging years through HY1996 increased 89 percent at the North Fork weir, primarily due to one landslide that occurred in the 1986 harvest area in 1995. Annual (sum of storms) suspended sediment loads in the years following logging decreased by 40 percent in one clear-cut watershed (KJE) and increased 123 to 238 percent in the other four clear-cut watersheds. Loads did not change significantly at most downstream sites, but at DOL increased by 269 percent. The median estimate of change in annual sediment load was +132 kg $ha^{-1}yr^{-1}$ for five clear-cut watersheds and –19 kg $ha^{-1}yr^{-1}$ for five partially clear-cut watersheds. Increases in sediment loads were greatest during those events with increased storm flows. In clear-cut watersheds where sediment loads increased, the correlations between departures from pretreatment sediment load and storm flow models were 0.66 (BAN to HY1995), 0.70 (CAR to HY2003), 0.62 (EAG to HY2003), and 0.86 (GIB to HY1995). Sediment increases at EAG have been greater than at CAR due to near-channel tunnel collapses. Storm event loads in EAG remained elevated a decade after harvest, while, at CAR, yields were close to the pretreatment level in year 10 (*fig. 6*). Suspended sediment levels from both subwatersheds, especially EAG, increased sharply in year 12 (HY2003), the first above-average runoff year since HY1999. Although sediment levels did not increase the first year after thinning, they certainly may have been influenced by the larger enhanced flows of HY2003. Prolonged impacts from logging in the South Fork (Keppeler and others, 2003) suggest that the episodic nature of sediment releases requires patience regarding conclusions about recovery.

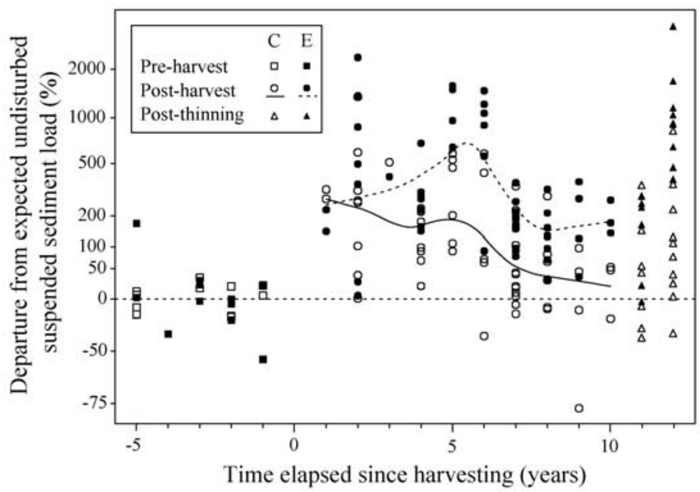


Figure 6—Sediment loads observed in North Fork clear-cut units C and E from hydrologic year 1986 through 2003.

Discussion and Conclusions

Although the variability is great, the impacts of clear-cut logging and forest regrowth on peaks and flows in the North Fork of Caspar Creek are fairly clear and quantifiable. Earlier analyses of selective logging in the South Fork of Caspar Creek (Ziemer 1981) had failed to show significant changes in peak flows, except in the smallest events at the beginning of the rainy season. Those results are not necessarily at odds with the North Fork study and may be attributable in part to differences in silvicultural methods (selective versus clear-cut logging). If thinning is a valid analog for selection cutting, our analysis suggests that the South Fork response should have been smaller than that in the North Fork. In addition, the North Fork analyses were more sensitive because multiple unlogged subwatersheds of the North Fork were available for use as controls. Low variability in the pretreatment relationship is critical to an effective watershed experiment, and the responses in North Fork watersheds slated for treatment were more closely related to North Fork subwatershed responses than to the South Fork response. In fact, the mean of two unlogged subwatersheds provided a better control than any individual subwatershed.

An empirical statistical model describes impacts on flow peaks and volumes in terms of antecedent basin wetness, proportion of area cut, time since logging, and event size. The effect of vegetation removal is greatest when the wetness index is low and diminishes as basin wetness increases. However, no conditions were observed under which the impacts were reduced to zero. The result is not unexpected given that effective rainfall is increased substantially by the loss of canopy interception throughout the rainy season (Reid and Lewis 2006).

A somewhat surprising result is that flow peaks and volumes 10 years after logging were similar to those in 100-year-old redwood forest. Further research will be necessary to understand this result, but it suggests that leaf area recovers very rapidly after harvest, and/or that evapotranspiration rates per unit leaf surface are much greater in younger forests. In fact, evidence suggests both may be true. Crown closure and maximum leaf area in one redwood plantation was attained within 15 years.[4] In riparian Douglas-fir forests of western Oregon, Moore and others (2004) found that a 40-year-old, rapidly growing stand used 3.3 times more water during the growing season than an old-growth stand.

Pre-commercial thinning resulted in smaller flow changes than would have been expected from equivalent clear-cuts. This may be partly related to the influence of canopy structure on evaporation rates. Calder (1990) reported that interception rates in mature spruce forest were almost unchanged after thinning one-third of the stand. He speculated that increased ventilation to lower levels of the canopy could increase evaporation rates. Reduced competition for soil water could also permit increased transpiration by vegetation that remains after thinning.

Variability in suspended sediment yield is much greater than variability in flow. Results are less consistent among clear-cut subwatersheds and much less predictable in downstream watersheds. One North Fork subwatershed that was clear-cut (KJE) experienced a decrease in sediment loads. The others experienced substantial increases. Of the two that are still being measured, neither has returned to pretreatment levels, and one (EAG) is yielding significantly more sediment than the other (CAR). One downstream site (DOL) had larger than expected sediment yields, apparently because of increased channel erosion, while those on the main stem have not experienced elevated sediment yields, apparently because of increased sediment storage. Unusual windstorms in combination with increased wind exposure in stream buffer zones resulted in blowdown that created many new sediment storage sites in the formerly wood-deprived main stem.

The sediment results are less directly extensible to other watersheds than the flow results, because they depend on events and conditions unlikely to be repeated in every coastal watershed. This is especially true as one moves downstream from first and second order streams to locations where channel complexity is greater. The results of the Caspar Creek sediment studies are probably not useful for making quantitative predictions, but they have helped us to understand many controlling factors and links among erosion, sediment delivery, and sediment transport. It has become clear that sediment impacts from regulated logging in the North Fork have been less severe than those from the tractor logging that took place in the South Fork (Keppeler and others 2003), and the research suggests opportunities for further reducing impacts. For example, limiting the rate of harvest in a given watershed would clearly limit increases in peak flows and flow volumes. Sediment yield increases in the North Fork were related to flow increases, so limiting harvest rates should also be effective in limiting sediment impacts. To further limit sediment yields in the North Fork would have required extending streamside protection zones farther upstream, but the incremental benefit of doing so is difficult to quantify, and it probably would not have greatly reduced sediment yields in DOL where much of the channel and bank erosion occurred downstream from the logged watershed (EAG).

Today much of the managed timber-producing area of north coastal California has been logged at least twice and may have experienced heavy impacts from tractor logging and road construction. The condition of the South Fork of Caspar Creek is probably more typical of areas being logged today than was the North Fork. It is becoming crucial for landowners, regulatory agencies, and the public to understand

[4] O'Hara, K.L.; Stancioiu, P.T.; Spencer, M.A. Manuscript in review. Understory stump sprout development under variable canopy density and leaf area in coast redwood. Canadian Journal of Forest Research.

the interactions between proposed future activities and prior disturbances. A third phase of Caspar Creek research is being initiated in the South Fork to examine the effects of re-entry on runoff and sediment production from previously tractor-logged redwood forests. Much remains to be learned about restoring impacted ecosystems and mitigating impacts from future harvests.

References

Calder, I. 1990. **Evaporation in the Uplands**. Chichester, UK: Wiley and Sons; 149 p.

Keppeler, E.T.; Lewis, J.; Lisle, T.E. 2003. **Effects of forest management on streamflow, sediment yield, and erosion, Caspar Creek experimental watersheds**. In: Renard, K.G.; McElroy, S.A.; Gburek, W.J.; Canfield, H.E.; Scott, R.L., eds. First interagency conference on research in the watersheds; 2003 Oct 27-30; Benson, AZ. Agricultural Research Service, U.S. Department of Agriculture; 77-82.

Lewis, J.; Eads, R. 2001. **Turbidity threshold sampling for suspended sediment load estimation**. In: Proceedings of the 7th Federal interagency sedimentation conference, volume 2; 2001 Mar 25-29; Reno, NV; Section III (Data Collection); 110-117.

Lewis, J.; Mori, S.R.; Keppeler, E.T.; Ziemer, R.R. 2001. **Impacts of logging on storm peak flows, flow volumes and suspended sediment loads in Caspar Creek, California**. In: Wigmosta, M.S.; Burges, S.J., eds. Land use and watersheds: human influence on hydrology and geomorphology in urban and forest areas. Water Science and Application, volume 2; Washington, DC: American Geophysical Union; 85-125.

Moore, G.W.; Bond, B.J.; Jones, J.A.; Phillips, N.; Meinzer, F.C. 2004. **Structural and compositional controls on transpiration in 40- and 450-year-old riparian forests in western Oregon, USA**. Tree Physiology 24: 481-491.

Reid, L.M.; Lewis, J. 2006. **Rates and implications of rainfall interception in a coastal redwood forest**. In: Standiford, R.B.; Giusti, G.A.; Valachovic, Y.; Zielinkski, W.J., technical coordinators. Proceedings of the Redwood Region Forest Science Symposium. 2004 March 15-17; Rohnert Park, CA. Gen. Tech. Rep. PSW-GTR-194. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture [this volume].

Thomas, R.B. 1989. **Piecewise SALT sampling for estimating suspended sediment yields**. Gen. Tech. Rep. PSW-GTR-83. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 11 p.

Ziemer, R.R. 1981. **Stormflow response to roadbuilding and partial cutting in small streams of northern California**. Water Resources Research 17(4): 907-917.

Ziemer, R.R., technical coordinator. 1998. **Proceedings of the conference on coastal watersheds: the Caspar Creek story;** 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 149 p.

Rates and Implications of Rainfall Interception in a Coastal Redwood Forest[1]

Leslie M. Reid[2] and Jack Lewis[2]

Abstract

Throughfall was measured for a year at five-min intervals in 11 collectors randomly located on two plots in a second-growth redwood forest at the Caspar Creek Experimental Watersheds. Monitoring at one plot continued two more years, during which stemflow from 24 trees was also measured. Comparison of throughfall and stemflow to rainfall measured in adjacent clearings indicates throughfall and stemflow accounted for 75.1 and 2.5 percent, respectively, of annual rainfall, while 22.4 percent was intercepted and evaporated by the forest canopy. Average interception loss remains above 20 percent even for the largest storms monitored. Models that predict pre-logging peakflows from below-canopy rainfall suggest that altered interception and transpiration could account for the 54 to 70 percent average increases in peakflow observed in five gauged watersheds for two years after clearcutting. Results such as these can be used to estimate the influence of interception loss on landslide frequency at sites for which relationships between landslide frequency and storm rainfall have been defined.

Key words: evapotranspiration, interception, landslides, peakflow, water budget

Introduction

The extent to which logging might influence flood frequencies and erosion rates has long been a focus of concern, but analytical methods for reliably predicting potential impacts are not well-developed because the influence of forests on hydrologic and erosional processes is not fully understood. For example, logging has been assumed to affect peakflows in rain-dominated areas primarily through changes in transpiration, so only the smaller, early-season peaks—those for which post-logging changes in antecedent soil moisture are likely to be largest—are expected to change after logging. However, recent studies in northwest California indicate that even mid-winter peakflows with two-year recurrence intervals increase after logging (Lewis and others 2001, Ziemer 1998).

Recent landslide surveys also provide unexpected results: data provided by Pacific Watershed Associates (1998) show landsliding rates in some recently clearcut redwood lands to be almost an order of magnitude higher than in adjacent second-growth forests. Logging had not been expected to increase landsliding markedly in these areas because trees were thought to influence stability primarily through root cohesion, and root cohesion was thought to remain substantially intact after logging of second-growth redwoods because most stumps remain alive and resprout. If altered cohesion indeed were not a major influence, some other mechanism for

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California

[2] USDA Forest Service Pacific Southwest Research Station, Redwood Sciences Laboratory, 1700 Bayview Drive, Arcata, CA 95521. email: lreid@fs.fed.us and jlewis@fs.fed.us, respectively.

destabilization must accompany logging.

A mechanism that might explain both observations is increased effective rainfall produced by a decrease in the amount of rain intercepted by and evaporated from foliage after logging. Such changes are appreciable elsewhere: annual interception of 22 to 49 percent was measured in New Zealand forests (Fahey 1964, Pearce and Rowe 1979), and losses of 20 to 45 percent were found in mature coastal forests of the Pacific Northwest (McMinn 1960, Spittlehouse 1998). We implemented a study at the Caspar Creek Experimental Watersheds to measure interception in a 120-year-old redwood forest, and to determine whether the process might be capable of influencing peakflow discharges and landsliding. The study is described in more detail by Lewis (2003), Reid and Lewis (in review), and Steinbuck (2002).

Methods

Six 1.2 m × 1.2 m rainfall collectors were distributed randomly across each of two forested 1-ha plots in North Fork Caspar Creek watershed, Mendocino County, California. The IVE site is in Iverson catchment on a southeast-facing slope at 220 m elevation. The stand at IVE has coastal redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*) as the primary canopy components, a patchy understory dominated by tanoak (*Lithocarpus densiflora*), and a basal area of 97 m^2/ha. The second site (MUN) is located at an elevation of 270 m on a north-facing slope in the Munn catchment, 1.5 km from IVE; here the canopy of redwood and Douglas-fir is more uniform, basal area is 108 m^2/ha, and the understory is sparse.

Each throughfall collector channeled water into a 150-l barrel suspended from a load cell, and data loggers recorded readings from load cells at five-minute intervals. At each plot, an identical control collector and a standard eight-inch tipping-bucket rain gauge were located in an adjacent clearcut. The analysis is based on data recorded between December 1, 1998 and November 8, 1999 at MUN and between December 5, 1998 and May 27, 2001 at IVE. One load cell failed after the 24th event at IVE, but that gauge had consistently reflected the plot average so its loss did not influence study results. Other missing data were filled in using relationships defined between readings at the various gauges.

At the long-term site, 24 trees (12 redwood, eight Douglas-fir, and four tanoak) were selected randomly and equipped with collars to divert stemflow into containers (Steinbuck 2002). Water depths were measured with a dipstick at two-day to four-week intervals between December 2, 1999 and April 20, 2001. Flows from six of the trees were routed through tipping-bucket gauges to record stemflow timing.

Results

Annual rainfall totals from standard rain gauges were four to seven percent higher than those from control collectors, so throughfall was calculated by comparison to control collectors instead of rain gauges, on the assumption that trap efficiency would be similar in forest and clearing. Sub-canopy collectors indicated that more than 98 percent of the post-event drip occurs within three hr of rainfall's end, but stemflow can continue to drain from the largest trees for 48 hr after major storms. Events were thus defined to be bounded by dry periods of at least 48 hr at the tipping bucket gauges.

Collectors operating during the first year showed total throughfall of 79 ± 6 percent (95 percent CI), with IVE having a slightly lower average (78 percent) than MUN (80 percent). The IVE plot, with a denser sub-canopy and less uniform canopy, showed higher variability among collectors (standard deviation of 0.16 compared to 0.04 at MUN). Annual throughfall ranged from 78 to 72 percent over three years of measurement at IVE and averaged 75.1 percent.

Lewis (2003) found that stemflow at IVE accounts for 2.5 percent of annual rainfall, leaving 22.3 percent to be trapped by foliage interception. For each species, event-based stemflow data show linear relations between event size and stemflow volumes, and this information was combined with Lewis' (2003) relations between tree diameter and annual stemflow to estimate stemflow for the plot for each event.

Stemflow was added to event-based throughfall and the result subtracted from rainfall to estimate interception loss for each event. Average interception rates are highest and most variable for the smallest storms and decrease with event size *(fig. 1)*, approaching an asymptote of about 21 percent for events larger than 70 mm.

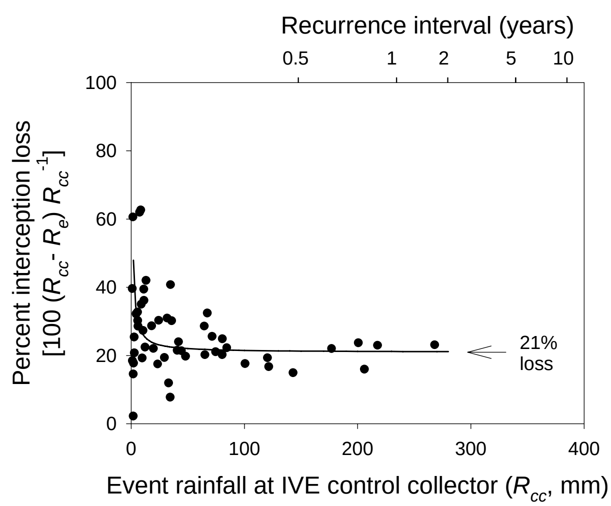


Figure 1–Variation of interception loss with storm rainfall.

Effective rainfall (R_e, mm, calculated as throughfall + stemflow) is strongly correlated with rainfall measured in the control collector (R_{cc}, mm) for events defined by 48-hr dry periods at IVE:

for $R_{cc} \geq 0.7$ mm $\qquad R_e = -0.548 + 0.786\, R_{cc} \qquad r^2 = 0.99 \qquad$ (1a)

for $R_{cc} < 0.7$ mm $\qquad R_e = 0 \qquad$ (1b)

Discussion

Mechanisms of interception loss

The difference between cumulative rainfall and cumulative throughfall was plotted at five-min intervals for each storm to identify the temporal distribution of interception loss during the storm (for example, *fig. 2)*. Results show that loss rates during rainfall are appreciable; summed through the season, in-storm loss accounts for about half the overall loss, and the remainder evaporates after rainfall ceases.

Although clouds are saturated during rainstorms, the air below usually is not, so some water can evaporate (for example, Gash 1979). Because the surface area of foliage in a forest canopy is large, even low evaporation rates (in terms of loss per unit area of water surface) can evaporate large volumes of water (in terms of loss per unit area of ground surface). Similar stands nearby have a ratio of one-sided leaf area to ground area of about 14 (Kevin O'Hara, UC Berkeley, pers. comm.), implying that the actual evaporation rate per unit area of ground surface could be 28 to 44 times the rate per unit area of leaf surface, depending on the cross-sectional shape of the needles, if all surfaces are wet.

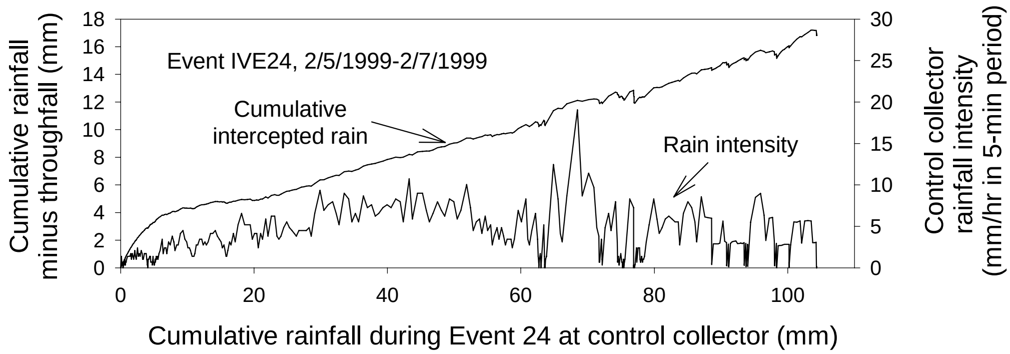


Figure 2–Cumulative intercepted rain and rainfall intensity versus cumulative rainfall at the control collector for Event 24 at IVE. The slope of the intercepted-rain curve defines the in-rain interception rate.

As an example, standard open-pan evaporation data from an area with similar climate 90 km north of Caspar Creek suggest that winter in-rain evaporation might average about 0.006 mm^3/hr (95 percent CI: 0.0005 to 0.012 mm^3/hr) per mm^2 of wetted surface, leading to an estimated loss of 0.17 to 0.26 mm^3/hr per mm^2 of ground surface for the Caspar canopy. This rate would produce about 7 to 10 mm of interception loss during the 40-hour event shown in *figure 2*. Canopy evaporation rates may be higher than open-pan rates because of increased airflow and turbulence. Some water may also be absorbed by leaves and bark for later evaporation. However, the relatively uniform loss rate during storms, as well as results of isotopic studies (Dawson 1996), suggest that absorption by leaves is less important than evaporation.

Interception and the seasonal water balance

We carried out a monthly water balance calculation, as described by Dunne and Leopold (1978 p. 238), for the North Fork Caspar Creek watershed. Rainfall and runoff have been measured in the watershed since 1963, soil moisture storage capacity can be estimated from information provided by Wosika (1981), and the present study provides estimates of interception loss. The forest-floor litter layer is also expected to intercept and evaporate rainfall, but this component was not measured during the study. Interception losses of two to five percent have been measured for litter in deciduous forests of eastern North America (Helvey and Patric 1965), and we assume a loss of three percent for these calculations.

The Thornthwaite equation for evapotranspiration implicitly considers both transpiration and interception but incorporates only solar input and temperature as driving variables, so the equation characterizes transpiration more effectively than interception. We assumed that the equation could provide the seasonal distribution of potential transpiration but would not accurately reflect its magnitude, so we rescaled the calculated monthly values using the water balance calculations to find a total annual potential transpiration that best predicts the measured mean annual runoff from North Fork Caspar Creek prior to second-growth logging.

Results suggest that interception accounted for about 70 percent of wet-season evapotranspiration before logging *(fig. 3)*. Transpiration may be partially suppressed during rain because water films can block stomatal openings. However, application of the estimated potential transpiration for each month to the average duration of rain in the month suggests that suppressed transpiration can compensate for no more than 13 percent of interception loss. Actual compensation is likely to be less because stomata are most concentrated on the parts of leaves most likely to remain dry.

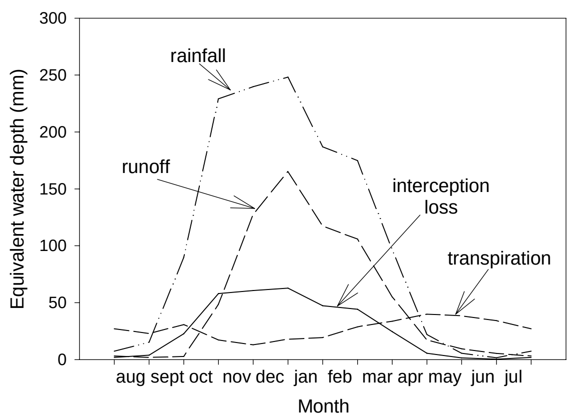


Figure 3—Average annual distribution of components of the North Fork Caspar Creek water budget for pre-logging conditions.

Modeling the effect of altered interception on peakflows

Reduced rainfall interception after logging would increase effective rainfall, thereby increasing both storm runoff and soil moisture recharge. Such changes tend to increase peakflow discharge, as does the accompanying decrease in transpiration. The expected peakflow response to such changes can be estimated by developing a model to predict peakflow from effective rainfall under forested conditions, and then reapplying the model to the same events after modifying effective rainfall to reflect canopy removal from logging *(table 1).* Predicted changes can then be compared to those observed in the treated watersheds. The remainder of this section describes this modeling strategy in more detail and presents modeling results.

Table 1—*Strategy for modeling the influence of interception on peakflows.*

1	Develop equations to predict peakflows in control watersheds from antecedent and in-storm effective rainfall under forested conditions
2	Calculate antecedent and in-storm effective rainfall for each post-treatment storm to reflect conditions after logging at each treatment watershed
3	Apply equations from step 1 to effective rainfalls from step 2 to estimate peakflows for each control watershed, had each been logged similarly to each treated watershed
4	Regress pretreatment peakflows in each treated watershed against mean of peakflows at H and I for the same storms (the "H-I mean"), producing five calibration relations
5	Apply calibration relations to the H-I mean for each post-treatment storm to estimate the expected peakflow for the storm in each treated watershed under forested conditions
6	Apply calibration relations to the results of step 3 to estimate peakflows in each treated watershed on the basis of expected interception and transpiration after logging

Models to predict peakflows from daily rainfall records at Caspar Creek *(table 1,* step 1*)* were constructed for the control watersheds because the controls have the longest record of peakflows for forested conditions. Three general runoff modes may be important in the area: surface quickflow, rapid subsurface stormflow, and groundwater-fed flow. These components respond at different time scales and so would be associated with different descriptors of rainfall. The most useful predictor found for peakflow at the Henningson control watershed (P_H, $m^3ha^{-1}s^{-1}$) indeed incorporates three measures of rainfall *(fig. 4a)*:

$$\ln P_H = -10.9 + 1.42 \ln A0_{85} + 0.00448\, A2_{90} - 151\, A2_{99}^{-1} - 1.67\text{x}10^{-6}\, A2_{99}^{2} \qquad (2)$$

with $r^2 = 0.84$, and where $A0_{85}$ is calculated as the rainfall on the day of the peak plus 0.85 times that of the previous day, $A2_{90}$ is a standard antecedent precipitation index calculated for the second day before the peak with a recession coefficient of 0.90, and $A2_{99}$ is an equivalent index having a recession coefficient of 0.99; in both cases, daily transpiration estimated from the water budget is subtracted from rainfall before the index is calculated. The $A0_{85}$ index reflects rain falling during the rise to peak and is likely to be associated with quickflow. $A2_{90}$, which has a half-life of eight days, is set to 0.0 for calculated negative values; this index may be most relevant to subsurface stormflow. In contrast, $A2_{99}$ is allowed to accumulate negative values, and, with a half-life of 70 days, is expected to reflect a dominant control on groundwater-fed flow. The short- and long-term indices most strongly influence results.

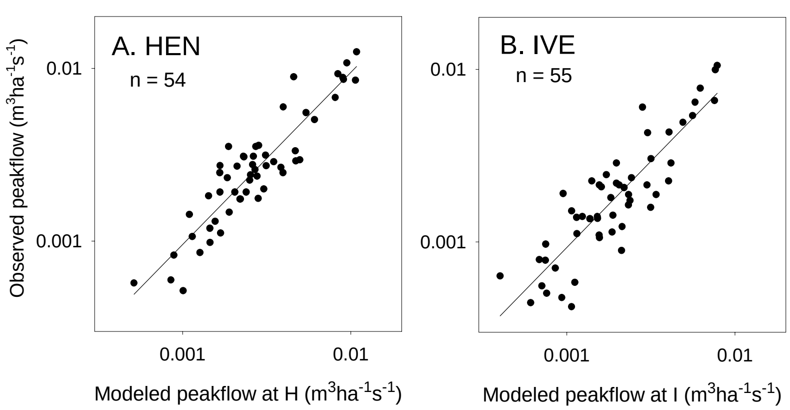


Figure 4—Observed and modeled peakflows at control watersheds A. Henningson and B. Iverson.

Peakflows at a second control watershed, Iverson, are characterized by longer lags and lower unit-area discharges than at Henningson, suggesting that the influence of groundwater may be greater there. The most effective peakflow predictor for Iverson indeed shows a different form than that found for Henningson:

$$\ln P_I = -11.0 + 1.31 \ln A0_{99} + 0.00520\, A2_{88} - 146\, A2_{99}^{-1} \quad r^2 = 0.77 \tag{3}$$

In this case *(fig. 4b)*, the index associated with quickflow ($A0_{99}$) is nearly equivalent to the two-day effective rainfall preceding the peak, and the intermediate index has a recession coefficient of 0.88.

The effect on peakflows of hypothetical logging in each control watershed was then predicted using equations (2) and (3) and incorporating effective rainfall calculated for each post-treatment storm in each treated watershed (*table 1*, steps 2 and 3). Estimates of effective rainfall must take into account interception by both foliage and forest floor litter; after logging, the litter layer is augmented by logging debris, which sometimes is burned. Kelliher and others (1992) found that interception per unit leaf area on live trees is about 3.6 times greater than on slash in a *Pinus radiata* plantation in New Zealand, and this value is assumed for the present analysis. Calculations for clearcut conditions incorporate decreased canopy interception and transpiration, increased litter interception for unburned watersheds, and no increase in litter interception for burned watersheds.

The modeled results for hypothetically clearcut control watersheds must then be converted to apply to the watersheds that were actually logged. This conversion is possible using relations calibrated before treatment between peakflows in control and treatment watersheds. These calibration relations are also used to estimate peakflows that would have occurred in treated watersheds, had they not been logged. Here we use the calibration relations first to quantify the observed changes in peakflow after logging (*table 1,* step 5), and then to calculate the effects of altered interception on peakflows in the treated watersheds (*table 1,* step 6).

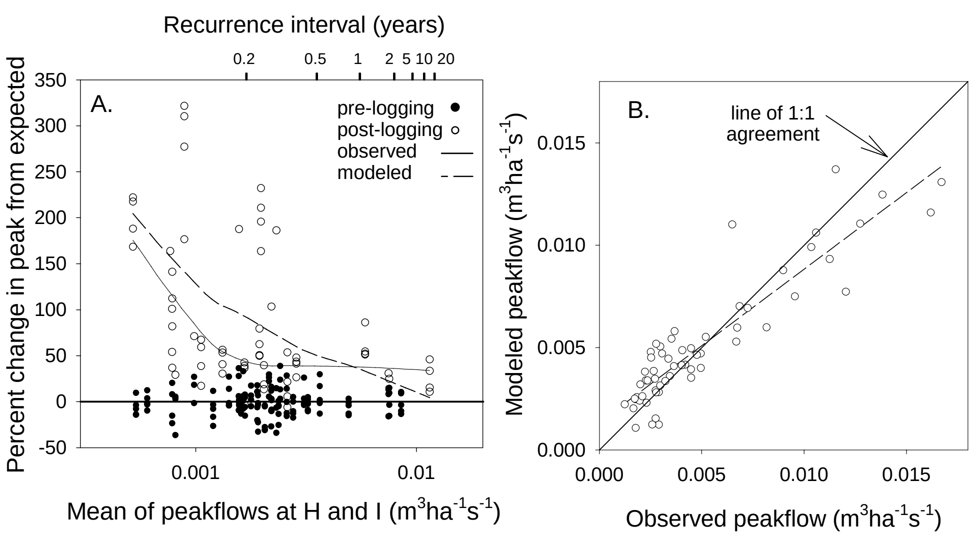


Figure 5—A. Percent deviation of observed and modeled peakflows from expected peakflow before and after logging at five clearcut watersheds as a function of mean peakflow at two control watersheds. B. Comparison of modeled and observed peakflows for two years following logging in the five clearcut watersheds.

More than 95 percent of the canopy was removed in five gauged watersheds at Caspar Creek between 1989 and 1992. Peakflows in each watershed during the five to sever yr preceding logging correlate closely to the average of corresponding peakflows at the control watersheds (*table 1*, step 4), allowing estimation of expected peakflows in the treated watersheds under forested conditions for events taking place after logging *(table 1,* step 5; *fig. 5a)*. Results using these calibration relations suggest that the five watersheds experienced a 54 to 70 percent increase in average peakflow for flows with a greater than 0.15-yr recurrence interval in the first two years after logging *(table 2)*; the average peak in this period corresponds to about a 0.37-yr recurrence interval flow. Small peakflows increase the most, but a loess regression suggests that the proportional increase approaches an asymptote of about 1.34 for flows with recurrence intervals greater than 0.3 yr; the largest peakflow (estimated recurrence interval of 13 yr) in the two years following logging increased an average of 26 percent. These data are also presented by Ziemer (1998, his Figure 3) but our analyses differ: we exclude data from more than two years after logging and calculate recurrence intervals using the 17-yr record now available.

Table 2—*Observed and modeled peakflow changes for two years following logging.*

Gauge	Area (ha)	Percent logged	Observed peakflow increase (percent)	Modeled increase from interception (percent)	Modeled increase from interception and transpiration (percent)
Ban	10	95.0	56	44	58
Car	26	95.7	54	50	67
Eag	27	99.9	58	40	64
Gib	20	99.6	70	45	69
Kje	15	97.1	67	35	78
Mean		***97.5***	***61***	***43***	***67***

The calibration relations between peakflows in treated and control watersheds were next used to convert the peakflows predicted for hypothetically logged control watersheds to corresponding peakflows in each treated watershed (*table 1*, step 6). For the two years after logging, modeled average peakflows for treated watersheds are 58 to 78 percent higher than expected under forested conditions, compared to the observed increases of 54 to 70 percent *(table 2)*. Modeled results tend to overestimate changes for small peaks and underestimate them for large peaks *(fig. 5b;* dashed curve on *fig.5a)*, but, overall, changes in interception and transpiration appear to be sufficient to account for observed increases in peakflow.

Finally, modeled results were recalculated assuming no change in transpiration after logging. The resulting mean increase in average peakflow is 43 percent *(table 2)*, suggesting that about two-thirds of the observed peakflow increase may be due to altered interception and the rest to altered transpiration.

Potential effects of interception on landslide frequency

Increased effective rainfall after logging is expected to contribute to increased sediment production from erosion processes associated with rain or wet conditions, such as bank erosion, gullies, shallow landslides, and earthflows. The potential influence of altered interception on rates of shallow landsliding can be calculated for areas where relationships between storm rainfall and landslide rates have been identified.

A relationship between areal density of landslides and storm rainfall has been defined for deforested lands in a New Zealand watershed (Reid and Page 2003). Pine plantations in the same watershed show canopy interception rates of 35 percent (Pearce and others 1987), but we carry out calculations assuming that the asymptote of 21 percent interception loss for the largest storms at Caspar Creek applies for forested conditions in order to test the potential influence of the Caspar Creek rates. The original landslide relationships for the New Zealand site were defined using gauge rainfall, which does not account for interception by grass. If grasses are assumed to intercept four percent of rainfall, gauge rainfall can be transformed to effective rainfall in the grassland areas, and the relationships can be recalculated to reflect only effective rainfall. The long-term rainfall records for the area can then be modified to reflect effective rainfall under forested conditions. Long-term landsliding rates for forested conditions were estimated by applying the recalculated landslide relationships to the modified long-term rainfall records to predict landslides generated by each storm. Results suggest that reforestation of grasslands at the New Zealand site would decrease landslide frequency by about 50 percent due to altered storm interception alone if interception rates were similar to those at Caspar Creek. Increased root cohesion would further reduce landsliding after reforestation, as would decreases in seasonal soil moisture caused by increased transpiration and foliar interception loss during smaller, non-landslide-generating storms.

Conclusions

Measurements in a second-growth redwood forest indicate that about 22.4 percent of the incoming rainfall does not reach the ground because it is intercepted by foliage. Foliage interception is expected to decrease after logging and then to gradually increase as vegetation regrows. Given the interception rates measured under forested conditions, the expected changes in interception and transpiration after logging are sufficient to account for the observed increases in peakflow in clearcut tributaries of the North Fork Caspar Creek watershed. Rainfall-peakflow models suggest that, on average, about two-thirds of the change is due to decreased interception and the rest to reduced transpiration.

Decreased interception after logging also holds implications for sediment generation. For example, a rainfall-landslide model developed for a deforested site in New Zealand suggests that shallow landslide rates would have doubled due to loss of canopy interception alone if rainfall interception rates had been equivalent to those measured at Caspar Creek.

Acknowledgements

Rand Eads, Elizabeth Keppeler, Elias Steinbuck, Randi Field, and Jason Fisher aided with various phases of the study. Funding was provided by the California Department of Forestry and Fire Protection and the USDA Forest Service. Robert Ziemer, Thomas Lisle, and an anonymous reviewer provided useful comments.

References

Dawson, T.E. 1996. **The use of fog precipitation by plants in coastal redwood forests**. In: LeBlanc, J., editor. Conference on Coast Redwood Forest Ecology and Management; 1996 June 18-20; Humboldt State Univ., Arcata, CA. Berkeley, CA: University of California Cooperative Extension, Forestry; 90-93.

Dunne, T.; Leopold, L.B. 1978. **Water in environmental planning**. San Francisco, CA: W.H. Freeman and Co.; 818 p.

Fahey, B.D. 1964. **Throughfall and interception of rainfall in a stand of radiata pine**. Journal of Hydrology (New Zealand) 3: 17-26.

Gash, J.H.C. 1979. **An analytical model of rainfall interception in forests**. Quarterly Journal of the Royal Meteorological Society 105: 43-55.

Helvey, J.D.; Patric, J.H. 1965. **Canopy and litter interception by hardwoods**. Water Resources Research 1: 193-206.

Kelliher, F.M.; Whitehead, D.; Pollock, D.S. 1992. **Rainfall interception by trees and slash in a young Pinus radiata D Don stand**. Journal of Hydrology 131: 187-204.

Lewis, J. 2003. **Stemflow estimation in a redwood forest using model-based stratified random sampling**. Environmetrics 14(6): 559-571.

Lewis, J., Mori, S.R.; Keppeler, E.T.; Ziemer, R.R. 2001. **Impacts of logging on storm peak flows, flow volumes and suspended sediment loads in Caspar Creek, California**. In: Wigmosta, M.S.; Burges, S.J., editors. Land use and watersheds: human influence on hydrology and geomorphology in urban and forest Areas. Water Science and Application Volume 2. Washington, D.C: American Geophysical Union; 85-125.

McMinn, R.G. 1960. **Water relations and forest distribution in the Douglas-fir region on Vancouver Island**. Publication 1091. Ottawa, Canada: Canada Department of Agriculture, Forest Biology Division; 71 p.

Pacific Watershed Associates. 1998. **Sediment source investigation and sediment reduction plan for the Bear Creek watershed, Humboldt County, California**. Unpublished report prepared for The Pacific Lumber Company. Arcata, CA: Pacific Watershed Associates.

Pearce, A.J.; O'Loughlin, C.L.; Jackson, R.J.; Zhang, X.B. 1987. **Reforestation: on-site effects on hydrology and erosion, eastern Raukumara Range, New Zealand**. In: Forest hydrology and watershed management, proceedings of the Vancouver symposium, August 1987. Publication 167. Wallingford, UK: International Association of Hydrological Sciences; 489-497.

Pearce, A.J.; Rowe, L.K. 1979. **Forest management effects on interception, evaporation, and water yield**. Journal of Hydrology (New Zealand) 18: 73-87.

Reid, L.M.; Lewis, J. In review. **Evaporation of rainfall from foliage in a coastal redwood forest**.

Reid, L.M.; Page, M.J. 2003. **Magnitude and frequency of landsliding in a large New Zealand catchment**. Geomorphology 49(1-2): 71-88.

Spittlehouse, D.L. 1998. **Rainfall interception in young and mature coastal conifer forests**. In: Alila, Y., editor. Mountains to sea: human interaction with the hydrologic cycle, 51st Annual Conference Proceedings; 10-12 June 1998. Victoria, B.C., Canada: Canadian Water Resources Association; 171-174.

Steinbuck, E. 2002. **The influence of tree morphology on stemflow in a redwood region second-growth forest**. Chico, CA: California State University; 55 p. M.S. thesis.

Wosika, E.P. 1981. **Hydrologic properties of one major and two minor soil series of the Coast Ranges of Northern California**. Arcata, CA: Humboldt State University; 150 p. M.S. thesis.

Ziemer, R.R. 1998. **Flooding and stormflows**. In: Ziemer, R.R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek Story. General Technical Report PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, USDA Forest Service; 15-24.

Use of Streambed Characteristics as Ecological Indicators of Long-Term Trends in Sediment Supply Associated With Forest Management on PALCO Lands[1]

Kate Sullivan[2] and Matt O'Connor[3]

In recent years, PALCO has significantly advanced ecological protections on its lands at the as it manages according to provisions in its Habitat Conservation Plan Agreement with federal agencies. This includes the use of watershed assessments and extensive monitoring. A key element of watershed management includes ecological goal-setting and monitoring over time to determine if objectives are achieved. Streambed sediment characteristics have become widely used as indicators of stream conditions relevant to the productivity of fish and aquatic organisms. It is hypothesized that practices in use on PALCO lands will reduce sediment supply from past historical levels with a corresponding change in defined streambed measures. However, despite a basis in geomorphic theory and a robust scientific literature regarding the measurement of parameters in a research context, there are few examples and little discussion of the potential for trend detection in these characteristics. In this paper we consider the use of sediment characteristics in a long-term program monitoring trends in sediment-rich northern California streams.

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] PALCO, P.O. Box 712, Scotia, CA 95565, (707) 764-4492.
[3] O'Connor Environmental Inc., Healdsburg, CA 95448, (707) 431-2810.

Watershed Analysis Results for Mendocino Redwood Company Lands in Coastal Mendocino and Sonoma Counties[1]

Christopher G. Surfleet[2]

Abstract

To assess the needs for conservation, restoration and condition of aquatic habitat within its land Mendocino Redwood Company (MRC) has been conducting watershed analysis. From watershed analysis completed to date, we estimate 73percent of the total sediment inputs over the last 30 to 40 years are road and skid trail associated. Of that percentage 30 percent is road and skid trail associated mass wasting, and 32 percent is road surface and point source erosion, the remaining 11percent is surface and point source erosion from skid trails. Hillslope mass wasting (not associated with roads or skid trails) represents 27 percent of the sediment inputs. Using controllable erosion as an indicator of future sediment yield, MRC estimates there is 2.2 million cubic yards of potential road sediment delivery to be controlled.

Watershed analysis has provided insights into aquatic habitat functions within coastal Mendocino and Sonoma Counties. The following qualitative indices by percent of streams demonstrate the quality of habitat functions: "on target" indicates habitat conditions that meet published targets for well functioning conditions, "marginal" indicates functional habitat conditions but not at optimal levels, and "deficient" indicates low habitat functions with need for improvement. Instream large woody debris (LWD) condition is mainly marginal and deficient with few streams being on target: one percent on target, 35 percent marginal, 35 percent deficient, and 29 percent no data. Stream shade conditions are mainly on target to marginal with some streams being deficient: 29 percent on target, 35 percent marginal, 12 percent deficient, and 24 percent no data. Stream temperature conditions for salmonids are found to be: 58 percent on target, 18 percent marginal, and 24 percent deficient. Salmonid spawning habitats are predominantly on target and marginal (15 percent on target, 35 percent marginal, three percent deficient, 48 percent no data). Salmonid rearing and over-wintering habitats are mainly marginal and deficient, with few on target streams (rearing habitat: one percent on target, 39 percent marginal, 13 percent deficient, 48 percent no data; over-wintering: two percent on target, 37 percent marginal, 13 percent deficient, 48 percent no data).

Generally speaking low LWD levels and high sediment inputs affecting rearing and over-wintering habitat for salmonids are the primary issues that need improvement, to a lesser extent stream temperature and spawning habitat. MRC has developed policies for improvement of riparian conditions for long term LWD recruitment needs of stream habitat. In the short term MRC is promoting the restoration of LWD in streams to improve current conditions. Sediment inputs are dominated by road issues. MRC has committed to upgrading and modernizing its entire road network, a process that will take approximately 30 years. To date MRC has made substantial headway in addressing road erosion and aquatic habitat impacts. In the five years that MRC has owned this land; MRC has removed 11 salmonid migration barriers, decommissioned approximately 10 miles of streamside logging roads, and controlled at least 400,000 cubic yards of controllable erosion. Further, a comprehensive monitoring program will test whether the MRC policies and restoration efforts are improving

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Hydrologist, 2535 NW Hayes Ave, Corvallis, OR 97330, (541) 738-2400. email: Surfleet@comcast.net

aquatic habitat and resource conditions.

Key words: aquatic habitat, forestry, Mendocino County, redwood, watershed analysis

Introduction

Mendocino Redwood Company, LLC (MRC) is a private landowner managing 232,000 acres of redwood and Douglas fir forest in Mendocino and Sonoma Counties, California. To assess the needs for conservation, restoration and condition of aquatic habitat within its land Mendocino Redwood Company (MRC) has been conducting watershed analysis. This report presents the summarization of results from watershed analysis efforts conducted by MRC. The results from the watershed analysis are used to formulate strategies for restoration and conservation of aquatic habitat in association with MRC's forest management activities.

Methods

The watershed analysis by MRC is conducted following modified guidelines from the Standard Methodology for Conducting Watershed Analysis (Washington Forest Practices Board 1995). MRC's approach to watershed analysis is to perform resource assessments of mass wasting, surface and point source erosion (roads/skid trails), hydrology, riparian condition, stream channel condition, and fish habitat. A prescription that guides land management activities is developed when current company policies do not address the issues and processes identified in the watershed analysis.

This report presents information from watershed analysis for 8 separate watershed analysis units[3] (WAU) conducted from 1997 to 2003; representing approximately 70 percent of MRC lands *(table 1)*. From each of the resource assessments of the watershed analysis key indicators of the watershed and aquatic habitat conditions are developed; only select variables are presented in this report to illustrate the general conditions observed regionally. These are:

- sediment input summaries for mass wasting, roads and skid trails;
- stream large woody debris (LWD) conditions;
- stream shade conditions;
- stream temperature conditions;
- fish habitat conditions for spawning, over-winter, and rearing life-stages.

[3] MRC land within major watersheds

Table 1—*Watersheds analyzed in this report.*

Watershed analysis unit	Watershed area (acres)	MRC owned (acres)
Albion River	27,500	15,800
Noyo River	68,000	20,000
Garcia River	73,000	11,800
Hollow Tree Creek	44,400	21,100
Navarro River	201,000	54,600
Willow/Freezeout Creeks	7,500	5,400
Gualala River	42,100	7,900
Big River	75,300	34,000

Sediment inputs are presented as a percentage of road, skid trail and mass wasting sediment inputs. The stream LWD, shade, temperature, and fish habitat conditions are presented in qualitative indices that demonstrate the quality of habitat functions: "on target" indicates habitat conditions that meet published targets for well functioning conditions, "marginal" indicates functional habitat conditions but not at optimal levels, and "deficient" indicates low habitat functions with need for improvement. The levels of habitat condition quality are developed for major tributaries and sections of major rivers within Calwater planning watersheds. The results are presented as percent of number of stream or river segments that demonstrate each habitat condition.

Stream and watersheds conditions are dynamic with natural disturbance occurring stochastically both temporally and spatially. It should not be expected that optimal habitat condition, at a regional scale, be "on target" everywhere at all times. Rather a range of habitat conditions should be expected spatially and temporally. Therefore interpretation of habitat condition is best considered through the distribution of conditions. A distribution of habitat condition skewed toward "on target" would be viewed more beneficial then a distribution skewed toward "deficient." Ultimately, the best indication of favorable habitat conditions should see the regional distribution skewed toward "on target" conditions over time, with deviations expected within watersheds following disturbances.

Sediment Inputs

Sediment input for each WAU is estimated from hillslope mass wasting, road associated mass wasting, road surface and point source erosion, and skid trail surface and point source erosion by Calwater planning watershed within each watershed analysis. The sediment inputs are calculated as an average rate per unit area for the entire time period assessed in each watershed analysis (typically the last three to four decades of the 20th century). From the sediment input rates by Calwater planning watershed the average percentage of sediment input was calculated and presented in this report.

The estimates of mass wasting sediment inputs are developed in watershed analysis through the interpretation of two to five sets of aerial photographs spanning a timeframe that varies from 30 to 40 years, depending on availability of aerial photographs. In addition, there is reconnaissance field-checking of the results. Mass wasting volumes and sediment delivery is estimated with a rate developed by aerial photograph dates completed in the mass wasting inventory. Mass wasting features associated with roads and skid trails are identified in the inventory. In this report the road and skid trail associated mass wasting are combined and reported as road associated mass wasting.

Road associated surface and point source erosion is estimated from a combination of field observations and use of a surface erosion model. Surface erosion is sheet wash and rill erosion from the road prism and point source erosion are gullies or wash-outs of fill material associated with watercourse crossings (excluding mass wasting). A road inventory was conducted in each WAU. Observations of past point source erosion and contributing road lengths for surface erosion sediment delivery is collected. A road-surface erosion model is used to estimate the amount of surface erosion from different road types and conditions observed from the road inventory. The model is found in *Standard Methodology for Conducting Watershed Analysis* (Version 4.0, Washington Forest Practices Board). Point source erosion observed in the field is added to the surface erosion estimate within each Calwater planning watershed to give a rate of road surface and point source erosion.

Future sediment yield is estimated by field observation of controllable erosion volume during the road inventory. Controllable erosion, a term developed by the North Coast Regional Water Quality Control Board for the Garcia River (NCRWQCB 2002), is that soil that can deliver to a watercourse, is human created, greater than 10 cubic yards in size and can be reasonably controlled by human activity. However for our purposes we measure and account for all sites, not just those over 10 cubic yards.

Sediment delivery from surface and point source erosion from skid trails was determined primarily from aerial photograph interpretation with field observations used to support the interpretation. The aerial photograph interpretation for skid trail activity consisted of determining the area harvested by ground based yarding by skid trail density (high, moderate, low) for each photo year. High skid trail density is estimated to contribute 600 tons/square mile/year of sediment[4]. Moderate skid trail density is estimated to contribute 400 tons/square mile/year of sediment, while low skid trail density contributing 100 tons/square mile/year. Results from the South Fork Caspar Creek in the early 1970s suggested that high density tractor logging, with practices used at that time, generated approximately 600 tons/square mile/year (Rice and others 1979) validating our skid trail delivery assumptions. The estimate was then divided by the MRC ownership in each Calwater planning watershed to provide a sediment rate (tons/square mile/year) for each planning watershed.

Stream LWD Condition

Through watershed analyses short-term (20 to 30 year) LWD-recruitment potential from riparian areas is evaluated. In addition, LWD has been sampled from stream segments throughout each WAU. Targets for number of Key LWD by stream size (see *table 6* “on target” category) have been derived to compare current LWD loading. The combination of LWD recruitment potential of riparian areas and instream LWD levels and consideration of the sensitivity of the stream channel to LWD provides for the LWD demand of stream segments *(table 2)*. Through the development of LWD demand the habitat condition is represented *(table 3)*.

[4] This is double the high density skid trail sediment delivery estimates that were used in the watershed analysis reports up to 2003. Therefore, the sediment estimates in this report were doubled from those presented in watershed analysis.

Table 2—*In-stream LWD demand.*

Riparian LWD recruitment potential[1]	Key LWD[2]	Channel LWD sensitivity rating		
		Low	Moderate	High
LOW	On Target	Low	Moderate	High
	Off Target	High	High	High
MODERATE	On Target	Low	Moderate	Moderate
	Off Target	High	High	High
HIGH	On Target	Low	Moderate	Moderate
	Off Target	Moderate	High	High

[1] Riparian LWD recruitment potential ranks large dense conifer stands as high, while hardwood or less dense riparian areas as low, with moderate in between.
[2] Large stable pieces of LWD, see Bilby and Ward (1989).

Table 3—*LWD habitat condition descriptions.*

On Target	>80 percent of watercourses have low or moderate LWD demand, and >80 percent of stream segments meet target number of key LWD pieces.
Marginal	50 to 80 percent of watercourses have low or moderate LWD demand, and >80 percent of stream segments have at least half of the target key LWD pieces desired.
Deficient	<50 percent of watercourses have low or moderate LWD demand, and low numbers of functional or key LWD.

Stream Shade Habitat Condition

Estimates of watercourse shading are derived from field observations and aerial-photograph interpretation. MRC determines effective shade for all perennial watercourses from curves that predict effective shade as a function of bankfull width (EPA 1999, 2000). The habitat condition for stream shade is represented *(table 4)*.

Table 4—*Habitat condition quality for stream shade.*

On Target	>90 percent of perennial watercourses that are within or contribute to the stream/river segment have on-target effective shade.
Marginal	70 to 90 percent of perennial watercourses that are within or contribute to the stream/river segment have on-target effective shade or >70 percent canopy cover.
Deficient	<70 percent of perennial watercourses that are within or contribute to the stream/river segment have on-target effective shade or <70 percent canopy.

Stream Temperature Habitat Condition

Stream temperature has been collected within MRC lands since 1992. The MWAT value (annual maximum seven day average of the daily average temperature) for temperature observations was calculated. Comparing these MWAT values to optimal species-specific temperature ranges (EPA 2000) allowed us to rate water temperature quality for cold water species within watercourses *(table 5)*. To determine stream-temperature quality for individual streams or rivers, we selected the lowest species-specific stream-temperature rating among the salmonid species historically present in that particular watercourse.

Table 5—*Stream temperature habitat condition quality.*

MWAT (°C)	Species historically present		
	Coho only	**Steelhead only**	**Coho and Steelhead**
<15	On Target	On Target	On Target
15 to 17	Marginal	On Target	Marginal
17 to 19	Deficient	Marginal	Deficient
>19	Deficient	Deficient	Deficient

Salmonid Habitat Condition by Life Stage

The quality of fish habitat for spawning, rearing, and over-wintering habitats was rated based on targets derived from (Bilby and Ward 1989, Bisson and others 1987, CDFG 1998a, Montgomery and others 1995, Washington Forest Practices Board 1995) *(table 6)*.

The habitat data are combined into indices of habitat condition for the different salmonid life stages. Measured fish habitat parameters were weighted and given a numeric scale to develop a condition rating for individual life history stages. Parameters were divided into subsets that correspond with individual life history stages (spawning, summer rearing, and over-wintering habitat). Parameters were scored as follows: 1 (deficient), 2 (marginal), and 3 (on target). Parameter weights were applied to the total calculated score as shown below.

Table 6—*Fish habitat conditions.*

Fish habitat parameter	Feature	Fish habitat quality	Fish habitat parameter	Feature
		Deficient	Marginal	On target
Percent Pool/Riffle/Flatwater (By length) (A)	Anadromous Salmonid Streams	<25 percent pools	25-50 percent pools	>50 percent pools
Pool spacing (# pools/bankfull/reach length) (B)	Anadromous Salmonid Streams	≥ 6.0	3.0 to 5.9	≤ 2.9
Shelter rating (shelter value x percent of habitat covered) (C)	Pools	<60	60 to 120	>120
Percent of pools that are ≥3 ft residual depth (D)	Pools	<25 percent	25 to 50 percent	>50 percent
Spawning gravel (E)	Pool Tail-outs Quantity	<1.5 percent	1.5 to 3 percent	>3 percent
Percent embeddedness (F)	Pool Tail-outs	>50 percent	25 to 50 percent	<25 percent
Subsurface fines (L-P watershed analysis manual) (G)	Pool Tail-outs	2.31 to 3.0	1.61 to 2.3	1.0 to 1.6
Gravel quality Rating (L-P watershed analysis manual) (H)	Pool Tail-outs	2.31 to 3.0	1.61 to 2.3	1.0 to 1.6
Key LWD +Root wads/328 ft of stream (I)	Streams≤40 ft. BFW	<4.0	4.0 to 6.5	>6.6
	Streams ≥40 ft. BFW	<3.0	3.0 to 3.8	>3.9
Substrate for over-wintering (J)	All habitat types	<20 percent of Units Cobble or Boulder Dominated	20 to 40 percent of Units Cobble or Boulder Dominated	>40 percent of Units Cobble or Boulder Dominated

Spawning Habitat

E (0.25) + F (0.25) + G (0.25) + H (0.25)

Summer Rearing Habitat

A (0.20) + B (0.15) + C (0.15) + D (0.15) + F (0.15) + I (0.20)

Overwintering Habitat

A (0.20) + B (0.15) + C (0.15) + D (0.10) + I (0.20) + J (0.20)

We rate the overall habitat condition as follows:

1.00 - 1.66 = Deficient

1.67 - 2.33 = Marginal

2.34 - 3.00 = On Target

Results

Sediment Inputs

From watershed analysis completed to date, we estimate 73 percent of the total sediment inputs over the last 3 to 4 decades are road and skid trail associated. Of the total sediment inputs 30 percent is road associated mass wasting, 32 percent is road surface and point source erosion, the remaining 11 percent is surface and point source erosion from skid trails. Hillslope mass wasting (not associated with roads or skid trails) represents 27 percent of the sediment inputs. Using controllable erosion as an indicator of future sediment yield, MRC estimates there is 2.2 million cubic yards of potential road sediment delivery to be controlled. The majority of this controllable erosion, approximately 90 percent, is represented at watercourse crossings with culverts installed.

Stream LWD, Shade and Temperature Habitat Condition

The habitat condition for Instream LWD, stream shade and stream temperature conditions are presented in *Figures 1* and *2. Figure 1* demonstrates that stream shade and temperature conditions are generally favorable for the species present within the MRC lands. The distribution of habitat condition quality for stream temperature skews toward on target conditions, particularly stream temperature. However a large portion of streams exhibit only marginal shade quality and deficient temperature quality suggesting that although shade and temperature conditions are generally favorable, improvements can be made. *Figure 2* demonstrates that instream LWD conditions are not favorable. The majority of the streams exhibit marginal or deficient LWD conditions with few streams being on target. This distribution skews toward deficient conditions.

Salmonid Habitat Condition by Life Stage

The quality ratings for salmonid habitat conditions by spawning, rear, and over-wintering habitat conditions are presented in *figure 3. Figure 3* demonstrates that salmonid habitat conditions vary by life stage. Spawning habitat demonstrates a distribution slightly skewed toward on target conditions however the majority of the observations indicate marginal conditions. Rearing and over-wintering habitat conditions skew slightly toward deficient conditions with the majority indicating marginal conditions; only a few streams show on target conditions. The general trend for all life stages demonstrates needs for improvement, particularly rearing and over-wintering habitat conditions.

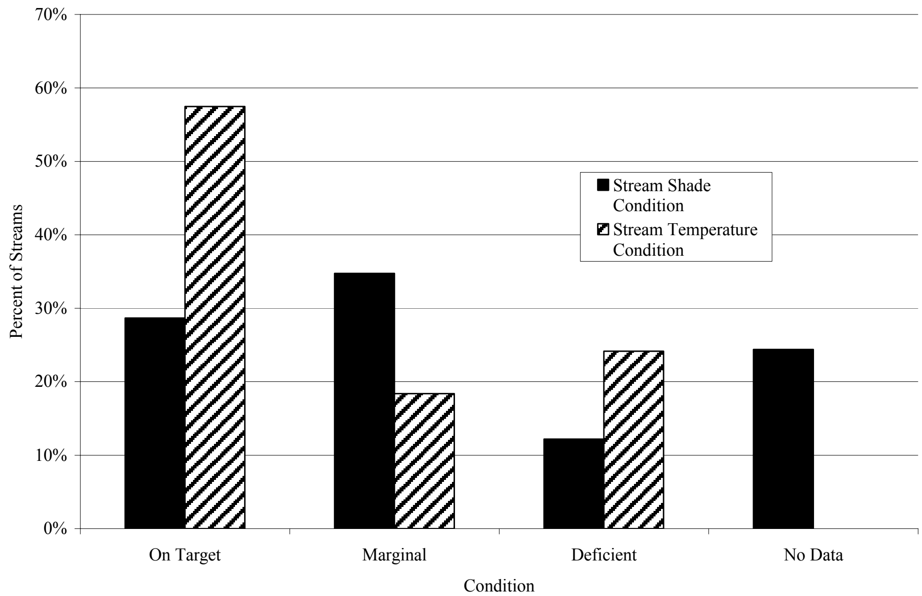


Figure 1—Stream shade and temperature condition.

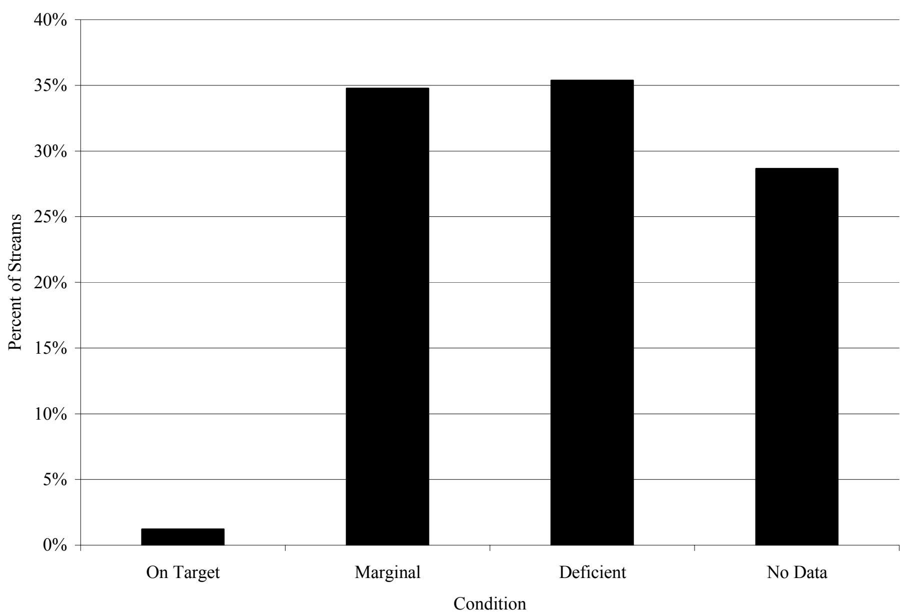


Figure 2—Instream LWD condition quality.

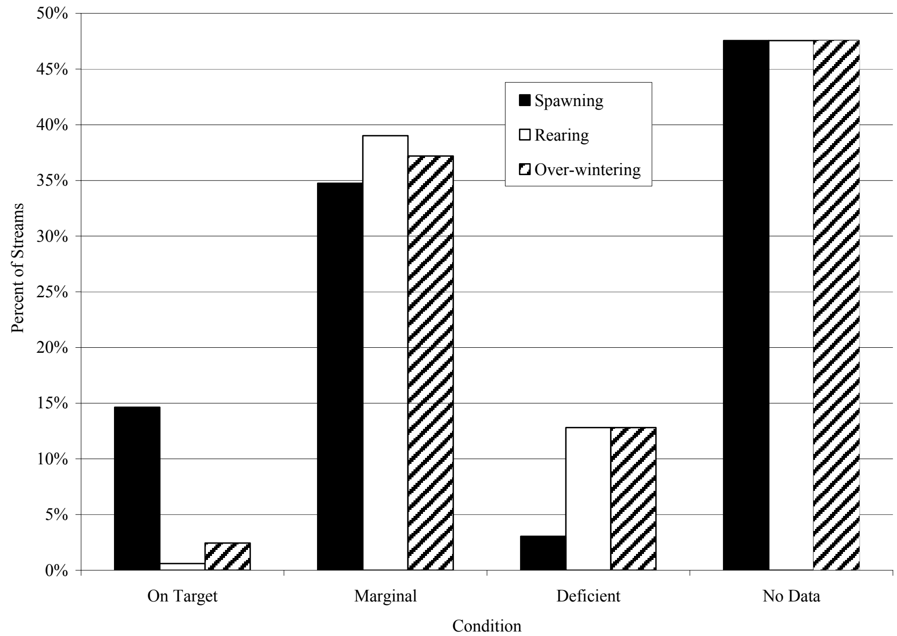


Figure 3—Salmonid habitat condition by life stage.

Discussion

The sediment input information directs our attention to the past effects of forest roads in sediment inputs. Although tractor yarding and hillslope mass wasting has created significant sediment inputs forest roads are highest. This suggests the single most important process that will control significant sediment inputs is in the appropriate design, placement and management of forest roads. This is further substantiated when considering the magnitude of controllable erosion on roads to be addressed (2.2 million yd^3).

The high amount of road associated sediment inputs indicates that a greater proportion of sediment, in the watersheds studied, is occurring from human activities. This indicates an increase in sediment inputs compared to a natural background rate that would not have road sediment associated with it. High sediment yield can be exhibited in stream conditions through several of the variables that relate to salmonid habitat quality such as decreased pool depths and frequency.

Stream shade and stream water temperature habitat condition quality shows a distribution of conditions skewed toward on target and marginal conditions. The data suggests that improvements in stream shade have the potential to improve stream temperature quality. However the habitat conditions suggest a reasonable distribution of stream temperature conditions.

Instream LWD and the riparian conditions to support LWD recruitment as shown by the LWD habitat condition show a need for improvement. A combination of forest harvest in riparian areas and extensive LWD clearing from streams in the past have contributed to these conditions. Large woody debris (LWD) is widely recognized as an important part of the aquatic ecosystem and a vital component of high quality habitat for anadromous fish (for example, Bilby and Likens 1979, Bisson and others 1987, Swanson and Lienkaemper 1978). Improved instream LWD levels and recruitment needs to be managed for.

The increased sediment inputs observed, primarily from roads, and low LWD conditions are apparent in the salmonid habitat conditions within the MRC lands. Salmonid rearing habitat quality requires cold water with deep and frequent pools, over-winter habitat requires deep pools or structure (such as LWD) for aquatic organisms to escape high water flows. Spawning habitat requires sufficient spawning gravels with low levels of fine sediment. From the regional distribution of habitat conditions the conclusion reached is that reduction of sediment inputs and increased LWD are the major factors to improve aquatic habitat conditions and to a lesser extent stream temperature and spawning habitat.

Efforts for Watershed and Aquatic Habitat Improvements

MRC has developed policies for improvement of riparian conditions for long term LWD recruitment needs of stream habitat. In the short term MRC is promoting the restoration of LWD in streams to improve current conditions. Efforts by the California Conservation Corp and the Department of Fish Game to place LWD in streams is encouraged and supported on MRC lands. Further through efforts with State and Federal agencies MRC is attempting to receive permission for greater placement of LWD in streams.

Sediment inputs are dominated by road issues. MRC has committed to upgrading and modernizing its entire road network, a process that will take approximately 30 years. To date MRC has made substantial headway in addressing road erosion and aquatic habitat impacts. In the five years that MRC has owned this land; MRC has removed 11 salmonid migration barriers, decommissioned approximately 10 miles of streamside logging roads, and controlled at least 400,000 cubic yards of controllable erosion.

A comprehensive monitoring program is being developed to test whether the MRC policies and restoration efforts are improving aquatic habitat and resource conditions. This monitoring will be conducted with the intention of informing management decisions to reduce the effects of the forest management on aquatic habitats. This adaptive management process should not only reduce effects on aquatic habitats but work to improve aquatic habitat conditions over time.

References

Bilby, R.E.; Ward, J.W. 1989. **Changes in characteristics and function of woody debris with increasing size of streams in Western Washington.** Transactions of the American Fisheries Society 118: 368-378.

Bilby, R.E.; Likens, G.E. 1979. **Importance of organic debris dams in the structure and function of stream ecosystems.** Ecology, 61(5): 1107-1113.

Bisson P.A.; Bilby, R.E.; Bryant, M.D.; Dolloff, C.A.; Grette, G.B.; House, R.A.; Murphy, M.L.; Koski, K.V.; Sedell, J.R. 1987. **Large woody debris in forested streams in the Pacific Northwest: past, present, and future.** In: Streamside Management: Forestry and Fishery Interactions. Seattle, WA: University of Washington Institute of Forest Resources; 143-190. Contribution 57.

California Department of Fish and Game. 1998. **California Stream Restoration Manual.** 3rd edition. The Resources Agency, Sacramento, CA.

Montgomery D.R.; Buffington, J.M.; Schmidt, R.D. 1995. **Pool spacing in forest channels.** Water Resources Research 31: 1097-1104.

North Coast Regional Water Quality Control Board. 2002. Garcia River watershed water quality attainment plan for sediment. Amendment to the Water Quality Control Plan for the North Coast Region (Basin Plan).

Swanson, F.J.; Lienkaemper, G.W. 1978. **Physical consequences of large organic debris in Pacific Northwest streams.** General Technical Report PNW-GTR-69. Portland, OR: Pacific Northwest Research Station, Forest Service, U.S. Department of Agriculture.

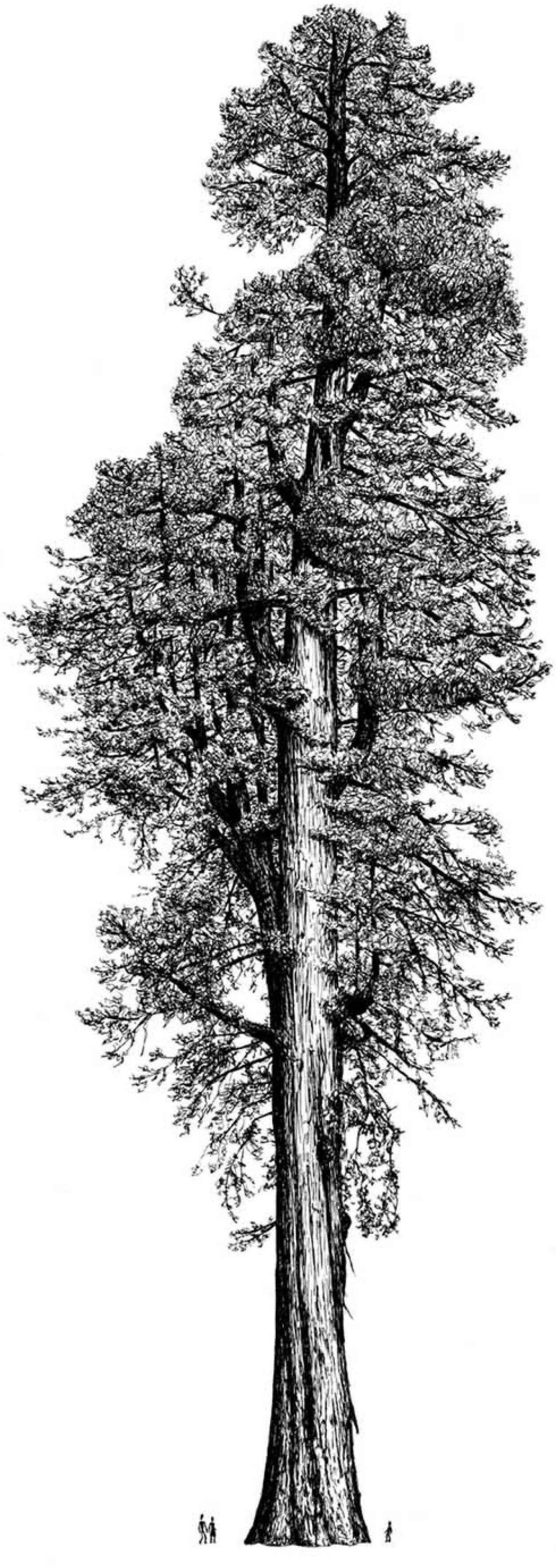
United States Environmental Protection Agency. 1999. South Fork Eel River Total Maximum Daily Loads for sediment and temperature. Region IX, San Francisco, CA.

United States Environmental Protection Agency. 2000. Navarro River Total Maximum Daily Loads for temperature and sediment. Region IX, San Francisco, CA.

Washington Forest Practice Board. 1995. Standard methodology for conducting watershed analysis. Version 4.0. WA-DNR Seattle, WA.

SESSION 4

Wildlife and Fisheries I



Detecting the Upstream Extent of Fish in the Redwood Region of Northern California[1]

Aaron K. Bliesner[2] and E. George Robison[3]

Abstract

The point at which fish use ends represents a key ecological and regulatory demarcation on state and private forest land in the Redwood region. Currently, the end of fish use and other key demarcations with stream classification are measured or estimated based on judgments of Registered Professional Foresters and aquatic biologists with little guidance from empirical study of the issue in this region.

This study establishes a standard protocol for the detection of the end of fish use in streams, which reduces the effort required to produce quality data. During electrofishing the maximum distance traveled between fish presence in delineated habitat units was 289 m. This distance decreased as the slope of the channel increased. In 63 percent of the streams the distance was less than 20 m and in 96 percent the distance was less than 100 m. Fish were found in all habitat units the most abundant was pools at 56 percent of the time. This study has indicated that a pool based method will not give an accurate indication of the end of fish use in streams. A more accurate method should be based on distance were the all habitat units are shocked. In streams with low gradient a minimum of 300 m should be surveyed during the study. In streams were a gradient break of a minimum of eight to12 percent exist this study has indicated the 60 m is sufficient to survey to indicate the Class I, Class II break.

Key words: channel gradient, Class I, Class II, electrofishing protocol, headwater streams

Introduction

Current regulations in California dictate the degree of riparian buffer zone protection to be allocated based on the stream's classification. In California there are three categories, Class I (fish bearing), Class II (aquatic life), and Class III (without aquatic life) (CDF 2004). In recent years the distinction between where Class I ends and Class II begins as you move upstream has become increasingly more contentious. During the last five years the California Department of Fish and Game has refuted 85 percent of the watercourse classifications submitted in timber harvest plans during pre-harvest inspections (BOF 2003).

Extensive research has been conducted on the critical habitat requirements for salmonids in the Pacific Northwest Ecoregion (CDFG 1998, Spence 1996). However, little work has been conducted with trout near the maximum upstream range. Work

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Graduate Student, Department of Forestry and Watershed Management, Humboldt State University, 1 Harpst Street, Arcata, CA 95521. email: akbliesner@yahoo.com
[3] Professor, Department of Forestry and Watershed Management, Humboldt State University, 1 Harpst Street, Arcata, CA 95521.

has been conducted on the upper limits of fish in Washington, Oregon, British Columbia and Alaska (Latterell and others 2003, MELP 2001, ODF and ODFE 1995, Sullivan 1997). The upper extent of fish distribution remains poorly understood in the redwood region of Northern California and a clear field tested methodology for testing the upper limits of fish in headwater streams has yet to be designed for this region.

The California Forest Practice Rules make no distinction of species of fish required to deem a stream a Class I. In this region resident cutthroat trout *(Oncorhynchus clarki clarki)* and rainbow trout *(Oncorhynchus mykiss)* are able to maintain viable populations in rugged headwater environments (McPhail 1967, Wydoski and Whitney 1979). The headwater environment can be affected by land management practices. Biological factors such as sources of food, water quality, biological interactions, flow regime, and habitat structure (Karr 1991), particularly in managed systems may influence the distribution of fish in headwater streams (Gordon and Forman 1983, Odum 1985, Rapport 1992, Steedman and Regier 1987).

Latterell and others (2003) found that timber harvest in the state of Washington had very little effect on the upstream extent of resident trout populations unless impassable culverts were present. Past management practices and current regulations may in fact lead to an extension of the upstream limit of fish. The practice of removing large woody debris obstructions and other barriers up until the late 1960s may have allowed fish easier access to headwater regions (Narver 1971). This coupled with hydraulic processes associated with timber harvest (for example, reductions in evapotransporation leading to increased summer runoff and an increase in the active channel up into the watershed) allow for increased opportunity for fish to colonize headwater portions of the stream (Ziemer and Lisle 1998).

The protocol developed for Oregon requires a minimum of 45.7 m of stream to be electrofished without fish present and a minimum of six pools before the end of fish use can be determined (ODF and ODFE 1995), similar methods were used in Washington. In British Columbia the distance of 100m electrofished was sufficient without fish present to be classified as a Class II stream MELP 2001. More recently Latterell and others (2003) used 400 m minimum unless the gradient of the stream exceeded 30 percent. In the state of California methods for determining the presence/absence of coho salmon *(Oncorhynchus kisutch)* were developed by the Department of Fish and Game (Preston and others 2002). These methods describe the electrofishing of ten pools before absence determined.

Data are not yet available to ascertain whether or not these methods are accurate in determining the end of fish use in Northern California. For this reason this protocol has been designed to be the most detailed and comprehensive field investigation into the end of fish usage in headwater streams. In this paper, a field sampling method is developed and described that insures consistent comparable results and accurate determination of the end of fish use. The objective of this paper is to examine the end of fish use in streams under rigorous field examinations. Then develop a protocol that will define the class I, class II break in small headwater streams that is both accurate and cost effective. The development of this protocol is a portion of a larger study designed to correlate landscape features to the end of fish use for all tributaries within the redwood region of Northern California.

Methods

All data was collected within the 84,987 hectare land holdings of the Pacific Lumber Company near Scotia, California. The tributaries within the major watersheds sampled were Freshwater Creek, the Eel River, Elk River, Van Duzen River, and Mattole River. The maximum inland extent of the study range follows closely the inland extent of redwood in this region.

Electrofishing Protocol

The starting point for electrofishing was determined using maps and current class I, class II breaks. Moving upstream, habitat units were delineated as defined by McCain and others (1990). Visual inspections were conducted until fish were no longer visibly present. At this point electrofishing was started. During all field operations there was one person electrofishing and one person netting to help insure that fish were not missed. Each habitat unit was electrofished until the last fish was documented. At this point a 20 pool electrofishing survey was started. The survey consisted of shocking all habitat units but keeping only a tally of the number of pools to indicate the point to stop the survey. None of the previously stated methods for determining the last fish delineate the habitat units and few shock other units besides pools. If a fish was encountered during the 20 pool survey, the survey was started again. Pools were defined as per McCain and others (1990) and had a maximum depth greater than 0.25 m. No streams were encountered that did not meet this requirement (in other words, 20 pools with depths greater than 0.25 m).

Often in headwater streams portions of the active channel can become dry with the majority of the water going underground. Large wood can also obstruct the active channel completely making electrofishing impossible. In theses cases the unit codes were subsurface or complex in the case of large wood and not electrofished. These units were not used as part of the 20 pool count. Each unit was shocked for ample time to achieve an adequate depletion and shock time reflected the quality of the habitat. All shock times were recorded.

During the first year of this study additional units were shocked in streams were suitable habitat appeared to test the protocol. If suitable habitat was found and there were no blockages present above the 20 pool survey then a rapid 20 pool protocol was conducted to test for resident populations of fish. In this method 20 pools were electrofished and units in between were ignored.

Fish were identified to species and fork length was measured (mm). Amphibians were removed from the shock field and identified to species. Fish and amphibians were replaced in the units they were removed from during electrofishing.

Correlating Landscape Features

At each site three flow measurements were taken. In headwater streams it is difficult to use the velocity area approach to quantify discharge because headwater streams tend to be too shallow to obtain accurate readings. Instead, flow measurements can be taken accurately with a bucket of a known volume. In this method the bucket is placed at a drop where the water is funneled down to a width less than the diameter of the bucket. The time that it takes the bucket to fill was recorded. This procedure was repeated three times at each site and an average of the three was recorded as the discharge.

Temperature (ambient and water), conductivity, pH and a turbidity sample were taken at the start of each new reach to calibrate the electrofisher to water conditions.

Step and fall vertical heights (m) were recorded from the water surface at the bottom to the water surface at the top and the nature of the blockage was recorded (for example; rock, wood, or bedrock). Three depth and three wetted width measurements were taken when possible along with a max depth (m) and pool sill depth (m). The total length (m) of each was also recorded. Several bank full depths (m) were taken within the reach whenever possible. The substrate dominate and co-dominate of each unit was estimated in terms of composition and competency.

Gradient measurements were taken 100 m below and 100 m above the last fish documented. A water level was created and consisted of a 15 m hose filled with water, with both open ends placed between two stadia-rods and held with a temporary tie. The water level was calculated simultaneously at each rod and the distance between the two rods was recorded to determine the overall gradient. Orange flagging was left at the upstream location of measurement, enabling the downstream surveyor to locate the upstream spot and maintain a continuous survey. This was repeated throughout the entire reach until the survey was completed. To calculate gradient the following equation was used $g = (h\text{-}h^2{*}d\text{-}1) *100$ (Walkotten and Bryant 1980) where g is the gradient, h is the downstream measurement, h^2 is the upstream measurement, and d is the distance between measurements. For streams where instream complexity or location prevented surveys, gradient was calculated using Terrain Navigator Pro®[4] (version 3.0 2004).

For each stream surveyed the total watershed area was calculated. The end of fish use points collected were located with GPS equipment when possible and then located on a map. The majority of the sites selected for this study were in steep valleys and heavily wooded. At times these two factors made it impossible to receive accurate GPS readings. At sites where GPS coordinates were not possible to obtain, stream lengths were used to identify end of fish. This data was then entered into mapping software (Terrain Navigator Pro®[4] version 3.0, 2004) and watershed areas and slopes were delineated.

Results

A total of 37 tributaries in Freshwater Creek, the Eel River, Elk River, Van Duzen River, and Mattole River watersheds were sampled during the early portion of summer 2003 (May 16 to July 10) and 2004 (April 7 to July 16). Of these 37 tributaries, eight streams that were sampled contained no fish. The 29 remaining steams with fish present were used for the development of the final end of fish this protocol described earlier. Results from this extensive electrofishing survey show that pools should be at least 0.25 m in depth to be counted as a pool for this protocol. The minimum distance traveled during the 20 pool protocol was 104 m.

Every creek received at least the extended protocol detailed above, of electro-fishing twenty pools and all habitat types in between. A total of 60 pools were electroshocked on Cummings Creek West, the stream with the largest watershed area above the end of fish use. In three streams, 40 pools were electroshocked. During these extended efforts no fish were detected. This indicated that the 20 pool protocol

[4] The use of trade of firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any project or service.

was sufficient and after the first year the rapid protocol in addition to on top of the original method was abandoned. In all cases the initial standard protocol was sufficient in defining the end of fish presence.

At or near the end of fish in streams the only fish captured were either cutthroat or rainbow trout. The average fork length of all trout collected at the upstream limit of their distribution was 154 m with a (range of 102 to 254 m) (standard deviation = 53 mm). Only one trout demonstrated signs of being anadromous, displaying standard signs of smoltification (change in body shape, silver color, and loss of parr marks). This particular fish was captured in came from Graham Gulch, which has the most direct rout to the ocean of any tributaries sampled in this study Young-of-the-year fish were not present within a maximum of 57 m of the end of fish presence, with the exceptions of two creeks, Dunlap Gulch and an unnamed tributary to Lawrence Creek. At these two creeks the last fish present was a young-of-the-year fish.

Stream gradient appeared to play a major role in excluding fish from reaches. Stream gradient was obtained using the water level for all but eight streams, where in-stream complexity and location made use of the water level impossible. Overall, average gradient for all streams surveyed was 10 percent with a range of two percent to 17 percent (standard deviation = 13 percent) *(table 1)*. The average gradient where fish were present was 53 percent (standard deviation = 4 percent) and the average gradient where fish were not present was 11 percent (standard deviation = 16 percent) *(table 1)*. The maximum average stream gradient where fish were present was in Devil's Elbow Creek at 19 percent (standard deviation = 13 percent) *(table 1)*.

Watershed areas ranged in size from 15 ha to 680 ha for all creeks *(table 2)*. Excluding the eight streams without fish present; the smallest watershed area was Fox Creek at 26 ha *(table 2)*. Graham Gulch covers the largest watershed area at 568 ha. The average watershed area for the streams with fish present was 177 ha (standard deviation = 115 ha) *(table 2)*.

A critical component in scrutinizing the electrofishing protocol was the number of habitat units between fish present and the distance between these units. The average number of units between fish was six (standard deviation = seven) *(table 2)*. The average distance between units with fish present was 33 m (standard deviation = 58 m) *(table 2)*. Two creeks, Graham Gulch and Clapp Creek East, proved to have inconsistent numbers of units between fish and subsequently large distances between units with fish present, when compared to the other 35 creeks surveyed in this study. In Graham Gulch, there were 36 units between fish, covering a total distance of 289 m *(table 2; fig. 1)*, which is four times greater than the same distance measurement for any other creek. Clapp Creek East had a maximum of seven units between fish, which was not abnormal, but a distance of 68 m was covered between fish. As a result of the anomalous natures of Graham Gulch and Clapp Creek East, data from these creeks were excluded, and an additional analysis was performed.

When Graham Gulch is excluded, the highest number of units between fish was 12 units and the average was four units (standard deviation = three) *(table 2; fig. 2).* Clapp Gulch East was very complex and most of the units that made up the distance between fish were not electrofished. The average distance between fish present without Graham Gulch and Clapp Gulch East, was 20 m (standard deviation = 19m) *(fig. 2).*

Table 1—*Average gradient (percent) taken for 36 creeks in Northern California using a water level, unless otherwise stated. Values are geometric means (values in parentheses are standard deviations).*

Creek	Gradient with fish present	Gradient without fish present	Total stream gradient
Allen Creek	--a	56.7 (120.7)	56.6 (120.7)
Balcom Creek	--a	6.4 (5.3)	6.5 (5.3)
Bell Creek	7.4 (3.6)	4.3 (3.24)	5.3 (3.5)
Blanton Creek	3.3 (--b)	4.3 (--b)	3.8 (--b)
Browns Gulch	2.4 (2.5)	4.0 (4.5)	3.1 (3.6)
Byron Creek	--a (--b)	4.3 (--b)	4.2 (--b)
Clapp Creek East	8.8 (--b)	29.3 (--b)	19.0 (--b)
Clapp Creek West	5.1 (--b)	27.6 (--b)	16.4 (--b)
Corner Creek	11.2 (7.0)	27.0 (17.4)	20.9 (16.0)
Corrigan Creek	6.1 (11.6)	16.7 (12.1)	11.7 (12.7)
Cummings Creek East	5.4 (1.3)	40.5 (107.0)	21.3 (80.6)
Cummings Creek West	13.0 (4.8)	20.9 (28.4)	13.7 (16.6)
Dauphiny Creek	4.7 (3.0)	9.3 (2.7)	7.0 (3.6)
Devil's Elbow Creek	18.5 (13.0)	23.4 (11.5)	20.6 (12.4)
Dunlap Gulch	4.0 (2.7)	4.2 (4.3)	4.1 (3.3)
Fish Creek	5.1 (2.8)	5.9 (5.0)	5.5 (3.9)
Fox Creek	3.5 (3.1)	3.5 (2.6)	3.5 (2.7)
Graham Gulch	5.9 (2.0)	12.8 (7.0)	9.5 (6.2)
Mardell Creek	5.4 (3.8)	6.1 (3.2)	5.8 (3.4)
McCann Creek	19.4 (29.1)	--c	19.4 (29.1)
Poison Oak Creek East	6.1 (4.5)	15.3 (6.7)	16.2 (6.8)
Poison Oak Creek West	6.1 (4.5)	14.1 (6.6)	10.4 (6.8)
Root Creek East	6.9 (4.2)	76.3 (144.0)	56.8 (125.4)
Root Creek West	--a (--b)	11.9 (--a)	11.9 (--a)
Shively Creek East	3.0 (2.1)	5.5 (4.6)	4.2 (3.7)
Shively Creek West	3.5 (2.4)	8.0 (16.9)	6.2 (13.0)
Strawberry Creek	3.9 (3.3)	8.2 (7.1)	6.3 (6.0)
Strongs Creek Main	5.2 (--b)	19.5 (--b)	12.4 (--b)
Strongs Creek East	1.0 (--d)	5.7 (27.5)	2.4 (27.3)
Strongs Creek West	--a	44.6 (73.5)	44.4 (73.5)
Unnamed Paired Creek to Van Duzen River East, Confluence	2.6 (2.5)	11.5 (10.9)	6.4 (8.4)
Unnamed Paired Creek to Van Duzen River West	--a	3.8 (4.2)	3.8 (4.2)
Unnamed Tributary to Van Duzen River East	9.6 (3.0)	5.8 (4.2)	6.5 (4.2)
Unnamed Tributary to Van Duzen River West	--a	18.4 (15.6)	18.4 (15.6)
Unnamed Tributary to Lawrenece Creek (Road 8)	10.0 (6.3)	17.2 (23.5)	13.4 (16.7)
Unnamed Tributary to Yager Creek	3.6 (18.8)	36.0 (12.1)	33.6 (22.8)
West Fork Elk River	0.7 (--b)	18.1 (--b)	9.4 (--b)
Total	4.8 (4.5)	17.7 (16.2)	14.1 (13.4)

[a] No fish present in creek
[b] Gradient calculated using Terrain Navigator Pro®
[c] Culvert blocking fish passage
[d] One gradient measurement taken

Table 2—*Summary of habitat units sampled below the Class I, Class II break for 36 creeks in Northern California.*

Creek	Total units	Units with fish	Units without fish	Pools	Low gradient riffles	Vertical units [b]	High gradient riffles	Runs	Farthest between [c]	Farthest between distance (m) [d]	Area (ha)
Allen Creek [a]	0	0	0	0	0	0	0	0	0	0	250
Balcom Creek [a]	0	0	0	0	0	0	0	0	0	0	110
Bell Creek	4	3	1	2	2	0	0	0	1	2.7	129
Blanton Creek	9	9	4	5	4	0	0	0	0	2	680
Browns Gulch	4	1	3	1	0	0	0	0	3	13.7	212
Byron Creek [a]	0	0	0	0	0	0	0	0	0	0	528
Clapp Gulch East	36	14	22	11	2	0	0	1	7	68	117
Clapp Gulch West [a]	0	0	0	0	0	0	0	0	0	0	33
Corner Creek	1	1	1	0	0	0	0	0	0	--	454
Corrigan Creek	1	1	1	0	1	0	0	0	0	--	35
Cummings Creek East	14	8	6	1	3	1	1	0	3	14.6	135
Cummings Creek West	9	1	8	0	0	0	1	0	8	7.6	243
Dauphiny Creek	10	4	6	4	0	0	0	0	2	5.8	153
Devil's Elbow Creek	27	13	14	6	2	0	5	0	7	59.7	138
Dunlap Gulch	81	15	66	10	3	0	0	2	12	47.6	121
Fish Creek	1	1	1	0	0	0	0	0	0	--	111
Fox Creek [e]	23	10	13	13	0	0	0	0	--e	--	26
Graham Gulch	64	6	58	6	0	0	0	0	36	283.9	568
Mardell Creek	8	2	6	2	0	0	0	0	6	39	228
McCann Creek	1	1	0	1	0	0	0	0	0	--	165
Poison Oak Creek East	29	12	17	10	1	0	0	1	3	7.4	247
Poison Oak Creek West	29	12	17	10	1	0	0	1	3	7.4	94
Root Creek East	20	5	15	3	3	3	6	0	4	6.7	174
Root Creek West a	0	0	0	0	0	0	0	0	0	0	61
Shively Creek East	127	88	39	0	13	26	0	0	0	--	75
Shively Creek West	101	83	18	0	10	8	0	0	1	5.1	208
Strawberry Creek	1	1	0	1	0	0	0	0	0	--	375
Strongs Creek Main	22	6	16	6	0	0	0	0	3	38.1	70
Strongs Creek East	2	1	1	0	1	0	0	0	1	1.7	233

Creek	Total units	Units with fish	Units without fish	Pools	Low gradient riffles	Vertical units [b]	High gradient riffles	Runs	Farthest between [c]	Farthest between distance (m) [d]	Area (ha)
Strongs Creek West a	0	0	0	0	0	0	0	0	0	0	245
Unnamed Paired Creek to Van Duzen River East, Confluence	20	5	15	5	0	0	0	0	6	44.4	15
Unnamed Paired Creek to Van Duzen River West	13	4	9	4	0	0	0	0	3	32	145
Unnamed Tributary to Van Duzen River East	5	3	2	2	1	0	0	0	1	2.9	297
Unnamed Tributary to Van Duzen River West a	0	0	0	0	0	0	0	0	0	0	330
Unnamed Tributary to Lawrence Creek (Road 8)	39	25	14	14	7	0	3	1	2	7.6	104
Unnamed Tributary to Yager Creek	2	1	1	1	0	0	0	0	1	13.2	45
West Fork Elk River	73	25	48	22	3	0	0	0	9	56.3	55
Totals	776	356	420	140	52	38	16	6		--	7208
Average (standard deviation)									3 (6)	33.9 (58.3)	195 (157)

[a] No fish present in creek – entire creek classified as Class II.
[b] Vertical units include cascades, steps, and falls.
[c] Farthest between refers to the total units where fish were not present.
[d] Farthest between distance refers to the total distance between units where fish were not present.
[e] Rapid 20 pool protocol used – only pools were electro-shocked.

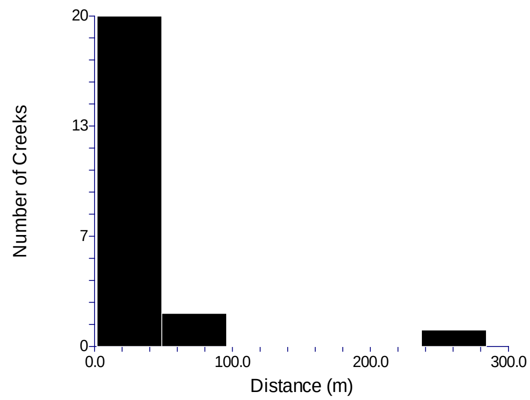


Figure 1—Histogram showing the maximum distance between units with fish present during electrofishing surveys of 24 headwater streams in Humboldt County.

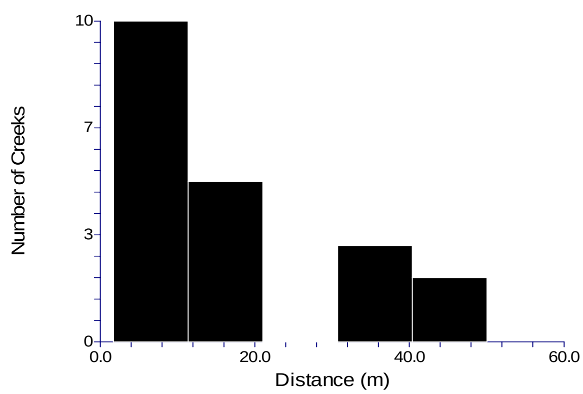


Figure 2—Histogram showing the maximum distance between units with fish present during electrofishing surveys of 24 headwater streams in Humboldt County, excluding two inconsistent creeks, Graham Gulch and Clapp Gulch East.

Fish at or near the upstream limit of their distribution were found in each kind of habitat unit encountered *(fig. 3)*. Fifty-six percent of the trout captured were found in pools, followed by 21 percent in riffles. Only two percent of the trout were captured in runs, which were also the least encountered habitat unit in these headwater streams.

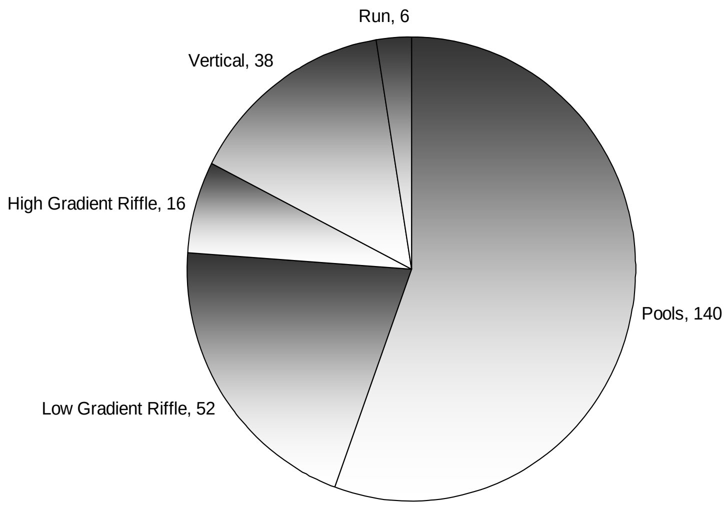


Figure 3—The distribution of fish in habitat units at or near the upstream limit of their distribution in 24 creeks in Northern California.

Discussion

In the absence of a clear barrier to upstream migration (for example, waterfall or impassible chute) a method is required for determining the end of fish use in streams. Methods in the past (Latterell and others 2003, MELP 2001, ODF and ODFE 1995, Sullivan 1997) have not provided a thorough explanation for choosing the set protocols for defining the end of fish use points in streams. This is critical to accurately define the end of fish presence in headwater streams. This does not include vertical units (for example, cascades and steps). Current protocols in Oregon, Washington and British Columbia stipulate the electroshocking of only pools during an end of fish survey. Data collected for this study has indicated that methods based solely on pools is inadequate for documenting the last fish present almost 50 percent of the time. For this reason all habitat units should be eletrofished for accurate placement of the Class I Class II break. Therefore, methods should not be based on pools but on the distance covered in the survey. Graham Gulch had the largest span of units surveyed between the last two fish detections (36) covering a total distance of 289 m. For this creek, 300m would be an acceptable distance to survey to define the Class I, Class II break. Graham Gulch happened to be a lower gradient reach. However the majority of streams with steeper gradients surveyed during this study required far less distances to be covered to define the point were fish end (with 63 percent of the streams being under 20 m and 96 percent being less than 100m).

This study has indicated that the presence/absence of rainbow and cutthroat trout in head water streams is dependent on gradient and the presence of large vertical steps in headwater streams; which is similar to results found in Washington by Latterell and others (2003). Unlike Latterell and others (2003), data from this study indicates that fish due inhabit other habitat units at or near the end of fish presence, other than pools. For this reason we suggest that all stream units should be electroshocked at or near the end of fish presence. As the survey moves closer to the last fish, more electrofishing was required to find fish present in each unit. This is an indication that reaches close to the Class I, Class II break are less populated with fish. Abrupt changes in average channel gradient (for example, 13 percent) can be attributed to the end of fish presence for the majority of streams used in this study. Using this information in concert with the distance covered, survey protocol effort can be greatly reduced. The maximum distance between the last two fish detections covered in streams where a change in gradient occurred was 60 m.

Because of the diversity of watershed sizes encountered (15 ha to 680 ha), methods based on map based criteria, such as watershed area, are problematic. For this reason the watershed area dataset is not useful in determining the Class I, Class II breaks.

The existing 20 pool protocol presented above has proven to be more than sufficient at defining the last fish present within the stream. A streamlined protocol, of 300 m should be electrofished in stream systems were there is not an apparent change in stream gradient greater than a minimum eight to 12 percent and 60 m should be electrofished when this threshold is exceeded. These two methods in combination should be precise enough to define fish absence in headwater streams and robust enough to capture end of fish points in streams where fish populations are low and where distances between habitat units containing fish are high, while decreasing the work load.

References

Board of Forestry and Fire Protection (BOF), Interim Committee. 2003. **Interagency presentation on watercourse classification.** In: 2003 Report of the Watercourse Identification Committee to the Board of Forestry.

California Department of Fish and Game. 1998. **California salmonid stream habitat restoration manual. 3d ed.** Sacramento, CA: Inland Fisheries Division, California Department of Fish and Game; 495 p.

California Department of Forestry and Fire Protection. 2004. **The forest practice rules 14CCR 923.3[943.3,963.3]**.

Gordon, M.; Forman, R.T.T. 1983. **Landscape modification and changing ecological characteristics.** In: Mooney, H.A.; Gordon, M., eds. Disturbance and ecosystem components of response. New York: Springer-Verlag; 12-28.

Karr, J.R. 1991. **Biological integrity: a long-neglected aspect of water resource management**. Ecological Applications 1: 66-84.

Latterell, J.J.; Naiman, R.J.; Fransen, B.R.; Bisson, P.A. 2003. **Physical constraints on trout (*Oncorhynchus* spp) distribution in the Cascade Mountains: a comparison of logged and unlogged streams**. Canadian Journal of Fisheries and Aquatic Sciences 60: 1007-1017.

McCain, M.; Fuller, D.; Decker, L.; Overton, K. 1990. **Stream habitat classification and inventory procedures for Northern California**. FHR Currents: R-5's Fish Habitat Relationships Technical Bulletin; 15 p.

McPhail, J.D.; Lindsey, C.C. 1986. **Zoogeography of the freshwater fishes of Cascadia (the Columbia system and rivers north to the Stikine)**. In: Hocutt, C.H.; Wiley, E.O., eds. The zoogeography of North American freshwater fishes. New York: John Wiley and Sons; 615-637.

Ministry of Environment Lands and Parks (MELP). 2001. Region 6-BC MELP. **Guidelines for non-fish bearing status report preparation.** Habitat Protection Section. British Columbia: Skeena Region; 3 p.

Narver, D.W. 1971. **Effects of logging debris on fish production**. In: Krygier, J.T.; Hall, J.D., eds. Forest land uses and stream environment: proceedings of a symposium. Corvallis, OR: Oregon State University; 100-111.

Odum, E.P. 1985. **Trends expressed in stressed ecosystems.** Bioscience 35: 419-422.

Oregon Department of Forestry (ODF), Oregon Department of Fish and Wildlife (ODFW). 1995. **Surveying forest streams for fish use**. ODF Forest Practices Section. Salem and Portland, OR: ODFW Habitat Conservation Division; 29 p.

Preston, L.; Gilroy, M.; Jong, B. 2002. **Coho salmon presence/absence modified ten pool survey protocol.** Northern California – North Coast Region. : California Department of Fish and Game; 7p.

Rapport, D.J. 1992. **Evaluating ecosystem health**. Journal of Aquatic Ecosystem Health 1: 15-24.

Spence, B.C.; Lomnicky, G.A.; Hughes, R.M.; Novitzki, R.P. 1996. **An ecosystem approach to salmonid conservation.** TR-4501-96-6057. Corvallis, OR: ManTech Environmental Research Services Corp. Available from: The National Marine Fisheries Service, Portland, OR.

Steedman, R.J.; Regier, H.A. 1987. **Ecosystem science for the Great Lakes: perspectives on degradative and rehabilitative transformations.** Canadian Journal of Fisheries and Aquatic Sciences 44(Suppl 2): 95-103.

Sullivan, K.; Liquori, M.; Periann, R.; Glass, D.; Light, J. 1997. **Protocol for surveying the physical characteristics of small streams: Version 1.0 survey level methods**. Washington Forest Protection Association; 14 p.

Walkotten, W.J.; Bryant, M.D. 1980. **An instrument to measure stream channel gradient and profiles.** PNW-345. Portland, OR: Pacific Northwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 5 p.

Wydoski, R.S.; Whitney, R.R. 1979. **Inland fishes of Washington.** Seattle, WA: University of Washington Press.

Ziemer, R.R.; Lisle, T.E. 1998. **Hydrology**. In: Naiman, R.J.; Bilby, R.E., eds. River ecology and management: lessons from the Pacific coastal ecoregion. New York: Springer-Verlag; 43-68.

Using Site-Specific Habitat Information on Young to Late-Successional Avifauna to Guide Use and Management of Coastal Redwood and Douglas-Fir Forest Lands[1]

Sal J. Chinnici,[2] Laura C. Bradley, [3] Daniel R. Dill, [4] and David Bigger[5]

Abstract

Conservation of avifauna in coastal redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudosteuga menziesii*) managed forestlands has historically involved two opposing goals. First, landowners want to minimize restricted acreage in order to maximize commercial utilization of forest products. Second, poor scientific understanding of species habitat needs has often led to conservation strategies that are overly conservative, and therefore restrictive. One solution to this conundrum is to better define critical species habitat needs. The Pacific Lumber Company (PALCO) has been collecting data on a variety of avian species that require different levels of protection based on their conservation status and habitat needs. In this paper, we report on the specific habitat requirements of marbled murrelets (*Brachyramphus marmoratus*), golden eagles (*Aquila chrysaetos*), and Cooper's hawks (*Accipiter cooperii*). For each species, we examined the habitat characteristics surrounding nesting areas and compared them to random locations. We discuss how this research can be used in modifying species-specific conservation strategies.

Key words: avifauna, conservation, Cooper's hawk, Douglas-fir, golden eagle, habitat, marbled murrelet, redwood

Introduction

Land management and conservation of species has traditionally involved resource managers attempting to find compromise between competing, and even opposing interests. For threatened, endangered, or other critical species, management solutions often involve land restrictions or set-asides on public or private land, for example the Late Seral Reserves of the Northwest Forest Plan (USDA and others 1993), or the Marbled Murrelet Conservation Areas (MMCAs) of the PALCO Habitat Conservation Plan (HCP) (PALCO 1999).

Conservation solutions involving land restrictions or set-asides can result in the loss of economic use of private lands. As a result, the development, or negotiation of restrictions on the use of land for the benefit of species can be contentious. Decisions

---

1 This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
2 PALCO, P.O. Box 712, Scotia, CA 95565. email: schinnici@scopac.com
3 PALCO, P.O. Box 712, Scotia, CA 95565. email: lbradley@scopac.com
4 PALCO, P.O. Box 712, Scotia, CA 95565. email: email: ddill@scopac.com
5 Sustainable Ecosystems Institute, 2100 SW River Parkway, 4th Floor, Portland OR 97280. email: bigger@sei.org

involving land use restrictions should be based on the best available scientific information (National Research Council 1995).

Unfortunately, there is often little scientific information available relevant to the specific habitat needs, or potential for disturbance to an individual species. Thus, habitat retention or seasonal buffers for species protection may be extremely conservative and restrictive for the landowner and may also be an inefficient conservation strategy. In addition, it is difficult to assess the efficacy of mitigation without feedback through effectiveness monitoring, or enhanced knowledge of a species habitat needs through surveys and habitat investigations. Therefore, in an attempt to address this ongoing conundrum, we conducted surveys, monitoring, and research on the habitat needs of various listed and other sensitive species over the last several years. In this paper we report on our research of the specific habitat requirements of marbled murrelets, golden eagles, and Cooper's hawks, and discuss how that research can be used to modify species-specific conservation strategies.

Study area and species

PALCO lands encompass approximately 87,817 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)*. For this study, we examined site-specific habitat information for species primarily associated with late successional habitat (marbled murrelet), mid-successional habitat (Cooper's hawk), and young successional habitat (golden eagle).

The marbled murrelet is a small seabird that nests in large trees. The marbled murrelet is listed federally as a threatened species in Washington, Oregon, and California, and is a California state endangered species. Within the study area, marbled murrelets have been primarily associated with old growth or late seral habitat and were most likely to be detected near old growth dominated by redwood (Meyer and others 2004, Stauffer and others 2004).

In 1999, PALCO entered into an agreement with the federal and state governments that included the sale of the Headwaters Forest, a large, un-entered old growth stand of murrelet habitat. This agreement also set-aside for 50 years approximately 3,237 ha of habitat and buffers in a series of MMCAs (PALCO 1999). The MMCAs not only included most of the remaining old growth murrelet habitat on PALCO lands, but also included large buffers of young growth with scattered old growth trees as a conservative measure for habitat retention. Murrelet nesting habitat in the MMCAs is currently protected from disturbance during the nesting season by 0.40 km radius buffers, containing approximately 36.4 km in which timber harvest operations and other land management is restricted for six months each year.

Golden eagles are a California Fully Protected Species, a California Species of Special Concern, and are federally protected under the Bald Eagle Protection Act. In the study area they are inhabitants of the interface of the Douglas-fir, hardwood, and prairie ecotypes. In these mountainous zones, they find suitable nesting substrate in predominant Douglas-fir trees, juxtaposed with large areas of suitable foraging habitat in prairies and young seral stage forest.

Although golden eagles have been extensively studied throughout much of their distribution, (for example, Watson 1997), relatively little is known about golden eagles in north coastal California. We started to extensively survey for golden eagles in 2002. During the breeding season (January to May or June), we conducted intensive stand searches covering >9,308 ha. Stand searches consisted of from four to five biologists traversing forest stands in a transect pattern, searching the forest floor, trees, and canopy for any sign of raptors and their nests. We also conducted ocular surveys from 178 observation stations covering >34,803 ha. Ocular surveys consisted of biologists positioned at prominent viewpoints on ridges and in watersheds conducting stationary surveys of the surrounding landscape with binoculars and spotting scopes. As a result of these efforts, we found eight previously unknown golden eagle nests on or near PALCO lands.

The Cooper's hawk is listed as a California Species of Special Concern whose breeding numbers are thought to have declined in recent decades. Nest site characteristics of Cooper's hawks in the coastal redwood region of northern California are not well known due to the scarcity of nests recorded in the region (Harris 1996). To our knowledge, our work is the first to explore Cooper's hawk nest site selection in the coastal redwood region. PALCO's surveys for other species (for example, northern spotted owls [*Strix occidentalis caurina*]), as well as follow-up stand searches conducted in response to incidental sightings, have led to the discovery of 27 Cooper's hawk nests representing 14 nesting territories, between 1999 and 2003.[6] Because little is known about the ecology of Cooper's hawks in our region there has been little information available on which to base nest protection measures.

Methods

For marbled murrelets, we used 21 confirmed nesting locations. These included 17 sites with eggshells on the ground, three actual nest sites, and one downed chick site. Across the landscape, we randomly selected 63 sites within old growth redwood forests to compare with habitat attributes of murrelet nesting locations. We used Geographic Information System (GIS) technology to describe the type and amount of old growth near each site, distance to nearest stream, and percent canopy closure at each site. We then conducted *t*-tests to compare habitat features between murrelet nesting sites and random sites.

For golden eagles, we collected plot information at eight nests at four spatial scales: 16 m, 0.85 km, 1.6 km, and 3.0 km radius plots. We measured diameter at breast height (d.b.h.) and tree height at the six 16-m plots. For the remaining spatial scales, vegetative types were grouped into foraging and non-foraging habitat, or nesting habitat. The foraging habitat included open ground, prairies, brush lands, scattered oak woodlands, and young forests <10 years old (Madders and Walker 2002, Watson and Whitfield 2002). Forests >10 years old were classified as nesting habitat. We then compared nest plots to 31 random plots. Each random plot was within potential golden eagle nesting habitat and ≥ 3.0 km from a known nest. There was no plot overlap. We conducted independent *t*-tests to compare nest and random plots.

[6] This information was gathered as a result of wildlife surveys of PALCO's forested stands from 1999 to 2003.

For Cooper's hawks, habitat variables were recorded at the most recently used nests in each nesting territory (*n* = 14). The nest site was defined as an 11.3 m radius (0.04 ha) plot centered on the nest tree. Variables recorded at the nest tree included: tree species, d.b.h., tree height (m), and crown percent (crown depth/tree height). Habitat variables recorded at nest sites included: mean canopy cover, tree height, crown percent, tree density, and understory height (m). Random points were selected that were within forested habitat containing trees ≥ 20.5 cm in diameter. Fifty points were generated and a subset of 14 were selected as site level vegetation plots for comparison as paired sites with the most recently used nest in each territory. A paired random site was between 1.6 and 3.2 km, from a nest site.

Again using GIS, we examined habitat level features surrounding the nest and random sites. These variables included: distance to water, edge, and road, stand area, and approximate stand age. We also examined the amount of habitat within different stand types and size classes within 61 m. This is the protection buffer required by CDFG during the breeding season (1 March to 31 August). We used *t*-tests to compare different structural characteristics and habitat features between nest and random sites.

Results

Several marbled murrelet nest site attributes differed significantly from random sites, suggesting that marbled murrelet preference for nesting habitat may be more specific than previously thought (for example, Evans Mack 2003). First, approximately twice as much un-harvested old growth forest surrounded nest sites than random sites *(table 1)*. Further, proportionally more nest sites (0.857 ± 0.076 *s.e.*) were within 500 m of residual old growth stands compared to 0.619 ± 0.061 of the random sites (G_1 = 4.54, P = 0.03). Proportionally more nest sites (0.714 ± 0.099) were in stands with ≥ 75 percent canopy closure than random sites (0.429 ± 0.062; G_1 = 5.28, P = 0.02), and nest sites were closer to a perennial stream (67 m) than random sites (121 m, t_{82} = 1.894, P = 0.06).

Table 1—*Mean (s.e.) ha of un-harvested old growth near marbled murrelet nesting and random sites.*

Distance from site	Nest	Random	*t*	*P*-value
100 m	2.2 (0.2)	1.3 (0.3)	-2.598	0.012
500 m	35.8 (3.8)	15.1 (7.1)	-2.752	0.008

All golden eagle nest trees were in live large Douglas-fir trees. On average, the d.b.h. of nest trees were greater (182.5 cm ± 18.1) than trees in random plots (27.1 ± 11.3, paired-t = 8.857, P = 0.000). Nest trees were also taller (67.9 m ± 5.1) than trees in random plots (19.4 ± 4.0, paired-t = 6.408 P = 0.001). Finally, across different spatial scales, nest plots on average contained 31 percent more foraging habitat than the random plots *(table 2)*.

New Cooper's hawk nests were within 125 m of the previous year's nest and all nine nesting territories receiving more than one visit in the years following nest discovery were reoccupied in subsequent years. All Cooper's hawk nests were in live trees, 57 percent in hardwood, 29 percent in Douglas-fir or Grand fir *(Abies grandis)*, and 14 percent in redwood. The d.b.h. of nest trees ranged from a 25.4 cm grand fir to a 116.8 cm redwood with a mean of 51.1 cm. The mean d.b.h. and height of hardwood nest trees was significantly greater than random, and nest trees also had significantly shallower crowns than random trees *(table 3)*. At the nest site scale (11.3 m from nest tree), Cooper's hawks selected even-aged, mid-seral sites containing larger diameter hardwoods, taller trees with shallow crowns, and a tall understory *(table 4)*. The nest sites were also twice as far from roads as random sites *(table 4)*. Within 61 m of a Cooper's hawk nest, there were three times more hardwood (0.55 ha) than random sites (0.17 ha, U = -2.719, P = 0.007) and there was significantly less conifer-dominated habitat (0.61 ha) compared to random sites (1.00, U = -2.699, P = 0.007).

Table 2—*Mean (s.e.) ha of foraging habitat around golden eagle nest (n = 6), and random sites (n = 31).*

Distance from site	Nest	Random	*t*	*P*-value
0.85 km	81.6 (10.1)	52.9 (5.7)	2.325	0.026
1.60 km	324.7 (44.7)	220.0 (19.2)	2.386	0.022
3.00 km	1072.3 (87.2)	811.1 (45.8)	2.599	0.013

Table 3—*Mean (s.e.) values of tree characteristics at Cooper's hawk nests (n = 14) and random trees (n = 14).*

Characteristic	Nest	Random	*t*	*P*-value
Nest tree height - hardwood (m)	25.0 (1.0)	12.1 (1.4)	7.698 [1]	0.000
Nest tree height - conifer (m)	29.0 (3.9)	33.2 (5.8)	-0.551 [1]	0.564
d.b.h. - hardwood (cm)	46.7 (2.8)	27.4 (8.1)	2.523 [1]	0.027
d.b.h. - conifer (cm)	57.2 (13.5)	49.0 (8.9)	0.505 [1]	0.623
Crown percent	29.1 (3.4)	49.2 (5.3)	-3.322 [2]	0.006

[1] Independent sample *t*-test
[2] Paired *t*-test

Table 4—*Mean (s.e.) values of habitat characteristics at Cooper's hawk within 11.3 m of nest sites (n = 14), and random sites (n = 14).*

Characteristic	Nest	Random	*t*	*P*-value
Tree height (m)	27.8 (2.0)	21.7 (2.5)	2.635	0.021
Crown percent	34.3 (3.6)	53.97 (6.1)	-2.743	0.017
Understory height (m)	8.7 (1.5)	5.6 (1.0)	2.077	0.062
Hardwood density (trees/ha >28-cm d.b.h.)	266.4 (128.4)	61.8 (21.8)	2.164	0.050
Distance to road (m)	191.4 (41.4)	87.1 (19.3)	2.159	0.050

Discussion

Due to the extreme difficulty in locating marbled murrelet nests, most of the information on marbled murrelet inland nesting habitat is based on presence/absence information obtained from audio-visual surveys in old growth and residual old growth forests. However, these surveys are designed simply to categorize old growth forests as 'nesting' or 'non-nesting' murrelet habitat (Evans Mack and others 2003), and not designed to assess relative suitability of old growth forests for nesting (reviewed by Lank and others 2003).

Clearly, not all old growth forests are equally suitable for nesting marbled murrelets. In southern Humboldt County, marbled murrelets selected nesting habitat that had proportionally more un-harvested old growth, a closed canopy, and near streams. In contrast, scattered residual old growth was not selected. These results were consistent with the findings of other studies that compared nesting sites (located by radio telemetry) to randomly generated points on the landscape (reviewed by Lank and others 2003). Although radio telemetry has been considered the most rigorous and unbiased method for locating murrelet nest sites, our results demonstrate the value of using data from opportunistic nest and eggshell findings to investigate habitat selectivity.

These results have important implications for the conservation and management of marbled murrelet nesting habitat in the region. Relative to landowner regulation, murrelet habitat is categorized as nesting or non-nesting habitat based on the results of audio-visual surveys. Those stands classified as nesting, or "occupied" have essentially been given equal conservation value when, in reality, scattered residual old growth stands appear to actually have less nesting value. Results from this study support ranking old growth forest stands in terms of nesting suitability, thereby providing a scientific basis for development of a range of conservation strategies. For instance, an un-harvested old growth stand with a dense canopy near perennial streams may be more suitable for nesting than an open canopy residual old growth stand relatively far from a stream. The un-harvested stand may justify a high level of protection, while more permissive strategies may be appropriate for the scattered, open canopy stand. Interestingly, such a ranking is consistent with regulatory approaches for riparian management, and in some cases may reduce the complexity associated with selecting lands to set aside for murrelet conservation.

Although much is known about golden eagles in other regions or countries this is the first study to investigate habitat relationships in coastal northern California. We found that golden eagles need a large, predominant tree for a nest platform. Similar to our results, Menkens and Anderson (1987) found that tree nesting golden eagles selected one of the largest trees in the stand in both d.b.h. and height.

In terms of foraging habitat, we found that golden eagles select nesting sites that have ample foraging habitat within their nest territory to provide prey for eaglets *(table 2)*. Our results are similar to those in other golden eagle studies, where others have found that open ground for foraging can be a limiting factor, especially in managed forests. For example, McGrady and others (2002) and Madders and Walker (2002) found that eagles prefer areas that are open rather than timbered, and some of their research has shown a correlation between a decrease in breeding success with an increase in afforestation (Watson 1997, Whitfield and others 2001). McGrady and others (1997) noted that golden eagles generally avoid plantation forests because they were probably unable to hunt prey amongst the closely spaced trees. In fact, Bruce and others (1982) found that all 21 nests studied in western Washington were located within 500 m of a clearcut or open field.

In summary, we found: 1) several previously unknown nests of this species, 2) nesting on managed forestland, and 3) extensive foraging habitat surrounding the nests. Thus, considering our results, and the findings of others, we conclude that timber harvesting can be compatible with golden eagle habitat management, as long as seasonal disturbance minimization measures are used during the critical period for nesting golden eagles, and scattered predominant trees are retained for nest structures. Furthermore, given the importance of early seral patches to golden eagle foraging, a decreased timber harvest cycle, and/or an aggressive afforestation policy could actually decrease the overall suitability of habitat for golden eagles on managed lands.

In the coastal redwood region Cooper's hawks selected densely vegetated, even-aged, mid-seral stands that contained tall trees with shallow crowns and a relatively open understory. Most were located in areas where hardwood (for example, tanoak) was the dominant component or comprised a large portion of the stand. Similar nest site structural characteristics have been reported by others (for example, Bosakowski and others 1992, Garner 1999, Reynolds and others 1983). Interestingly, in the redwood-dominated nest stands that did not contain many hardwoods, the birds usually selected either grand or Douglas-fir, rarely selecting redwood as the actual nest tree (2 of 27 nests). Three nests (all in different territories) were located within timber harvesting plans. The CDFG recommends that a 61 m disturbance buffer be established around nest trees during the breeding season. However, after site-specific evaluations of two of the nesting territories it was decided to establish larger buffers around the nest trees to ensure that the core nesting territories were protected. Habitat buffers established included key elements such as the current and previous year's nest trees, roost trees and plucking areas. We also utilized riparian management zones and other species (for example, northern spotted owl) protection buffers to ensure that the nest area was not isolated from adjacent forested habitat.

As found in this and other studies (Jones 1979, Rosenfield and Bielefeldt 1996, Rosenfield and others 1995), Cooper's hawks exhibit strong nest stand reoccupancy. Our results indicate that the establishment of small, yet well-defined habitat buffers should enable the Cooper's hawk to return to their nesting territories in subsequent years, maintaining a balance between timber harvesting and protection for this non-listed species across the managed landscape.

Conclusions

This study is based on habitat associations, and not on cause and effect data. However it does demonstrate important considerations for management of marbled murrelets, golden eagles, and Cooper's hawks that could lead to further testing of habitat hypotheses. For example, based on direct evidence of nesting, murrelets appear to select old growth forest stands with high amounts of canopy closure. The conservation strategy of the PALCO HCP to include in set-asides virtually all occupied murrelet stands based on audio-visual surveys, whether old growth or residual old growth, may have been overly conservative. In addition, this strategy has resulted in hardship to the landowner in terms of restrictions on activities during the harvest season, and lost opportunities for harvest in stands of marginal or non-nesting habitat.

For golden eagles, our investigations into the habitat needs of the species led to a modification of survey area and strategy. The development of the modified strategy has resulted in a more reasonable approach for the landowner in terms of resources involved in surveys, and a reduction in areas seasonally restricted for harvesting activities. Lacking good data, the original conservative survey area was too broad-based, and overly restrictive. Data from this study are useful in refining management guidelines for this species, including information for maintenance of suitable nesting habitat.

For the Cooper's hawk, our results indicate that a conservation strategy for this species can be compatible with timber management. Based on our observations, site-specific protection can be developed by including key elements of nest territories in habitat buffers that incorporate riparian management zones and possibly other species buffers. In addition, retention of habitat components such as large hardwood trees appears important to maintaining Cooper's hawks on the managed landscape.

Finally, the results of this study also have implications for further research and monitoring for these and other species. For example, predictive modeling of habitat using GIS and forest growth models may be based on the habitat information provided in this study so that land managers can evaluate future harvest planning for the potential to impact species, and to develop methods to conserve important habitat areas.

Acknowledgments

We wish to thank the entire Wildlife and Forestry Teams at PALCO for their energy and assistance in gathering the information necessary to prepare this paper. We especially want to thank the field crews who spent countless hours searching the woods and the sky for sign of these animals, including T. Diaz, B. Mauney, A. Logie, M. Freitas, and A. Pontious. We especially thank D. and C. Laughlin who did everything necessary, including climbing trees to put golden eagle chicks back in their nests, and P. Herrera for providing detailed information on the location of marbled murrelet eggshells and downed chicks in the State Parks. We also thank the scientists at the local CDFG, and the USFWS for consulting with us on the conservation of these species over the years. Finally, we want to thank J. Barrett for his thoughtful reviews, and encouragement throughout this project.

References

Bosakowski, T.; Smith, D.G.; Speiser, R. 1992. **Nest sites and habitat selected by Cooper's hawks *Accipiter cooperii,* in northern New Jersey and southeastern New York**. Journal of Raptor Research 106(4): 474-479.

Bruce, A.M.; Anderson, R.J.; Allen, G.T. 1982. **Observation of golden eagles nesting in western Washington.** Journal of Raptor Research 16: 132-134.

Evans Mack, D.; Ritchie, W.P.; Nelson, S.K.; Kuo-Harrison, E.; Harrison, P.; Hamer, T.E. 2003. **Methods for surveying marbled murrelets in forests: a revised protocol for land management and research.** Pacific Seabird Group. Unpublished document supplied by the Pacific Seabird Group.

Garner, H.D. 1999. **Distribution and habitat use of sharp-shinned and Cooper's hawks in Arkansas.** Journal of Raptor Research 33(4): 329-332.

Harris, S.W. 1996. **Northwestern California birds: a guide to the status, distribution, and habitats of the birds of Del Norte, Humboldt, Trinity, northern Mendocino, and western Siskiyou counties, California.** Arcata, CA: Humboldt State University Press; 323 p.

Jones, S. 1979. **The accipiters - goshawk, Cooper's hawk, sharp-shinned hawk**. Habitat management series for unique or endangered species. Tech. Note 335. Denver: USDI Bureau of Land Management; 51 p.

Lank, D.B.; Parker, N.; Krebs, E.A.; Tranqilla, L.M. 2003. **Geographic distribution, habitat selection, and population dynamics with respect to nesting habitat characteristics of Marbled Murrelets (*Brachyramphus marmoratus*).** Burnaby, British Columbia: Center for Wildlife Ecology, Simon Fraser Univ.; 61 p.

Madders, M.; Walker, D. 2002. **Golden eagles in a multiple land-use environment: a case study in conflict management.** Journal of Raptor Research 36 (1 Supplement): 55-61.

McGrady, M.J.; Grant, J.R.; Bainbridge, I.P.; McLeod, D.R.A. 2002. **A model of golden eagle (*Aquila chrysaetos*) ranging behavior.** Journal of Raptor Research 36 (1 Supplement): 62-69.

McGrady, M.J.; McLeod, D.R.A.; Petty, S.J.; Grant, J.R.; Bainbridge, I.P. 1997. **Golden eagles and forestry.** Forestry Commission Research Information Note 292. Edinburgh, Scotland: The Forestry Commission.

Menkens, Jr., G.E.; Anderson, S.H. 1987. **Nest site characteristics of a predominantly tree-nesting population of Golden Eagles.** Journal of Field Ornithology 58(1): 22-25.

Meyer, C.B.; Miller, S.L.; Ralph, C.J. 2004. **Stand-scale habitat associations across a large geographic region of an old-growth specialist, the marbled murrelet.** Wilson Bulletin 116(3): 197-210.

National Research Council. 1995. **Science and the Endangered Species Act.** Washington, DC: National Academy Press; 271 p.

PALCO. 1999. **Habitat conservation plan for the properties of The Pacific Lumber Company, Scotia Pacific Company LLC, and Salmon Creek Corporation.** Scotia, CA: The Pacific Lumber Company; 138 p.

Reynolds, R.T. 1983. **Management of western coniferous forest habitat for nesting accipiter hawks.** Gen. Tech. Rep. RM-102. Fort Collins, CO: Rocky Mountain Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture.

Rosenfield, R.N.; Bielefeldt, J. 1996. Lifetime nesting area fidelity in male Cooper's hawks in Wisconsin. The Condor 98: 165-167.

Rosenfield, R.N.; Bielefeldt, J.; Affeldt, J.L.; Beckmann, D.J. 1995. **Nesting density, nest area reoccupancy, and monitoring implications for Cooper's hawks in Wisconsin.** Journal of Raptor Research 29(1): 1-4.

Stauffer, H.B.; Ralph, C.J.; Miller, S.L. 2004. **Ranking the importance of habitat patches to a threatened species using estimates of occupancy adjusted for uncertainty of detection.** Ecological Applications 14: 1374-1383.

U.S. Department of Agriculture, Forest Service. 1993. **Forest ecosystem management: an ecological, economic, and social assessment.** Report of the Forest Ecosystem Management Assessment Team, Portland, OR; 856 p.

Watson, J. 1997. **The golden eagle.** London: T. & A.D. Poyser; 378 p.

Watson, J.; Whitfield, P. 2002. **A conservation framework for the golden eagle (*Aquila chrysaetos*) in Scotland.** Journal of Raptor Research 36 (1 Supplement): 41-49.

Whitfield, D.P.; McLeod, D.R.A.; Fielding, A.H.; Broad, R.A.; Evans, R.J.; Haworth, P.F. 2001. **The effects of forestry on golden eagles on the island of Mull, western Scotland.** Journal of Applied Ecology 38(6): 1208-1220.

Ecology and Management of Northern Spotted Owls on Commercial Timberlands in Coastal Northern California[1]

Lowell Diller,[2] Keith Hamm,[2] Joel Thompson,[2] and Trent McDonald[3]

Northern spotted owls *(Strix occidentalis caurina)* have been studied on Simpson Resource Company's (formerly Simpson Timber Company) approximately 450,000 acres in coastal Del Norte and Humboldt Counties for the last 14 years. With the exception of a few small isolated old growth stands representing less than 1percent of the ownership, Simpson's land base is made up of a mosaic of second and third growth forests. The primary silvicultural technique has been and continues to be even age management with clearcutting.

Surveys for spotted owls on Simpson's ownership have occurred continuously since 1989 when the first "hooting transects" were established. The following year in 1990, mark-recapture demography and population density studies were initiated. Since then, a Habitat Conservation Plan (HCP) was developed for the species in 1992 and two master's studies were completed in 1993 and 1997. The first master's study was done by Lee Folliard from the University of Idaho and focused on the nesting habitat associations of spotted owls. The data from this study provided much of the basis for the conservation strategy of the HCP. The second master's study, which quantified the habitat associations of spotted owls with low versus high nesting success, was done by Darrin Thome from Humboldt State University (HSU). An "in-house" radio telemetry study was conducted from 1998 to 2000 to quantify night-time ("foraging") habitat associations of spotted owls. As a corollary to the spotted owl studies, two master's studies out of HSU were conducted on dusky-footed woodrats *(Neotoma fuscipes)*; the primary prey of spotted owls in this region. Both of these studies estimated the abundance of woodrats in different aged stands. However, the first by Keith Hamm was conducted in coastal redwood *(Sequoia sempervirens)* forests, while the second thesis by Kevin Hughes was in the Douglas-fir *(Pseudotsuga menziesii)* zone.

During the course of these studies, 783 nests have been located, 1,428 owls have been captured and banded, and movement and foraging data have been obtained using night vision scopes and telemetry on 35 owls. Some of the most important conclusions generated to date from the extensive dataset are as follows:

1. Despite extensive past and current timber harvesting activities, portions of the study area have the highest densities of spotted owls reported in the northwest. Landscapes with the highest densities of spotted owls are a mosaic of mature second growth and recently harvested clearcuts (Diller and Thome 1999).

---

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Stillman Resource Company, PO Box 68 Korbel, CA 95550, (707) 668-4428. email:Ldiller@simpson.com

[3] Western EcoSystems Technology, Cheyenne, WY.

2. Survival rates of adult owls have been high (mean annual rate = 0.85) and the owl population was relatively stable throughout the early 1990s. However, consistent with range-wide trends, presumably mostly due to unfavorable spring weather conditions and reduced fecundity, spotted owl numbers have declined in recent years.

3. Structures used for nesting by spotted owls were approximately equally divided between "enclosed" structural nests (for example, broken tops, cavities and lateral platforms) and "open" nests (for example, abandoned small mammal and bird stick nests and debris platforms) (Folliard and others 2000). Success of nests did not appear to be influenced by the type of nest, but turnover in the pair did have a negative influence on nesting success (Thome and others 2000).

4. Based on a habitat selection model developed using telemetry locations, night-time activity and foraging of spotted owls was associated with sites that were low on the slope, had relatively large trees, but also consisted of a mosaic of stand ages six to 20, 21 to 40 and >41 years of age. Direct observations of spotted owls at night indicated that they did forage in recent clearcuts, but the greatest amount of time was spent in 10 to 20 and >60 year old stands.

5. Both master's studies on spotted owls indicated that residual older trees (trees left from the initial entry) were an important element associated with nesting habitat for spotted owls (Folliard and others 2000, Thome and others 1999). In addition, a habitat selection model developed for successful spotted owl nests in second growth forests indicated nests were most likely to occur low on the slope in mature second growth (60 to 100 years) with substantial amounts of residual basal area. However, in an area around the nest stand, a high amount of edge and good foraging habitat suitability were also included in the model to predict a high probability of locating a successful nest.

6. A mosaic of young and older forest stands provide ideal habitat for spotted owls in coastal northern California, because the owls roost and nest in the older forests, while the young forests provide habitat for their primary prey, dusky-footed woodrats.

In addition to the conclusions described above, there are other factors not specifically addressed in any of the studies on Simpson's ownership that we believe contribute to the occurrence of spotted owls on commercial timberlands. The mild climate and abundant rainfall provide ideal growing conditions, and forest structure regenerates in a fraction of the time that may be required in harsher regions with poorer forest site classes. In addition, the region has an abundance of hardwoods (for example, tanoak, *Lithocarpus densiflorus*, madrone, *Arbutus menziesii* and California Bay *Umbellularia californica*) whose coppice growth insures a significant hardwood component in most managed stands despite efforts to control them. The hardwood component maintains vegetative diversity that is important to promote structural complexity in managed forests.

Our current knowledge of the ecology of spotted owls is sufficient to provide a high level of confidence in knowing how to provide for the future habitat needs of the species on managed timberlands lands. However, it is obviously essential that timber management be implemented in a manner consistent with the needs of spotted owls. The important habitat heterogeneity created by a mosaic of young and old stands is virtually assured by the current California forest practice rules (FPR) regulating even-age silvicultural systems. This habitat component could become problematic if extensive uneven-age management were practiced in coastal northern California.

Probably the single most critical factor in any silvicultural system is the retention and future recruitment of residual structure with late seral habitat elements (e.g. large green wildlife trees and snags). Under Simpson's spotted owl HCP, this type of structure is being retained and recruited as individual tree clumps (10 to 20 trees) or habitat retention areas of 0.5 acre or greater in size. However, the majority of future residual structure will likely be associated with Class I and II riparian management zones that will be the product of current water and lake protection zones required by California FPR and Simpson's aquatic conservation planning.

Although we may be able to provide for the future habitat needs of spotted owls, other threats may exist that are beyond our control. The invasion of northern barred owls *(Strix varia varia)* into northern California, although still a minor component on Simpson's ownership, creates a future unknown factor in the long-term persistence of spotted owls in coastal northern California.

References

Diller, L.V.; Thome, D.M. 1999. **Population density of northern spotted owls in managed young-growth forests in coastal northern Californi**a. The Journal of Raptor Research 33(4): 275-286.

Folliard, L.B.; Reese, K.P.; Diller, L.V. 2000. **Landscape characteristics of northern spotted owl nest sites in managed forests of northwestern California**. The Journal of Raptor Research 34(2): 75-84.

Thome, D.M.; Zabel, C.J.; Diller, L.V. 1999. **Forest stand characteristics and reproduction of northern spotted owls in managed north-coastal California forests**. Journal of Wildlife Management 63(1): 44-59.

Thome, D.M.; Zabel, C.J.; Diller, L.V. 2000. **Spotted owl turnover and reproduction in managed forests of north-coastal California.** Journal of Field Ornithology 71(1): 140-146.

Structural Characteristics of an Old-Growth Coast Redwood Stand in Mendocino County, California[1]

Gregory A. Giusti[2]

Abstract

This paper compares stand characteristics of Old Growth coastal redwood stand densities and forest structure found throughout the northern tier of the range of coast redwood *(Sequoia sempervirens).* Tree densities are relatively low compared to commercially managed stands of coast redwood. Tree size classes distributions vary from <5cm to >254cm (2″ dbh to >100″). Dominant tree species include tanoak *(Lithocarpus multiflora),* Douglas-fir *(Psuedotsuga menzisii)* and Coast Redwood *(Sequoia sempervirens).* Snag densities are given for a stand of old-growth redwoods in Mendocino County.

Introduction

Forestry practices have historically tended to simplify the structural components of coast redwood forests (O'Dell 1996). However, recruitment of the structural elements commonly found in older forests is increasingly recognized as an important management objective in younger forest stands to address issues of biological diversity and forest integrity (Mladenoff 1993, Spies 2003). Obviously, old forest features develop as a function of time; as such they often include stand characteristics not associated with younger stands. Old forest characteristics are often the result of stochastic disturbance events—for example, storms, fire, and landslides, whose legacies shape forest structure and composition and which may be difficult to mimic.

The management for old forest characteristics in coastal redwood forests is a topic of concern and often-heated debate among managers and policy leaders in California. Unfortunately, there exists a limited knowledge base regarding the structure and composition of older forests from which to make sound management decisions. This has lead to the reliance on data sources from other forests in the Pacific Northwest to help guide management decisions across the redwood region. 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.

Study Site

Many of the sites that are now redwood parks along the north coast were recognized for their splendor even during a time when old growth forests were still relatively common. Montgomery Woods State Reserve is no different in this regard from the other impressive stands in the California State Park system.

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] University of California Cooperative Extension, 890 N. Bush Street, Ukiah, CA 93637, 707-463-4495. email: gagiusti@ucdavis.edu

The site is located in a narrow canyon in the headwaters of Big River. It represents one of the few stands of uncut redwoods remaining in Mendocino County. The Reserve is 456 ha (1,142 acres) located on the Comptche-Ukiah road about 24 Km (15 miles) west of Ukiah and 48 Km (30 miles) east of the town of Mendocino. The Reserve was originally acquired in the 1940s through the efforts of the Save-the-Redwoods League. From an original gift of 9 acres, donated by the landowner Robert Orr in 1945, the grove has grown to its present size. The uniqueness of this particular site is the fact that beyond the original nine acre gift the surrounding acreage could be described as "common ground" that was the mainstay of the logging industry in Mendocino county. Most of the Reserve is very steep, inaccessible to the public and sparsely covered by trees. It represents an example of the type of redwood stands that would have extensively covered this part of the coast redwood range. The fact that this site was spared the loggers ax makes it unique.

Montgomery Woods, has the expected assemblage of associated flora including tanoak *(Lithocarpus densiflora)*, big-leaf maple *(Acer macrophyllum)*, California bay or laurel (*Umbellularia californica*), red alder (*Alnus rubra*), sword fern *(Polystichum munitum)*, redwood sorrel *(Oxalis oregana)*, rhododendron *(Rhododendron macrophyllum)*, huckleberry *(Vaccinium ovatum)*, salal *(Gaultheria shallon)*, and azalea *(Rhododendron occidentele)*. Sholars (in Sawyer and others 2000) has cataloged 47 species of lichens as part of the Bryophytic component for Montgomery Woods.

The study site is adjacent to the Big River. The sites parallel to the river are relatively flat. Transects that are perpendicular to the river vary from 0° to 45° slope. The average slope across the study area is 31.5°.

Methods

Currently, the study site includes 14 parallel transects spaced at 35 m intervals perpendicular to the river. Plot spacing is 30 m, with an average of four plots/transect. The sampled area is 5.46 ha (13 acres). By design, each transect includes sites in the alluvial plain and the adjacent upland. A total of 46, permanent, 0.04 ha (1/10 acre), circular plots have been established. Each plot is surveyed for slope, tree species, tree size, and habitat elements. Habitat elements include snags, down wood, rock outcroppings, leaf litter, and ground cover.

Results

Stand Density Characteristics

Tree density is relatively low with relatively few; very large individuals occupying the stand. A total of 1,313 trees were tallied within the plots. Of these 13 percent are coast redwood, seven percent Douglas-fir and 80 percent tanoak. The dominance of tanoak, in the smaller size classes is obvious in *figure 1*. Nearly 80 percent of all stems under 25.4 cm (10″) dbh are tanoaks. Conversely, all of the stems >203 cm (80″) dbh are redwood with the dominant Douglas-fir cohorts found between 101 to 152 cm (40 to 60″) dbh.

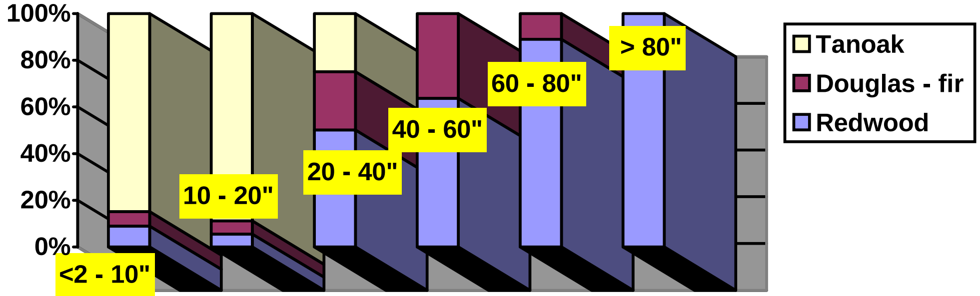


Figure 1—Size class distribution of old-growth coastal redwood stand (Montgomery Woods State Park, Mendocino County).

Trees diameters varied from <5.0cm to 254 cm (<2″ to >100″) dbh *(fig. 1)*. Tanoaks dominated the smaller sizes between 5 to 25.4cm (2 to 10”) with dominance inversely proportion to size class. The largest tanoak measured 91.4cm (36″) dbh. Douglas-fir was represented in the small and moderate sizes from <5.0 to 167.6cm (<2″ to 66″ dbh) and redwood was present in all sizes from <5.0 to 269.25cm (<2″ to 106″ dbh).

Size Class Distribution Characteristics

Redwood densities by size class varied from 62 trees/ha <25.4 cm (26 tpa <10″) to 2.4 trees/ha >203 cm (1 tpa >80″) *(table 1)*. Overall, redwood averaged 105 trees/ha (44 tpa) when all size classes are considered. DF density on the site, averaged 74 trees/ha (31 tpa) and varied between 43 trees/ha (18 tpa <10″) to 12 trees/ha (5 tpa) for the largest cohorts. Tanoak was both heavily dominant and limited in its size distribution being skewed toward the smaller size classes. Tanoak stems varied between 595 stems/ha (248 stems/acre) for trees <25.4 cm dbh (10″) to approximately 5 trees/ha (2 tpa) of those >51 cm (20″). Tanoak densities averaged 677/ha (282 tpa). The largest tanoak measured 91.44 cm (36 inches).

Table 1*—Distribution of trees size classes by species per acre, Montgomery Woods State Reserve.*

Species	<25.4 cm (10″) dbh	27.9-51 cm (11-20″) dbh	53-101 cm (21-40″) dbh	101-104cm (41-60″) dbh	154-203cm (61-80″) dbh	>203cm (80″) dbh	Total/ac
redwood	62 t/ha (26 tpa)	5 t/ha (2 tpa)	9 t/ha (4 tpa)	17 t/ha (7 tpa)	9 t/ha (4 tpa)	2.4t/ha (1 tpa)	105 t/ha (44 tpa)
Douglas-fir	43t/ha (18 tpa)	5t/ha (2 tpa)	5t/ha (2 tpa)	9t/ha (4 tpa)	12t/ha (5 tpa)	0	74t/ha (31 tpa)
tanoak	595t/ha (248 tpa)	77t/ha (32 tpa)	5t/ha (2 tpa)	0	0	0	677t/ha (282 tpa)

This data illustrates a relatively flat inverse “J” curve suggesting that tree recruitment is limited on the site *(fig. 2)*. Many of the smallest seedlings of all three species have very little live crown and are showing outward signs of senescence. Some of the redwood stems in the intermediate size classes (<51 cm (20″) appear to be the sprouts from the "fairy ring" from a nearby maternal tree unique to coast redwood regeneration.

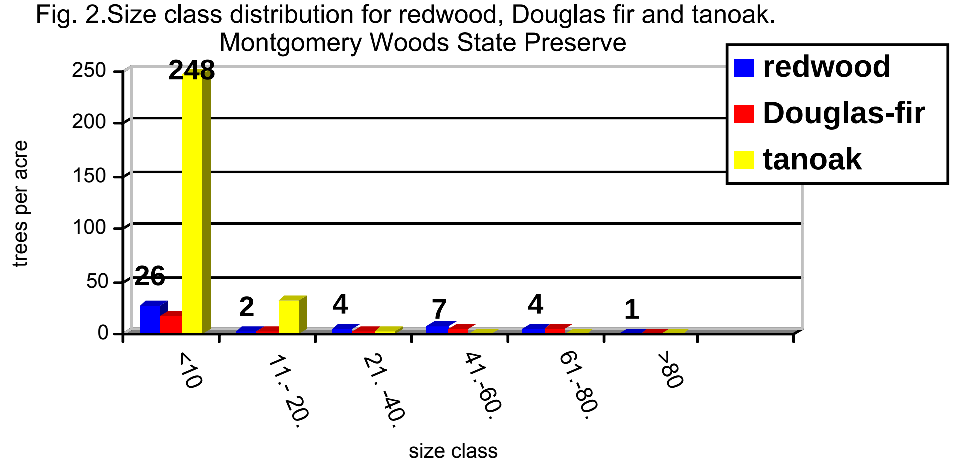


Figure 2—Size class distribution for redwood, Douglas fir and tanoak.

Snags

Snags, though present are relatively scarce. Because of the high density of tanoak stems, they represent the highest density of snags present (this site is not known to be infected with *Phytophthora ramorum*, the causative agent for Sudden Oak Death disease). Of tanoak snags, 18 percent of stems >25.4 cm (n = 115) exhibit characteristics commonly associated with age including: broken branches, tops, trunk hollows, fire scarring, and standing dead trees. Within this sample, 15 percent are true snags *(fig. 3)*, while three percent have trunk hollows potentially suitable for cavity use by several species. Snag size class distribution of tanoak is directly proportional to the ratio of stem size class availability with the most snags found in the smaller cohort class. Tanoak snag density of trees >25.4 cm (10″) = 21 trees/ha (9 tpa).

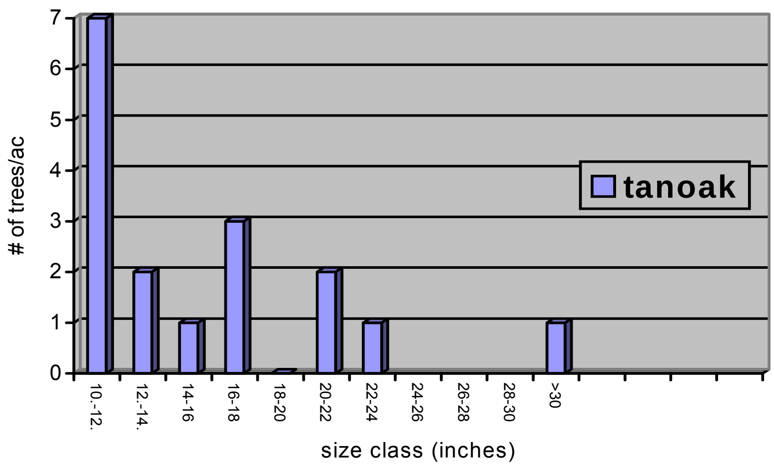


Figure 3—Size distribution of tanoak stems >10″ dbh having old forest characteristics, Montgomery Woods State Reserve.

Douglas-fir snag density is low. Of the 22 stems >51 cm (20″) only three snags were found. In all cases, each snag is >76cm (30″) dbh. This translates to a density of .5 snag/ha (.2/acre) suggesting a relatively few number of DF stems are readily accessible for snag habitat in older coastal redwood/Douglas-fir forests.

True snags of coast redwood were not observed within the transects. However, unique habitat characteristics often associated with standing snags are prevalent on living trees in the form of fire scars with trunk hollows. Virtually every large redwood on site has evidence of fire scarring visible on the trunk. Of those trees >76 cm (30″) dbh, 31 percent have trunk hollows of various sizes and shapes. Of those trees >101 cm (40″) nearly 29 percent have trunk hollows and 35 percent of those trees >152 cm (60″) have hollows. Coast redwood trees with trunk hollows are distributed at densities equaling .4/ha (.16 tpa) for all sizes.

Discussion

Forest Distribution and Characteristics

The coast redwood forest, being dispersed along a north-south axis, has characteristics affected by geographic location and localized climate and edaphic conditions (Sawyer and others 2000). Any comparison between redwood stands from differing geographic locations should be viewed as reference points and not as absolute factors. Furthermore, any numerical comparison between stands should also recognize that differences exist in structural attributes between trees growing on upland sites vs. trees growing on alluvial flats with alluvial deposits generally having higher tree densities then those stands on adjacent upland sites. *Table 2* compares stand characteristics of old growth coast redwood stands from the northern tier of its range illustrating the variation in the structural elements of any particular site.

Table 2—*Stand density comparison between Old Growth coastal redwood stands from different geographic locations with differing climatic and edaphic conditions.*

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

Structural Characteristics of Older Forests in the Northern Tier of the Redwood Range

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 were collected for a number of properties in old growth redwood stands that were being proposed for acquisition (HJW 1969a, 1969b, 1969c, 1969d). The information provides an opportunity to compare size class distributions on un-logged sites from another portion of the northern tier of the redwood region with that obtained at Montgomery Woods.

The resultant data from the approximately 9,000 aggregated acres cruised demonstrates a relatively similar size class distribution of conifers as witnessed in Montgomery Woods, though the northern sites had some larger cohorts. Unfortunately, hardwoods are not part of the data set. Given the acreages of the cruised parcels, one can surmise that both upland and riparian sites were sampled. The average conifer density of the three ownerships for coast redwood was 32 trees/acre, with tree sizes below 50″ dbh dominating in each case, trees >50″ dbh were common and averaged >2 trees/acre. Similarly, each property averaged approximately one tree >90″ dbh per acre. A comparison of the three property cruise reports demonstrates a similarity of size class distribution across each of the ownerships *(fig. 5)*.

Comparison between this data and the information obtained from Montgomery Woods *(table 2)* on relative redwood densities suggests that the sites are different. This is consistent with Sawyer (2000, see chapter 3) who suggests that within the central portion of the coast redwood range steep, friable Franciscan formation soils affect both density and composition of the redwood site being evaluated. This characterization is accurate for Montgomery Woods.

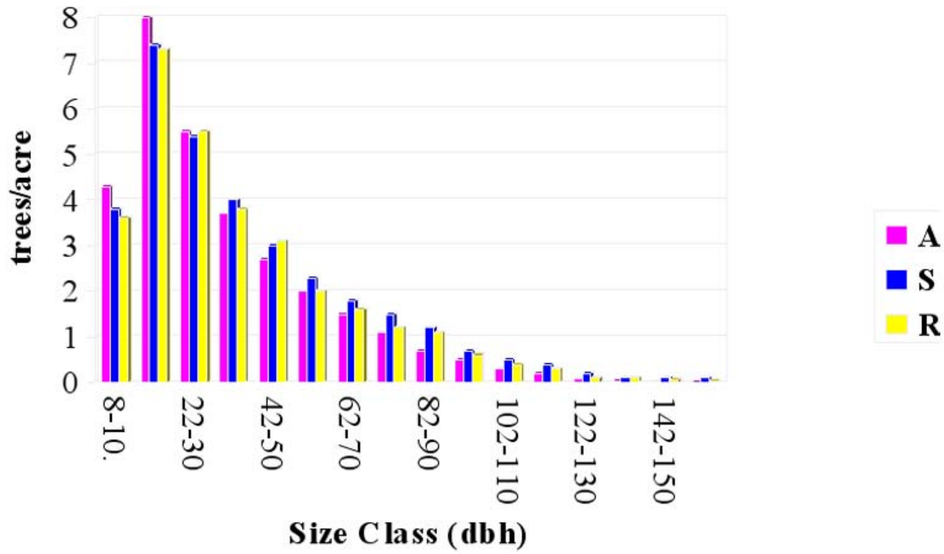


Figure 5—Size distribution of OG redwood trees ARCO, Simpson and Rellim Timber Companies. Del Norte County. 9,378 acres ≈ 32 trees/acre. (HJW 1969)

Habitat Relationships

The faunal component of the redwood type is not well studied (Cooperrider and others 2000) and those few examples of redwood–habitat interactions are exhaustedly cited (Gellman and Zielinski 1996, Zielinski and Gellman 1999) since so few works exist to guide decision-making.

The alteration of vast acreages of redwood forest, the result of commercial logging, makes it difficult to determine the extent and importance of many of the relationships between various habitat elements and dependent species. The Montgomery Reserve site suggests that the formation of a "true snag" from a redwood may be a rare occurrence with other species providing the necessary elements for snag dependent behaviors. However, those snag-like elements, for example, trunk hollows, found in older stands, are not easily recruited through conventional forestry management practices thereby necessitating the need to maintain those elements when they are found in a managed stand. A need exists to explore managerial alternatives to evaluate strategies to recruit unique habitat elements into redwood stands that have been simplified by past logging.

References

Gellman, S.T.; Zielinski. W.J. 1996. **Use by bats of old growth redwood hollows on the north coast of California**. J. Mammal 77(1): 255-265.

Hammon, Jensen, & Wallen (HJW). 1969a. **Timber inventory report. Proposed Rellim Redwood Company allocation of the northern purchase unit lands. Del Norte county**. A report to the US Dept. of Interior, Nov. 1, 1969. Humboldt State Univ. Humboldt Room, Ref. Library. Arcata, CA.

Hammon, Jensen, & Wallen (HJW). 1969b. **Timber inventory report. Proposed Simpson Timber Company allocation of the northern purchase unit lands. Del Norte county**. A report to the US Dept. of Interior, Nov. 1, 1969. Humboldt State Univ. Humboldt Room, Ref. Library. Arcata, CA.

Hammon, Jensen, & Wallen (HJW). 1969c. **Timber inventory report. Proposed Arcata Redwood Company allocation of the northern purchase unit lands. Del Norte county**. A report to the US Dept. of Interior, Nov. 1, 1969. Humboldt State Univ. Humboldt Room, Ref. Library. Arcata, CA.

Hammon, Jensen, & Wallen (HJW). 1969d. **Timber inventory report. Proposed Georgia-Pacific Timber Company allocation of the northern purchase unit lands**. Humboldt county. A report to the US Dept. of Interior, Nov. 1, 1969. Humboldt State Univ. Humboldt Room, Ref. Library. Arcata, CA.

Mladenoff, D.J.; White, M.A.; Pastor, J.; Crow, T.R. 1993. **Comparing spatial patterns in unaltered old growth and disturbed forest landscapes**. Ecological Applications 3(2): 294-306.

O'Dell, T.E. 1996. **Silviculture in the redwood region: a historical perspectiv**e. In: LeBlanc, J., ed. Proc. Coast Redwood Forest Ecology and Management. 18-20 June 1996. Humboldt State Univ., Arcata, CA.; 15-17.

Sawyer, J.O.; Sillett, S.C.; Popenoe, J.H.; LaBanca, A.; Sholars, T.; Largent, D.L.; Euphrat, F.; Noss, R.; Van Pelt, R. 2000. **Characteristics of redwood forests**. In: Noss, R., ed. The Redwood Forest: History, Ecology and Conservation of the Coast Redwood. Washington, D.C.: Island Press; 39-79.

Spies, T. 2003. **New findings about old growth forests**. Science Findings. No. 4. June 2003. Portland, OR. USDA Forest Service PNW Res. Sta. 11 pp.

Zielinski, W.J.; Gellman, S.T. 1999. **Bat use of remnant old-growth redwood stands.** Conservation Biology 13: 160-167.

The Conservation of Sensitive Plants on Private Redwood Timberlands in Northern California[1]

Clare Golec,[2] Tony LaBanca,[3] and Gordon Leppig[3]

Abstract

Redwood forests are not traditionally known for a diverse vascular plant flora or a large number of sensitive plant taxa. Approximately 90 percent of redwood forests are privately owned and most are managed for timber production. Historically, information on sensitive plants in the redwoods has been scarce because few sensitive plant surveys were conducted on public lands, botanists have typically not had access to private lands, and no botanical survey data was requested in the permitting process for projects on managed timberlands. Prior to 1999, most of the information available on sensitive plants in redwood forests was based upon early twentieth century herbarium specimens. In the late 1990s, state agencies began requesting sensitive plant survey data for projects conducted in managed redwood stands. Five years of plant surveys have resulted in an improved understanding of the ecology and distribution of many sensitive plant taxa. For taxa whose distribution and ecology have become better understood, landscape-level approaches to conservation are being developed. This paper presents an overview of the sensitive plant flora and management strategies used to conserve sensitive plants on privately-owned redwood forests.

Key words: managed timberlands, redwood flora, sensitive plant conservation

Introduction

Redwood forests *(Sequoia sempervirens)* are the dominant forest type of the Northern California Coastal Forest Ecoregion. These forests are considered “globally outstanding” for their biological distinctiveness and have “critical” conservation status because intact habitat is restricted to isolated small fragments and land use of the remaining habitat is often incompatible with maintaining most native species and communities (Ricketts and others 1999). Less than five percent of all redwood stands are un-entered old-growth (Noss 2000) with the majority of the remaining 95 percent managed for short rotation commercial timber production. Humboldt and Del Norte counties, for instance, have over 648,000 acres (262,000 hectares) of commercial redwood timberlands, compared with 57,000 acres (23,000 hectares) of old-growth forest in federal and state parks (Fox 1989).

California is famous for its floristic diversity and high percentage of endemic plants. Northwestern California, which includes the redwood region, is especially

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Environmental Scientist/Botanist, California Department of Fish & Game, 1031 S. Main Street, Fort Bragg, CA 95437. email: cgolec@dfg.ca.gov

[3] Environmental Scientist/Botanist, California Department of Fish & Game, 619 Second Street, Eureka, CA 95501. email: tlabanca@dfg.ca.gov and gleppig@dfg.ca.gov

renowned for its plant diversity and sensitive species (DellaSala and others 1999, Smith and Sawyer 1988). Yet despite the dominance and biological distinctiveness of redwood forests in the region, there is no landscape-level account of their sensitive plant flora.

Historically, information on sensitive plants in redwood forests on commercial timberlands has been scarce because few floristic plant surveys were conducted and botanists have typically not had access. Prior to 1999, most of what little information there was on sensitive plants was based upon early Twentieth Century herbarium specimens and limited occurrence information in databases such as California Natural Diversity Database (CNDDB) and California Native Plant Society's (CNPS) Inventory of Rare and Endangered Plants of California. No studies have assessed the abundance, diversity, ecology, and management impacts to sensitive plants in redwood timberlands. Stuart (1996) identifies one plant, Santa Cruz Cypress *(Cupressus abramsiana)* on a list of "threatened and endangered species found in redwood forests." Sawyer and others (2000) presents a regional assessment of the rare plants of Mendocino, Humboldt, and Del Norte County redwood forests, however all habitats of the region are considered, including taxa restricted to coastal sand dunes and montane serpentine habitats.

With so little information on sensitive plants in the redwoods, assessing status and distribution, analyzing potential threats from timber operations, and developing appropriate mitigation and conservation strategies is difficult. An understanding of the sensitive plants occurring in redwood timberlands is especially important given the large percentage of redwood stands in timber production and because timber operations can have significant adverse impacts on sensitive species. Schemske and others (1994) for instance, attributed logging as the primary cause of endangerment for over seven percent of plants listed as threatened or endangered by the U.S. Fish and Wildlife Service. Equally as important is information on life history and population demographics of sensitive taxa.

Beginning in the late 1990s, the California Department of Forestry and Fire Protection (CDF) and the California Department of Fish and Game (DFG), in conjunction with other stakeholders, began evaluating the presence of, and impacts to, sensitive plants during the preparation of Timber Harvesting Plans (THPs). Under the THP preparation and review process, land managers began conducting surveys for sensitive plants in potential habitats, disclosing occurrences to CDF and DFG, and developing mitigation measures to avoid or minimize impacts to these occurrences. This survey, disclosure, and mitigation process was conducted pursuant to the California Forest Practice Rules, Fish and Game Code Title 14 California Code of Regulations, and the California Environmental Quality Act (CEQA). DFG has three key objectives for conserving sensitive plants (Morey and Ikeda 2001):

- Maintenance of viable populations within vegetation and habitat types representing the range of conditions throughout which a species occurs.
- Protection of habitat and populations.
- Maintenance of genetic variability and integrity.

For a number of species, when a sensitive plant population is found in a THP area, DFG and other stakeholders have been monitoring these populations for response to timber management activities. Many of these THP-based monitoring programs involve the collection of simple census data—counting the number of plants in a site before operations and recounting the individuals at certain intervals after management activities. Other monitoring activities have a landscape-level focus that attempt to assess what the overall trend of a species is across an ownership or the region.

The sensitive plant surveys conducted for THPs in the past five years have surveyed less than 10 percent of the study area, yet this effort has resulted in the discovery of hundreds of rare plant occurrences. Based upon survey, mitigation, and monitoring data developed by DFG and other stakeholders since 2000, there now exists a much greater understanding of the distribution and composition of the redwood sensitive plant flora.

Method

The study area is redwood forests in Del Norte (DNT), Humboldt (HUM), and Mendocino (MEN) counties *(fig. 1)*. This area was chosen because the majority of commercial timber harvesting operations, as well as corresponding botanical survey effort, documented sensitive plant occurrences, and monitoring, are found within those counties.

Within the study area, redwood forests are defined as described in the redwood series by Sawyer and Keeler-Wolf (1995). In addition, associated vegetation types found wholly within or commonly interfacing with redwood forest are also incorporated. This includes coastal prairie, coastal scrub, chaparral, oak and pine woodland, grasslands, wetlands, and mixed evergreen forest.

We define sensitive plants as all vascular plants listed as endangered, threatened or rare[4], or those meeting the definitions of rare or endangered provided in Section 15380 of the CEQA Guidelines. Taxonomic nomenclature follows Hickman (1993). The documented sensitive plants of redwood forests in Del Norte, Humboldt, and Mendocino counties (hereafter northern redwood forest) were determined by the following means:

1. Analysis of the sensitive plant occurrence data in CNDDB (CNDDB 2005) and the Inventory of Rare and Endangered Plants of California (CNPS 2005);

2. Review of location and habitat data from herbarium specimens at Humboldt State University (HSC), California Academy of Sciences (CAS), University of California, Berkeley (UC), and the Jepson Herbarium (JEPS);

3. Assessment of the habitat and location information provided in the literature and regional floras (Hickman 1993, Hitchcock and others 1955-1969, Mason 1957, Munz and Keck 1970, Smith and Wheeler 1992); and

4. Review of the unpublished data such as DFG sensitive plant consultations, sensitive plant survey reports, and reports from timberland owners.

[4] Pursuant to Section 670.2, Title 14, California Code of Regulations; Section 1900, Fish and Game Code; Endangered Species Act, Section 17.11, Title 50, Code of Federal Regulations

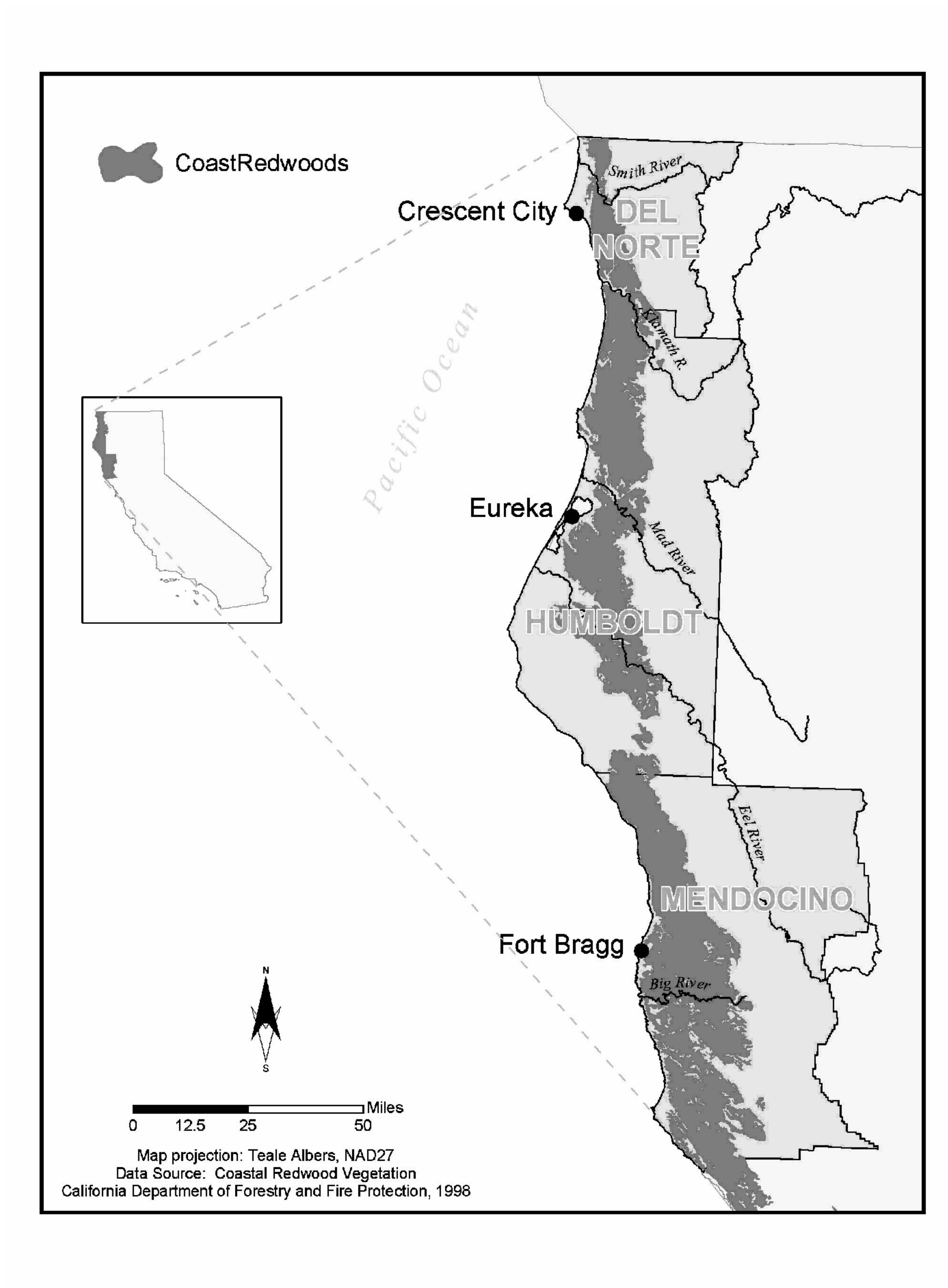


Figure 1—The northern redwood region study area.

Results

Forty-five sensitive plant species are documented as occurring in or closely associated with redwood forests of northwestern California *(appendix 1)*. Representation within major groups includes one fern, one club-moss, one gymnosperm, 17 monocotyledons, and 25 dicotyledons. The majority of the taxa are perennial herbs. There are four annuals, one shrub, and one tree. Families with the most taxa include Cyperaceae (eight taxa), Malvaceae (five), Asteraceae (five), and Poaceae (four). *Carex* is the largest genus with seven taxa.

The number and percentage of the sensitive plants of the redwood region[5] are presented in *table 1*. Of the 123 sensitive plants within the northern redwood region 37 percent are found in redwood forests. There are no documented federally listed species within the study area; however, four species are state listed. Forty-two percent of the sensitive taxa in northern redwood forests are CNPS Listed 1A or 1B (CNPS 2005) and 24 percent are considered to be globally restricted (G-rank 1 and 2).

Table 1—*Number and percentage of sensitive plants in the redwood region.*

Region/status	Number of taxa	Percentage
Redwood Region* (USGS quadrangle query)	**301**	**20 percent** (of 1,518 sensitive taxa in CA)
Northern Redwood Region* (USGS quadrangle query for DNT, HUM, MEN cos.)	**123**	**73 percent** (of 168 sensitive taxa in DNT, HUM, MEN cos.)
Redwood Forest* (in the Northern Redwood Region)	**45**	**3 percent** (of 1,518 sensitive taxa in CA) **27 percent** (of 168 sensitive taxa in DNT, HUM, MEN cos.) **37 percent** (of 123 sensitive taxa in northern redwood region)
Federal Listed Endangered and Threatened/Redwood Forest	**0**	N/A
State Listed Endangered, Threatened and Rare/Redwood Forest	**4**	**9 percent** (of 45 sensitive taxa in redwood forest)
State Rank S1/Redwood Forest	**14**	**31 percent** (of 45 sensitive taxa in redwood forest)
Global Rank G1 and G2/Redwood Forest	**11**	**24 percent** (of 45 sensitive taxa in redwood forest)
CNPS 1A and 1B/Redwood Forest	**18**	**40 percent** (of 45 sensitive taxa in redwood forest)

* See text for complete description. See Appendix 1 for State/Global rank and CNPS listing definitions.

A habitat analysis of the 45 sensitive taxa shows finer scale affinities to the redwood forest and associated adjacent vegetation *(table 2)*. Eighteen taxa (40 percent) occur in wetlands in California with indicator status of “facultative wetland” (estimated probability 67 percent to 99 percent), or “obligate” (estimated probability

[5] The redwood region is defined as all U. S. Geological Service 7.5 minute topographic quadrangles that overlap the coast redwood range as depicted by Fox (1989).

>99 percent) (Reed 1996). Nine species (20 percent) are forest generalists which can occur in various micro-habitats. The remaining 18 taxa reside in habitats that occur within or interface with redwood forests such as grassland, shrubland, and rock outcrops.

Table 2—*Number and percentage of sensitive plants in the redwood forest by general habitat.*

Generalized habitat	Number of taxa	Percentage
Wetland	18	40 percent
Forest/Grassland Interface	10	22 percent
Forest Generalists	9	20 percent
Forest/Shrubland Interface	4	9 percent
Forest/Rock Outcrops (mesic & xeric)	4	9 percent

The geographic distribution of sensitive plants occurring in redwoods is diverse, with species ranging from narrowly restricted to sub-cosmopolitan. Seventeen taxa (38 percent) are endemic to the California Floristic Province as described by Raven and Axelrod (1978), which includes southwestern Oregon. Pygmy cypress *(Cupressus goveniana ssp. pigmaea)*, North Coast *semaphore (Pleuropogon hooverianus)*, and the swamp harebell *(Campanula californica)* are endemic taxa restricted to the southern portion of this study area.

Sixteen taxa (36 percent) have distributions restricted to western North America, defined here as the area west of and including the Rocky Mountains and north to Alaska. Many of these taxa, such as the hair-leaved rush *(Juncus supiniformis)*, small groundcone *(Boschniakia hookeri)*, and Howell's montia *(Montia howellii)* have coastal Pacific Northwest distributions that extend from Alaska or British Columbia in the north to a southern terminus in northern California.

Eleven taxa (24 percent) are widespread, occurring either across North America, or in the case of running-pine *(Lycopodium clavatum)*, are sub-cosmopolitan. Many widespread taxa have regional distributions that follow the Pacific Northwest coast, with their southern terminus in northern California. Widespread species with California distributions restricted to the redwood region include livid *sedge (Carex livida)*, and Indian-pipe *(Monotropa uniflora).*

In summary, the redwood forest sensitive plant flora is comprised of a diverse mixture of locally restricted endemics, Pacific Northwestern taxa, and plants with wide global distributions. While some taxa are endemic to the redwood region, no sensitive species, indeed no plant species with the exception of redwoods, are endemic to redwood forests.

Discussion

The specific reasons for the rarity of many sensitive taxa of the redwood forest are unclear and little information exists on the full range of their "preferred" habitat attributes. How timber management activities affect sensitive plants is not fully understood. However, for most of these taxa the reasons for their rarity is likely a combination of being; 1) naturally scarce on the landscape; and 2) populations and habitats are adversely effected or eliminated by land management activities (Fiedler 2001, Fiedler and Ahouse 1992). Landscape alterations such as changes in average stand age and composition, soil structure, chemical and microbial ecology, pollinator communities, microclimate, and fire ecology, as well as the introduction of invasive non-native species are known to adversely affect sensitive plant species and ultimately their persistence on the landscape (Halpern and Spies 1995, Harper 1981, Schemske and others 1994).

DFG's general conservation approach is, as more data are available on a species' ecology and response to management activities, the greater the array of options available for managing the species. For instance, if a species is extremely rare and little is known about its response to timber management, the principle mitigation strategy will be avoidance *(fig. 2)*. If a species is more widespread, abundant locally, and data clearly indicate a positive response to certain management activities, then a broader range and intensity of impacts could be employed.

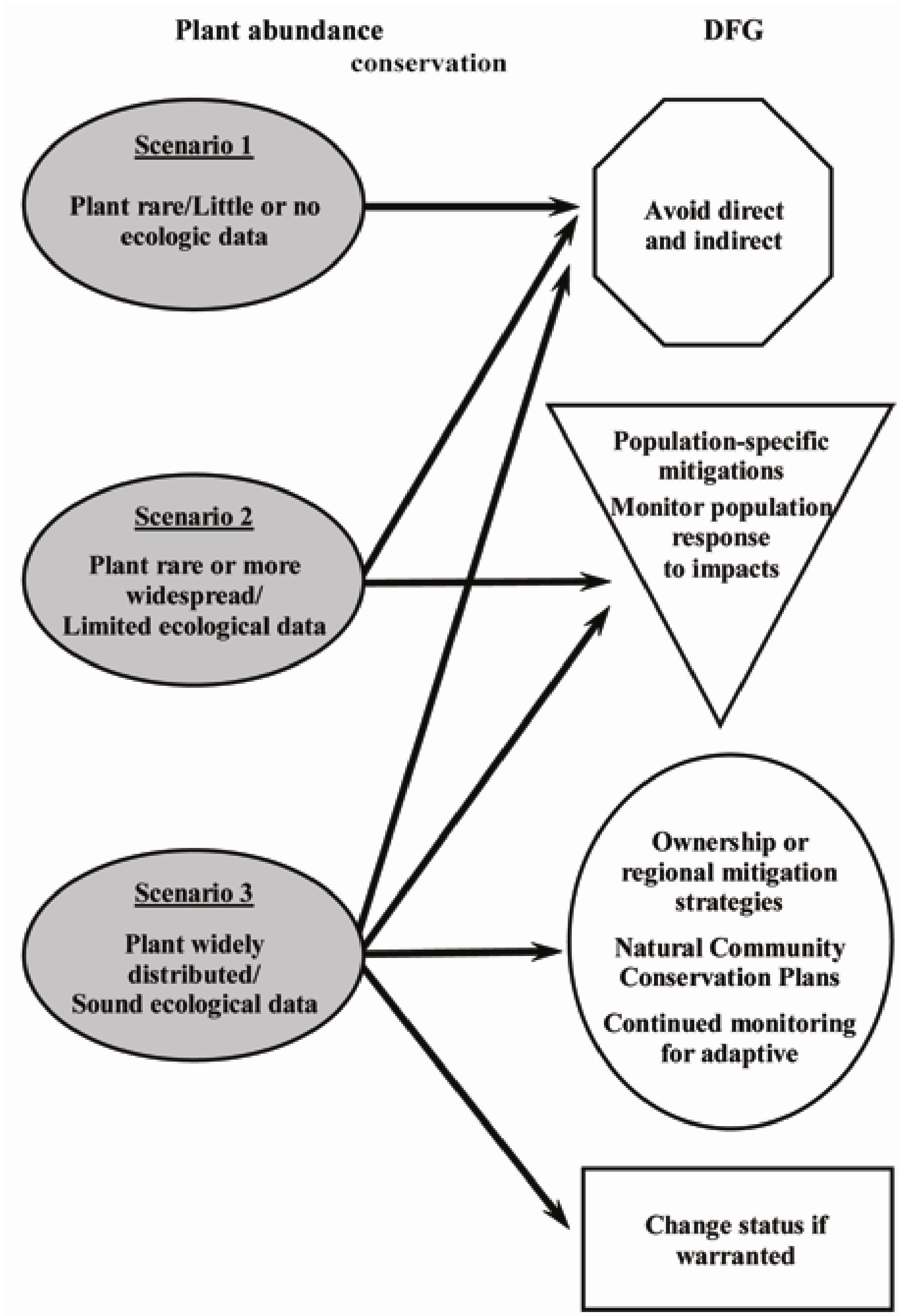


Figure 2—Sensitive plant conservation strategy model used by DFG on timberlands within the northern redwood region.

It is important to recognize that not all timber management activities are harmful to all sensitive plants, and indeed, some timber management activities can be beneficial to certain species. However, direct impacts that result in mortality are not beneficial to the species. Clearly, the sensitive plant flora now present on managed redwood timberlands is, in part, a result of over 100 years of intensive timber management. Certain sensitive plants appear to persist in managed timberlands because timber management activities, such as canopy removal, ground disturbance, and site preparation, can in some ways emulate natural disturbance and create habitat for certain taxa.

However it is critical to understand that almost no data exist on the long-term regional trends for the majority of sensitive species occurring in redwood forests, whether in managed timberlands or old growth stands. While some taxa may have always been uncommon, other taxa are sensitive to disturbance and may have been much more common prior to timber management practices. These taxa may be regionally declining in range or abundance, but we simply lack the data to detect it. Therefore, the commonly held view that if a species persists in managed timber lands, then timber management activities must not be harmful, or are even beneficial, is not generally substantiated by data. Extant occurrences may be remnants of a species' continuing regional demise.

Range contraction and the loss of historic occurrences in redwood forests are documented for some taxa. The livid sedge has not been seen in California since 1866 (CNDDB 2005) and is presumed extirpated. CNDDB data, combined with recent reconnaissance work, indicate that the coast fawn lily may have been more common in redwood forests than at present and may not persist in a managed redwood landscape. The disjunct and southern-most occurrence of Indian pipe in California appears to have been extirpated, though it is unclear if forest management or urbanization contributed to the presumed disappearance of this occurrence (LaBanca 2005).

The presence of a sensitive plant occurrence in a managed redwood stand is by no means substantial evidence to presume the continued existence of the occurrence after proposed timber management activities. Extrapolating a sensitive plants' response to proposed timber management activities is complex and not easily predicted based on previous disturbance history. We cannot solely use the current presence of a population as an inference that it has responded positively to past management because; 1) the occurrence may have not have been present during past operations, 2) it may not have been directly or fully impacted by past operations, and 3) there is typically little specific data on what was the previous management regime. Also, it is often difficult to construct an accurate assessment of the trends of a given population. For instance, a sensitive plant population that currently has 500 individuals may have had 50,000 individuals prior to the last stand entry, or it may not have been present at all.

Conclusion

A lack of solid distribution, life history, and population-response data have resulted in DFG being cautious in applying untested assumptions about how specific timber management may affect a given species. The lack of data highlights the need for land managers to report sensitive plant occurrences to CNDDB and the greater need, and greater benefit to land managers, of meaningful monitoring and ecological data that elucidates how best to manage for a given species.

For species that have a sufficient number of occurrences, strong inferences can be made about habitat characteristics and a plants' association with certain forest stand and land-use histories. For species with a larger number of occurrences and with useful data on response to environmental changes, DFG has worked with large industrial timberland owners to develop property-wide mitigation strategies that can be employed when occurrences are detected. As more information becomes available, DFG will continue to develop regional and state-wide management plans for sensitive taxa.

A landscape-level conservation strategy that maintains viable sensitive plant populations in habitat types representing the range of conditions throughout which a species occurs is the ultimate goal of the DFG sensitive plant program. In addition, the landscape-level strategy is far more comprehensive and flexible then a project by project approach. With the help of timberland owners and other stakeholders, conservation of sensitive plants in managed redwood stands will become more effective and easier to implement. Ultimately, a landscape-level conservation strategy based upon sound science will best protect this unique and diverse component of the region's flora.

References

Abrams, L.R.; Ferris, R. 1923-1960. **An Illustrated Flora of the Pacific States, Washington, Oregon, and California.** Stanford, CA: Stanford University Press; 4 volumes.

California Department of Fish and Game. 2000. **Guidelines for Assessing the Effects of Proposed Projects on Rare, Threatened, and Endangered Plants and Natural Communities.** Sacramento, CA. *Available at* http://www.dfg.ca.gov/whdab/pdfs/guideplt.pdf

California Native Plant Society. 2005. **Inventory of Rare and Endangered Plants** (online edition, v6-05d). California Native Plant Society. Sacramento, CA. *Available at* http://www.cnps.org/inventory

California Natural Diversity Database. 2005. **California Department of Fish and Game Natural Diversity Database**, July 2005. Special plants list.

DellaSala, D.A.; Reid, S.T.; Frest, T.J.; Strittholt, J.R.; Olson, D.M. 1999. **A global perspective on the biodiversity of the Klamath-Siskiyou ecoregion**. Natural Areas Journal 19(4):300-319.

Fiedler, P.L. 2001. **Rarity in vascular plants**. In: California Native Plant Society's Inventory of Rare and Endangered Plants of California. Special Publication Number 1, Sixth Edition 2001. Sacramento, CA: California Native Plant Society; 2-4.

Fielder, P.L.; Ahouse, J. 1992. **Hierarchies of cause: Toward understanding of rarity in vascular plant species**. In: Conservation Biology. Fiedler, P.L.; Jain, S.K. editors. New York, New York: Chapman and Hall; 23-47.

Fox, L.J. 1989. **A Classification, Map, and Volume Estimate for the Coast Redwood in California**. Final report to the Forest and Rangeland Resources Assessment Program. California Department of Forestry and Fire Protection. Sacramento, California.

Halpern, C.B.; Spies. T.A. 1995. **Plant Species Diversity in Natural and Managed Forests of the Pacific Northwest**. Ecological Applications 5(4):913-934.

Hickman, J.C. (editor) 1993**. The Jepson Manual, Higher Plants of California**. Berkeley, CA: University of California Press; 1440 p.

Hitchcock, C.L.; Cronquist, A.; Ownbey, M.; Thompson, J.W. 1955-1969. **Vascular plants of the Pacific Northwest**. Seattle, WA: University of Washington Press; 5 volumes.

LaBanca, T. 2005. ***Monotropa uniflora* in the Coastal Forests of Northwestern California**. In: The Ecology and Management of Rare Plants of Northwestern California; Proceedings from a North Coast Chapter California Native Plant Society Symposium, Sacramento, CA; CNPS Press; 61-69.

Mason, H.L. 1957. **A flora of the marshes of California**. Berkeley, CA: University of California Press; 878 p.

Morey, S.; Ikeda, D. 2001. **Conserving plants with laws and programs under the Department of Fish and Game**. In: California Native Plant Society's Inventory of Rare and Endangered Plants of California. Special Publication Number 1, Sixth Edition 2001. Sacramento, CA: California Native Plant Society; 12-16.

Noss, R.F. 2000. **More than just big trees**. In: Noss, R. F., ed. The Redwood Forest: History, Ecology, and Conservation of the Coast Redwoods. Washington D.C.: Save-the-Redwoods League, Island Press; 1-6.

Raven, P.H.; Axelrod, D.I. 1978. **Origin and relationships of the California flora**. University of California Publications in Botany. Vol. 72. Berkeley, CA: University of California Press; 134 p.

Reed, P.B. Jr. 1988. **National List of Plant Species that Occur in Wetlands: California (Region 0)**. Fish & Wildlife Service Biological Report 88(26.10).

Ricketts, T.H.; Dinerstein, E. ;Olson D.M.; Loucks, C.J.; Eichbaum, W.; DellaSala, D.; Kavanagh, K.; Hedao, P.; Hurley, P.T.; Carney, K.M.; Abell, R.; Walters, S. 1999. **Terrestrial ecoregions of North America: a conservation assessment**. Washington, D. C.: Island Press; 485 p.

Sawyer, J.O., Sillett, S.C.; Popenoe, J.H.; LaBanca, A.; Sholars, T.; Largent, D.L.; Euphrat, F.; Noss, R.F.; Van Pelt, R. 2000. **Characteristics of Redwood Forests**. In: The Redwood Forest, Noss, R.F. ed.; Washington D.C.: Save-the-Redwoods League, Island Press l; 39-80.

Schemske, D.W., Husband, B.C.; Ruchelshaus, M.H.; Goodwillie, C.; Parker, I.M.; Bishop, J.G. 1994. **Evaluating approaches to the conservation of rare and endangered plants.** Ecology 75:584-606.

Smith, G.L.; Wheeler, C.R. 1992. **A flora of the vascular plants of Mendocino County, California**. San Francisco, CA.: University of San Francisco.; 387 p.

Smith, J.P.; Sawyer, J.O. 1988. **Endemic vascular plants of northwestern California and southwestern Oregon**. Madrono 35:54-69.

Stuart, J.D. 1996. **Coast Redwood Forest Health**. In: LeBlanc, J., technical editor. Proceedings of the Conference on Coast Redwood Forest Ecology and Management. Berkeley, CA: University of California Cooperative Extension; 21-25.

Appendix 1—*Forty-five sensitive plant species are documented as occurring in or closely associated with redwood forests of northwestern California.*

Scientific name Common name	State status	State rank	Global rank	Federal status	CNPS status	General redwood forest habitat	Distribution	Geographic regions (Hickman 1993 and others)
Asplenium trichomanes ssp. *trichomanes* maidenhair spleenwort	None	S2.3	G5T5	None	2	Forest/Rocky Mesic	WS	northwestern Klamath Ranges in Del Norte County; widespread in North America, Europe, and Asia, and southern temperate areas
Astragalus agnicidus Humboldt milk-vetch	SE	S1.1	G1	None	1B	Forest Generalist	CA	Outer North Coast Ranges in southern Humboldt County and northern Mendocino County
Boschniakia hookeri small groundcone	None	S1S2	G5	None	2	Forest Generalist	WNA	northern San Francisco Bay Area, the Outer North Coast Ranges, to British Columbia
Calamagrostis crassiglumis Thurber's reed grass	None	S1.2	G3	None	2	Wetland/Obligate	WNA	Central Coast, North Coast, Washington, widespread outside of California
Calamagrostis foliosa leafy reed grass	SR	S3.2	G3	None	4	Forest/Rocky Xeric	CA	North Coast, and the Outer North Coast Ranges with many occurrences in the King Range in Humboldt County
Calystegia purpurata ssp. *saxicola* coastal bluff morning-glory	None	S2.2	G4T2	None	1B	Forest/Shrub Interface	CA	southern and central North Coast and the northern San Francisco Bay Area
Campanula californica swamp harebell	None	S2.2	G2	None	1B	Wetland/Obligate	CA	northern Central Coast and the southern North Coast
Carex arcta northern clustered sedge	None	S1S2	G5	None	2	Wetland/Obligate	WS	southern Outer North Coast Ranges, Oregon, Washington, widespread outside of California
Carex californica California sedge	None	S2?	G5	None	2	Forest/Grassland Interface	WNA	North Coast to WA; also in ID
Carex leptalea flaccid sedge	None	S2?	G5	None	2	Wetland/Obligate	WS	Central Coast, the North Coast and the Outer North Coast Ranges north to Alaska and east to Atlantic
Carex livida livid sedge	None	SH	G5	None	1A	Wetland Obligate	WS	North Coast (Mendocino Co.), scattered to AK, eastern United States
Carex praticola meadow sedge	None	S2S3	G5	None	2	Forest/Grassland Interface	WS	North Coast in Humboldt County, southern and central High Sierra Nevada, and North America
Carex saliniformis deceiving sedge	None	S2.2	G2	None	1B	Wetland/Facultative	CA	Central and North Coast

Scientific name Common name	State status	State rank	Global rank	Federal status	CNPS status	General redwood forest habitat	Distribution	Geographic regions (Hickman 1993 and others)
Carex viridula var. *viridula* green sedge	None	S1.3	G5T?	None	2	Wetland/Obligate	WS	North Coast, northern High Sierra Nevada (1 site), to Alaska, eastern North America, and also Japan
Cupressus goveniana ssp. *pigmaea* pygmy cypress	None	S2.2	G2T2	None	1B	Closed Cone Forest Interface	CA	central and southern North Coast
Erigeron biolettii streamside daisy	None	S3?	G3	None	3	Forest/Rocky Mesic	CA	Outer North Coast Ranges
Erigeron supplex supple daisy	None	S1.1	G1	None	1B	Forest/Grassland Interface	CA	northern and central North Coast
Erythronium revolutum coast fawn lily	None	S2.2	G4	None	2	Forest Generalist	WNA	North Coast and the Outer North Coast Ranges to British Columbia
Fritillaria roderickii Roderick's fritillary	SE	S1.1	G1Q	None	1B	Forest/Grassland Interface	CA	central North Coast and the North Coast Ranges
Gilia capitata ssp. *pacifica* Pacific gilia	None	?	G5T?	None	1B	Forest/Grassland Interface	WNA	northern and central North Coast, extending into Oregon
Glyceria grandis American manna grass	None	S1.3?	G5	None	2	Wetland/Obligate	WS	North Coast and the North Coast Ranges, extending to British Columbia, the eastern United States
Hemizonia congesta ssp. *leucocephala* Hayfield tarplant	None	S2S3	G5T2T3	None	3	Forest/Grassland Interface	CA	North Coast Ranges and the San Francisco Bay Area
Horkelia tenuiloba thin-lobed horkelia	None	S2.2	G2	None	1B	Forest/Shrubland Interface	CA	central and southern North Coast, the central and southern Outer North Coast Ranges, and the northwestern San Francisco Bay Area
Juncus supiniformis hair-leaved rush	None	S2.2?	G4?	None	2	Wetland/Obligate	WNA	northern and central North Coast, to Alaska
Lilium maritimum coast lily	None	S2.1	G2	None	1B	Wetland/Facultative	CA	southern North Coast, extirpated in the northern Central Coast
Lycopodium clavatum running-pine	None	S2S3	G5?	None	2	Forest Generalist	WS	North Coast, extending to Alaska, Montana, eastern North America, the Caribbean, South America, Eurasia, Africa, and Asia

Scientific name Common name	State status	State rank	Global rank	Federal status	CNPS status	General redwood forest habitat	Distribution	Geographic regions (Hickman 1993 and others)
Microseris borealis northern microseris	None	S1.1	G4?	None	2	WetlandObligate	WNA	Outer North Coast Ranges (Bald Mtn., Humboldt Co.) to Alaska
Mitella caulescens leafy-stemmed mitrewort	None	S2.3	G5	None	2	Forest Generalist	WNA	North Coast, Klamath Ranges to British Columbia, Montana
Monardella villosa ssp. *globosa* robust monardella	None	SH	G5TH	None	1B	Forest/Grassland Interface	CA	San Francisco Bay Area and the Outer North Coast Ranges
Monotropa uniflora Indian-pipe	None	S2S3	G5	None	2	Forest Generalist	WS	North Coast and Klamath Ranges, extending to British Columbia, eastern North America, Central America, northern South America, and eastern Asia
Montia howellii Howell's montia	None	S1.2	G3	None	2	Wetland/Facultative	WNA	Northwest California to British Columbia
Packera bolanderi var. *bolanderi* seacoast ragwort	None	S2.2	G4T4	None	2	Forest Generalist	WNA	North Coast, extending to Washington
Pleuropogon hooverianus North Coast semaphore grass	ST	S1.1	G1	None	1B	Wetland/Facultative	CA	southern North Coast and the northern Central Coast
Potamogeton epihydrus ssp. *nuttallii* Nuttall's pondweed	None	S2.2?	G5T5Q	None	2	Wetland/Obligate	WS	Outer North Coast Ranges, High Sierra Nevada, Modoc Plateau, to AK, eastern North America, CO.
Rhynchospora alba white beaked-rush	None	S3.2	G5	None	2	Wetland/Obligate	WS	central and southern North Western California (Mendocino and Sonoma cos.), southeast High Cascade Range (Lassen Co.); circumboreal
Sanguisorba officinalis great burnet	None	S2.2	G5?	None	2	Wetland/Facultative	WNA	central North Coast, northwestern Klamath Ranges, and northern Outer North Coast Ranges, extending to Alaska
Saxifraga nuttallii Nuttall's saxifrage	None	S1.1	G4?	None	2	Forest/Rocky Mesic	WNA	North Coast (Del Norte Co.), extending sporadically to Washington
Sidalcea calycosa ssp. *rhizomata* Point Reyes checkerbloom	None	S2.2	G5T2	None	1B	Wetland/Obligate	CA	central and southern North Coast and the northern Central Coast

Scientific name Common name	State status	State rank	Global rank	Federal status	CNPS status	General redwood forest habitat	Distribution	Geographic regions (Hickman 1993 and others)
Sidalcea malachroides maple-leaved checkerbloom	None	S2.2	G2	None	1B	Forest/Grassland Interface	WNA	North Coast, the Outer North Coast Ranges, northern and central Central Coast, San Francisco Bay Area, northern South Coast Ranges, and western Oregon
Sidalcea malviflora ssp. *patula* Siskiyou checkerbloom	None	S1.1	G5T1	None	1B	Forest/Grassland Interface	WNA	northern North Coast in Humboldt and Del Norte counties, and southwestern Oregon
Sidalcea malviflora ssp. *purpurea* purple-stemmed checkerbloom	None	S2.2	G5T2	None	1B	Forest/Grassland Interface	CA	central North Coast and the northern Central Coast
Sidalcea oregana ssp. *eximia* coast checkerbloom	None	S1.2	G5T1	None	1B	Wetland /Obligate (sp.)	CA	northern North Coast and the Outer North Coast Ranges
Tiarella trifoliata var. *trifoliata* trifoliate laceflower	None	S2S3	G5T5	None	3	Forest Generalist	WNA	Klamath Ranges, near the Humboldt and Trinity County line, to Alaska and Montana
Viburnum ellipticum oval-leaved viburnum	None	S2.3	G5	None	2	Forest/Shubland Interface	WNA	Northwestern California, the northern and central Sierra Nevada Foothills, and the San Francisco Bay Area; extending to Washington

Abbreviations:
SR: state listed rare species
SE: state listed endangered species
ST: state listed threatened species
SC: state candidate species

FE: federal listed endangered species
FT: federal listed threatened species
FC: federal candidate species

S-rank: reflection of overall condition of an element throughout its California range
G-rank: reflection of overall condition of an element throughout its Global range
T-rank: attached to the global rank and reflects the global situation of a subspecies or variety (G-rank will reflect the entire species)
S/G-rank H: all sites historical
S/G-rank Q: element very rare but there are taxonomic questions
S/G-rank X: all sites are extirpated (XC extinct in wild but exists in cultivation)
S1/G1: less than 6 element occurrences (EOs) or less than 1,000 individuals or less than 2,000 acres
  S1.1: very threatened
  S1.2: threatened
  S1.3: no current threats known
S2/G2: 6-20 EOs or 1,000-3,000 individuals or 2,000-10,000 acres
  S2.1: very threatened
  S2.2: threatened
  S2.3: no current threats known
S3/G3: 21-100 EO's or 3,000-10,000 individuals or 10,000-50,000 acres
  S3.1: very threatened
  S3.2: threatened
  S3.3: no current threats known
S4/G4: apparently secure; this rank clearly lower than 3 but factors exist to cause some concerns such as some threat or somewhat narrow habitat (NO THREAT RANK)
S5/G5: demonstrably secure to eradication (NO THREAT RANK)

CNPS List 1A: plants presumed extinct in California
CNPS List 1B: plants rare, threatened, or endangered in California, and elsewhere

CNPS List 3: plants about which we need more information, a review list

OBL: obligate wetland plants with >99 percent occurrence in wetlands;
FACW: facultative wetland plants with 67 to 99 percent occurrence in wetlands
FAC: facultative plants with 34 to 66 percent occurrence in wetlands
FACU: facultative upland plants with 1 to 33 percent occurrence in wetlands
UPL: obligate upland plants with <1 percent occurrence in wetlands
NI: no indicator (insufficient information) for the region (rated neutral)
NL: not listed (rated upland)
  plus sign (+): frequency toward higher end of a category
  minus sign (-): frequency toward lower end of a category
  asterisk (*): indicates tentative assignment based on limited information.

Distribution Codes:
  CA: California endemic
  WNA: Western North America
  WS: Widespread

Rare Plants of the Redwood Forest and Forest Management Effects[1]

Teresa Sholars[2] and Clare Golec[3]

Abstract

Coast redwood forests are predominantly a timber managed habitat type, subjected to repeated disturbances and short rotation periods. What does this repeated disturbance mean for rare plants associated with the redwood forests? Rare plant persistence through forest management activities is influenced by many factors. Persistence of rare plants in a managed landscape is not in itself an indication of viability, but may reflect an overall increase, equilibrium, or decline in numbers. Although the persistence of some species can seemingly mimic weedy behavior, it is important to distinguish pioneer species behavior from the weedy behavior of invasive exotics. Individual species will have different responses to disturbance based on their life history and habitat requirements, as well as the type, intensity, and frequency of disturbance. Human disturbance and natural disturbance regimes are frequently and mistakenly viewed as equivalent. In addition, human disturbance regimes commonly create habitat opportunities for invasive exotics that readily out-compete rare plants for habitat. Knowing why rare plants persist in the managed redwood forest is dependent on understanding their distribution, habitat, life history, and sensitivity to disturbance. This paper will examine forest management effects on 10 rare species of the redwood forest.

Introduction

Coast redwood forests are a relic vegetation type closely associated with a narrow ecological region that provides a humid and temperate environment. The redwood forest does not have a unique flora but does have many closely associated species and habitats. There are approximately 1,518 rare plants in California of which 86 percent are herbaceous plants, 12 percent shrubs, and 2 percent trees (CNPS 2003). In the northern redwood forest there are 45 rare vascular plant species that are closely associated with redwoods (Leppig and others in press), and 94 percent are herbaceous plants. The rare flora of the redwood forest is largely herbaceous in nature and diminutive in habit. Redwood forests are predominately a timber managed habitat type with over 80 percent privately owned (primarily industrial timberlands), and are subjected to repeated disturbance events and short rotation periods (Noss 2000). What does this repeated disturbance mean for rare plants associated with the redwood forests?

The redwood forest is subject to both natural and anthropogenic disturbances; however, in the last century the anthropogenic impacts have greatly increased with timber management. Disturbance is defined here as a "relatively" discrete event in

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Professor of Biology, Mendocino Coast Campus, College of the Redwoods, 1211 Del Mar Drive, Fort Bragg, CA 95437. email: teresa-sholars@redwoods.edu

[3] Environmental Scientist/Botanist, California Department of Fish and Game, 1031-A South Main Street, Fort Bragg, CA 95437. email: cgolec@dfg.ca.gov

time that is ecologically disruptive at a localized or landscape level and changes resources, substrates, or the physical environment (Imper 2001). Rare species are of particular concern given their greater risk of extinction and potential loss of their economic, medicinal, and ecological function (Kaye and others 1997). This higher risk of extinction coupled with scattered and localized distribution, diminutive habit, and physical immobility of rare plants increases their vulnerability to disturbance. Especially with landscape disturbance regimes such as timber harvesting, which involves large tree felling and yarding, heavy equipment use, and road building.

Forest management activities have been perceived as mimicking "natural" disturbance regimes such as fire, windthrow and flooding by providing forest openings and mineral soil exposure. However, forest management activities are more frequent, and less variable in size and intensity than natural disturbance regimes. Timber harvesting often consists of concentrated impacts over a localized and uniform area, which may not provide the necessary mosaic of undisturbed refugia for native plants, and sources for recolonization (Halpern and Spies 1995).

Timber Harvesting Plans every so often include comments such as: 1) rare species thrive in disturbance regimes; 2) timber harvesting encourages the spread and diversity of native plants; and 3) timber harvesting has little to no significant impacts on rare plants (CDF 2003, CDF 2004). These comments often do not address the fact that not all rare plants belong to one type of disturbance response group (such as early successional), and that while many rare plants may persist through disturbance regimes, they may not all thrive.

It is our premise that forest management activities may do one of two things. They either significantly reduce the viability and persistence of rare species, or they may create rare plant habitat. The heart of this paper is that it is important to differentiate between timber management activities that have negative affects on rare plants and activities that can have positive affects. Another important point is to recognize the difference between rare plants that are truly early successional species and those that are associated with forest understories or openings. Early successional species have evolved with landscape disturbance events such as fire, landslides and flooding. These types of events deforest portions of the landscape and expose mineral soil. Forest openings species are associated with open areas within the forest created by poor soils or small scale disturbances such as tree gaps. The habitat in forest openings has increased light exposure without significant changes to the forest microclimate or soil. Forest understory species have evolved beneath the forest canopy that provides a shady, moist, and sheltered habitat with a well-developed duff layer.

Methods

A subset of ten rare plants associated with the northern redwood forest region (Del Norte, Humboldt and Mendocino counties) were identified for this study, and were selected to include various habitat and life history requirements. The selected rare plants are as follows:

1. Humboldt milk-vetch (*Astragalus agnicidus* Barneby)
2. swamp harebell (*Campanula californica* (Kell.) Heller)
3. coast fawn lily (*Erythronium revolutum* Sm.)

4. thin-lobed horkelia (*Horkelia tenuiloba* (Torr.) Gray)
5. running-pine (*Lycopodium clavatum* L.)
6. leafy-stemmed mitrewort (*Mitella caulescens* Nutt.)
7. North Coast semaphore grass (*Pleuropogon hooverianus* (L. Benson) J.T. Howell)
8. seacoast ragwort (*Senecio bolanderi* (Gray) var. bolanderi)
9. maple-leaved checkerbloom (*Sidalcea malachroides* (H. & A.) Gray)
10. long-beard lichen (*Usnea longissima* Ach.)

These ten rare plants were analyzed in terms of available ecological information and potential forest management impacts. The analysis was conducted through the: 1) utilization of sensitive plant occurrence data in California Natural Diversity Data Base RareFind3 (CNDDB 2003) and California Native Plant Society's Electronic Inventory of Rare and Endangered Plants of California (CNPS 2003); 2) compilation of habitat and life history information provided in literature and flora treatments such as *The Jepson Manual* (Hickman 1993); 3) review of reports such as Timber Harvesting Plans, sensitive plant surveys, and sensitive plant monitoring and annual summaries; and 4) field observations of the authors.

Results

The subsequent two tables summarize the ecology *(table 1)* and responses to disturbance *(table 2)* of these ten redwood forest rare plants.

Table 1— *Ecological information on selected redwood forest rare plants.*

Scientific name	Life form / natural history	Habitat/ distribution	Light requirement/ shade tolerance	Moisture/ hydrology	Soil disturbance/ compaction	Reproduction/ dispersal/ seed bank
***Astragalus agnicidus* Humboldt milk-vetch**	Perennial herb to sub-shrub Early successional species and short-lived 5 to 10 years (Bencie 1995) Low tolerance of vegetation competition/canopy closure (Golec 2004) Temporal habitat, reliance on seed banks (Pickart and others 1992) Invasive exotics compete for habitat (Golec 2004) Sensitive to herbicides, even Imazapyr registered not to affect legumes, (Lundby 2003) Browsing commonly noted by rodents and other vertebrates (Pickart and others 1992, Golec 2004) All life stages present indication of population stability, alternately absence of seedlings an indication of a declining population (Pickart and others 1992)	Openings in redwood forest tanoak forest, and mixed evergreen forest, often disturbed sites and ridgelines with south to west aspects Outer North Coast Ranges in southern Humboldt County and northern Mendocino County (Hickman 1993) Belongs to small distinct taxonomical group of species (four species distinctive from the European and Asian species) that are all rare or highly localized (Barneby 1957) Closely associated with redwood region, limited distribution (CNDDB 2003)	Full to partial light Shade intolerant (Enberg 1990) Full light exposure affects seed coat permeability (Pickart and others 1992)	Xeric Susceptible to damping off	Tolerant to disturbance (Pickart and others 1992, Golec 2004) Plants more robust in habit on uncompacted soils (Golec 2004)	Pollinated commonly by bumble bees (*Bombus* sp.) with occasional visits from honey bees (*Apis mellifera* L.) and syrphid flies (Bencie 1997) Insect visitation is necessary for successful fruit set (Bencie 1997) Limited and clumped seed dispersal via dehiscence of legume (Bencie 1997) Seed caching by rodents Seeds have impermeable seed coat like many members of this family that allows for long term viability in the soil (Baker 1989) Highest density of seed found in the first 3 inches soil (Bencie 1995) Scarification and stratification of seed needed for good germination (Enberg 1990)
***Campanula californica* swamp harebell**	Short-lived perennial herb Weak stemmed and shallowly rooted species with clambering habit via stiffly recurved hairs along leaf margins and stem Forest openings species	Wet seeps & swales, marshes and swamps, and seasonally wet forest openings Northern Central Coast and the southern North Coast (Hickman 1993) Closely associated with redwood region, limited distribution (CNDDB 2003)	Full to partial light Shade intolerant	Mesic to hydric	Tolerant of soil disturbance if light and hydrology habitat requirements are met Not tolerant of soil compaction	Many small seeds in capsule, dehiscing through basal pores (Hickman 1996)

Scientific name	Life form / natural history	Habitat/ distribution	Light requirement/ shade tolerance	Moisture/ hydrology	Soil disturbance/ compaction	Reproduction/ dispersal/ seed bank
Erythronium revolutum **coast fawn lily**	Perennial bulbiferous herb with retractable bulbs Dormant during summer Forest understory species Slow establisher and long-lived with moderate tolerance of vegetation competition (Parsons 2003)	Mesic sites in redwood and mixed evergreen forests (such as north aspects), and margins of streams, swamps, and bogs North Coast and the Outer North Coast Ranges to British Columbia (Hickman 1993) Horticulturally valuable species and over collected	Full to partial light Shade tolerant	Mesic Frequently associated with wetlands, facultative wetland plant (Reed 1988)	Not tolerant to soil disturbance and compaction, many liliaceous species sensitive to soil compaction (Imper 2003) Requires excellent drainage and modification of compacted or other water-holding soils may be necessary (Hickman 1993)	"Bulb-building" and flowering takes 5-7 years from seed germination (Schmidt 1980)
Horkelia tenuiloba **thin-lobed horkelia**	Short-lived perennial herb Forest openings species	Forest openings, chaparral, grasslands, and seasonal wetlands, often sandy soils Central and southern North Coast, the central and southern Outer North Coast Ranges, and the northwestern San Francisco Bay Area (Hickman 1993)	Full to partial light	Xeric to seasonally hydric	Tolerant to soil disturbance but not compaction Requires excellent drainage and modification of compacted or other water-holding soils may be necessary (Hickman 1993)	Fruit an achene (Hickman 1993)
Lycopodium clavatum **running-pine**	Perennial and evergreen rhizomatous herb with shallow root system Slow establisher and long-lived Forest understory species ~80 years following disturbance needed to allow for higher levels of frequency to develop as well as for clonal expansion or a substantial increase in cover to occur (Nauertz 1999) Mats can range from very young to hundreds of years old (Golec 2001)	Mesic sites in coniferous forests and margins of marshes and swamps North Coast, extending to Alaska, Montana, eastern North America, the Caribbean, South America, Eurasia, Africa, and Asia (Hickman 1993) Closely associated with redwood region (CNDDB 2003) Special forest product (Nauertz 1999) with many uses, and over collected.	Partial light Shade tolerant Optimum canopy closure around 60% (Golec 2001, Nauertz 2002)	Mesic to hydric Not strongly associated with wetlands, facultative wetland plant (Reed 1988) Moisture and light limiting environmental factors (Golec 2001) Correlated with large woody debris that retain water and well-developed organic layer (Golec 2001)	Not tolerant, sensitive to soil disturbance and compaction (Golec 2001, Nauertz 2004) Adventitious roots enable travel over compacted soils (Nauertz 2004)	Reproduces sexually by spores and vegetatively by rhizomes Vegetative reproduction most commonly observed, and sexual reproduction rarely observed (Nauertz 2002) Spores dispersed by wind Spore germination and gametophyte formation takes place below soil and in the dark (Whitter 1988) Alternation of generations (gametophyte/sporophyte) cycle may take up to 20 years (Nauertz 2002)

Scientific name	Life form / natural history	Habitat/ distribution	Light requirement/ shade tolerance	Moisture/ hydrology	Soil disturbance/ compaction	Reproduction/ dispersal/ seed bank
Lycopodium clavatum **running-pine**				Sensitive to warmer temperatures and drier forest conditions (Nauertz 2004)		
Mitella caulescens **leafy-stemmed mitrewort**	Perennial rhizomatous herb with shallow root system Forest understory species	Mesic sites in mixed evergreen forest, redwood forest, and montane coniferous forests (commonly along riparian corridors), and meadows North Coast, Klamath Ranges to British Columbia, Montana (Hickman 1993)	Partial light Shade tolerant	Mesic to hydric Sensitive to desiccation	Not tolerant of soil disturbance and compaction Tolerant of soil deposition with high water	Capsule with many small seeds (Hickman 1993)
Pleuropogon hooverianus **North Coast semaphore grass**	Tall perennial rhizomatous bunchgrass Dormant during summer Forest openings species Threatened by herbicides, mowing, conifer encroachment (Showers 2002) Potentially threatened by invasive such as annual ryegrass (*Lolium multiflorum* Lam.), Harding grass (*Phalaris aquatica* L.) and pennyroyal (*Mentha pulegium* L.)	Seasonally wet to wet openings in redwood/mixed evergreen forest and oak woodland, and grasslands Southern North Coast and the northern Central Coast (Hickman 1993) Closely associated with redwood region (CNDDB 2003)	Full to partial light Shade tolerant	Mesic to hydric, often seasonally wet sites	Not tolerant of soil disturbance and compaction	Does not have a large persistent seed bank (Showers 2002)
Senecio bolanderi **var.** ***bolanderi*** **seacoast ragwort**	Perennial rhizomatous herb Forest understory to opening species Low tolerance to vegetation competition	Coastal scrub and coastal coniferous forest (includes redwood forest) and along streams North Coast, extending to Washington (Hickman 1993) Closely associated with redwood region (CNDDB 2003)	Full to partial light Shade tolerant	Mesic	Not tolerant of soil disturbance or compaction	Fruit with pappus of thin, minutely barbed deciduous bristles (Hickman 1993), possibly animal dispersed

Scientific name	Life form / natural history	Habitat/ distribution	Light requirement/ shade tolerance	Moisture/ hydrology	Soil disturbance/ compaction	Reproduction/ dispersal/ seed bank
Sidalcea malachroides **maple-leaved checkerbloom**	Perennial, rhizomatous and gynodioecious sub-shrub Forest openings species Small populations appear to have lower viability perhaps influenced in part by plant being functionally diociecous (Leppig 2004) Not tolerant of direct impacts that crush or uproot plants (Lundby 2003) Invasive exotics compete for habitat (Golec 2004)	Coastal scrub and prairie, and openings in coastal coniferous and mixed evergreen forests, often disturbed sites. North Coast, the Outer North Coast Ranges, northern and central Central Coast, San Francisco Bay Area, northern Outer South Coast Ranges, and western Oregon (Hickman 1993) Closely associated with redwood region (CNDDB 2003)	Full to partial light Shade intolerant	Mesic Sensitive to desiccation with total canopy removal (Golec 2004)	Tolerant of soil disturbance, but more robust on uncompacted soils	Fruit:5–10 segmented and indehiscent, generally with one seed per fruit segment (Hickman 1993) Heavy weevil predation of seeds
Usnea longissima **long-beard lichen**	Epiphytic lichen Long-lived, slow establisher Sensitive to air pollution and timber harvesting (Keon 2001, Bittman 2003) Dependent and pendant on older conifers, hardwoods and snags, presence correlated with stand age (Keon 2001) Most effective strategy for conservation of pendant lichens is the protection of gaps, wolf trees and old-growth remnant trees (Neitlich and McCune 1997) Encourage forest management practices to retain groups of occupied trees, and set-aside high quality populations (Bittman 2003)	North Coast Coniferous Forest, Broadleaved Upland Forest, grows in the “redwood zone” on a variety of trees (CNDDB 2003) San Francisco Bay Area northward to Humboldt County in the North Coast Ranges (Hale and Cole1988); AK to CA, western Cascades (McCune and Geiser 1997) Closely associated with redwood region, and currently no extant occurrences south of Sonoma Co. (Bittman 2003) Declining throughout Europe, Pacific Northwest may contain the best remaining populations of this species (Keon 2001)	Full light, generally high in the canopy	Moist microclimate	N/A	Disperses almost exclusively by vegetative fragmentation, fertile individuals very rare (Keon 2001) Wind dispersed and dispersal limited (Keon 2001) Retention of remnant trees containing species can enhance dispersal within regenerating stands (Keon 2001)

Table 2—*Positive and negative disturbance impacts to selected redwood forest rare plants.*

Scientific name	Positive disturbance impacts	Negative disturbance impacts
Astragalus agnicidus **Humboldt milk-vetch**	Canopy removal increases light and improves habitat Reduced vegetation competition and shading Seed scarification via soil disturbance and sun exposure/temperature Creation of seasonal road bank and fill habitat Potential dispersal through mechanical soil movement such as seasonal road grading	Plants crushed or uprooted Soil disturbance buries seed bank too deeply for germination (> 3") Heavy slash deposition obstructs light to herbaceous layer limiting germination/growth Displacement and loss of habitat with introduction and proliferation of invasive exotics Canopy closure with reforestation creates unsuitable habitat Mortality from herbicide application
Campanula californica **swamp harebell**	Partial canopy removal increases light and improves habitat Seasonal road drainage creates mesic to hydric habitat	Plants crushed or uprooted Desiccation from changes in hydrology and full light exposure, Heavy slash deposition obstructs light to herbaceous layer limiting germination/growth Displacement and loss of habitat with introduction and proliferation of invasive exotics Mortality with soil compaction Mortality from herbicide application
Erythronium revolutum **coast fawn lily**	Partial canopy removal increases light and improves habitat	Plants crushed or uprooted Desiccation from canopy removal and full light exposure Heavy slash deposition obstructs light to herbaceous layer limiting germination/growth Mortality with soil disturbance or compaction Mortality from herbicide application
Horkelia tenuiloba **thin-lobed horkelia**	Canopy removal increases light and improves habitat Reduced vegetation competition and shading	Plants crushed or uprooted Heavy slash deposition obstructs light to herbaceous layer limiting germination/growth Displacement and loss of habitat with introduction and proliferation of invasive exotics Mortality with soil compaction Mortality from herbicide application

Scientific name	Positive disturbance impacts	Negative disturbance impacts
Lycopodium clavatum **running-pine**	Partial canopy removal increases light and shading	Plants crushed or uprooted Greater then 50% canopy removal decreases habitat quality through desiccation and full light exposure Dense canopy closure with reforestation creates unsuitable habitat Frequent disturbance inhibits establishment Mortality from herbicide application
Mitella caulescens **leafy-stemmed mitrewort**	Partial canopy removal increases light and improves habitat	Plants crushed or uprooted Desiccation from changes in hydrology and full light exposure, Heavy slash deposition obstructs light to herbaceous layer limiting germination/growth Mortality with soil disturbance or compaction Mortality from herbicide application
Pleuropogon hooverianus **North Coast semaphore grass**	Canopy removal increases light and improves habitat Seasonal road drainage creates mesic to hydric habitat	Plants crushed or uprooted Desiccation from changes in hydrology Heavy slash deposition obstructs light to herbaceous layer limiting germination/growth Displacement and loss of habitat with introduction and proliferation of invasive exotics Canopy closure with reforestation creates unsuitable habitat Mortality from herbicide application
Senecio bolanderi* var. *bolanderi **seacoast ragwort**	Partial canopy removal increases light and improves habitat	Plants crushed or uprooted Heavy slash deposition obstructs light to herbaceous layer limiting germination/growth Desiccation from changes in canopy removal and full light exposure Mortality from herbicide application
Sidalcea malachroides **maple-leaved checkerbloom**	Canopy removal increases light and improves habitat Reduced vegetation competition and shading Creation of seasonal road bank and fill habitat Seed bank germination with soil disturbance and sun exposure/temperature Dispersal with mechanical soil movement such as seasonal road grading	Plants crushed or uprooted Heavy slash deposition obstructs light to herbaceous layer limiting germination/growth Desiccation from changes in canopy removal and full light exposure Dense canopy closure with reforestation creates unsuitable habitat Displacement and loss of habitat with introduction and proliferation of invasive exotics Mortality from herbicide application

Scientific name	Positive disturbance impacts	Negative disturbance impacts
Usnea longissima **long-beard lichen**	Partial canopy removal (non-host tree) increases light and improves habitat	Loss of habitat with removal of host tree or snag Mortality from broad scale herbicide application and smoke from prescribed burning

Discussion

The results indicate the response of these ten redwood forest rare plants to disturbance is correlated to the ecology and life history of the species. Important ecological results were: 1) forest understory species were dependent on shade and the moist forest microclimate, and were sensitive to soil disturbance and compaction (running-pine, coast fawn lily, and seacoast ragwort); 2) early successional species in some cases but not all increased with canopy removal and soil disturbance but not soil compaction (Humboldt milk-vetch and maple-leaved checkerbloom); and 3) forest openings species benefit with full light exposure but not necessarily with soil disturbance and compaction (North Coast semaphore, swamp harebell, and thin-lobed horkelia). Potential impacts to these species as a result of disturbance associated with timber management activities are strongly weighted toward having negative effects. Direct impacts (crushing or uprooting plants), heavy slash deposition (obstruction of light to herbaceous layer), and herbicide application are the primary negative impacts. Other secondary impacts are changes in hydrology or forest microclimate, soil disturbance and compaction, and introduction and proliferation of invasive exotics. The primary positive impacts were increase in light exposure with canopy removal, and augmentation of habitat and hydrology associated with seasonal roads. Overall, the most significant conclusion from the results is the paucity of comprehensive and published ecological data on redwood forest rare plants, and their response to disturbance regimes such as timber harvesting.

Clearly many rare plants are dependent on environmental factors like hydrology and amount of light exposure. Swamp harebell and North Coast semaphore grass need partial to full light exposure and at least seasonal hydric conditions. Thin-lobed horkelia benefits from full light exposure. Long-beard lichen requires full sun but is dependent on large tree structure, and may require decades to recover from canopy removal (Halpern and Spies 1995). Forest understory species such as leafy-stemmed mitrewort, coast fawn lily, running-pine, and seacoast ragwort are dependent on the shady and moist forest microclimate, and generally are affected by management activities that significantly alter these forest attributes. Swamp harebell and running-pine are shallowly rooted and are more sensitive to soil disturbance, compaction, and changes in hydrology (Halpern and Spies 1995). Humboldt milk-vetch and maple-leaved checkerbloom are early successional in nature, and these species' habitat requirements were the most compatible with timber harvesting activities such as canopy removal and exposure of mineral soil.

Persistence of rare plants in a managed landscape is not in itself an indication of viability, but may reflect an overall increase, equilibrium, or decline in numbers over time. Rare plant persistence is influenced by many factors as well as how management impacts affect these factors singularly or cumulatively. These factors can be abiotic or biotic in nature. Abiotic factors include topography, aspect, shade and light exposure, hydrology, forest microclimate, and soil type, chemistry and structure. Biotic factors include population size, number of reproductive plants, seed set, dispersal and seed banking, canopy cover, vegetation competition, invasive non-native displacement, organic layer and large woody debris, predation, pollinators and microbial relationships such as such as mycorrhizae and nitrogen fixation.

Rare plant persistence in a managed forest is also influenced by the species rate of establishment (fast vs. slow). The establishment of a reproducing individual can take one season or years. For example, Humboldt milk-vetch and maple-leaved checkerbloom have much faster rates of establishment than coast fawn lily and running-pine. Under appropriate habitat conditions, Humboldt milk-vetch and maple-leaved checkerbloom can develop into robust reproducing plants within several years unlike coast fawn lily that takes over five years to flower (Schmidt 1980). Running-pine has a lengthy and complex maturation. The spores of running-pine first develop into a subterranean gametophyte prior to developing into the familiar mat-forming sporophyte, and this maturation of the gametophyte can take over 10 years and is mycorrhizal dependent (Leppig 2004). In addition, slow establishers like lilies [fairy bells (*Disporum* spp.), western trillium (*Trillium ovatum* Pursh), and single-flowered clintonia (Clintonia uniflora (Schultes) Kunth)] and orchids [rattlesnake plantain (*Goodyera oblongiflora* Raf.), heart-leaved twayblade (*Listera cordata* (L.) R.Br.), and western coralroot (*Corallorhiza mertensiana* Bong.)] are more susceptible to local extirpation (Halpern and Spies 1995, Jules 1997). Bryophytes and lichens such as long-beard lichen are often lacking in younger stands due to slow growth rates as well as dependency on large tree structure, moist microclimatic requirements, and limited dispersal (Halpern and Spies 1995)

Timber management activities can alter factors influential in the persistence of rare plants. Alteration of these factors can result in direct, indirect, and cumulative impacts to rare plants. Whether negative or positive and to what degree, is based on the type, intensity, size and frequency of disturbance, and the individual species habitat and life history requirements. Potential direct impacts (“take”) can result from timber felling and yarding, road construction and maintenance, log decking, and other staging area activities. Indirect impacts can include canopy alteration (change in shade and light exposure), change of hydrology, disruption of symbiosis (such as mycotropic or mycorrhizal relationships), disturbance of root systems, burial of seeds below germination depths, exposure of mineral soil, reduction of vegetation competition, and slash accumulation. Lastly, cumulative impacts can result with frequent disturbance regimes (especially with slow establishing species), selective pressure toward a certain type of habitat or taxa, and increase habitat opportunities for invasive exotics.

To avoid negative impacts and ensure positive effects to rare plants within timber managed landscapes, it is important to understand the specific niche requirements. Understanding specific niche requirements increases the probability of finding rare species and effectively mitigating impacts (Sholars 2004). However, distribution of a rare plant and its dispersal mechanisms are equally important. Not all plants are limited by available habitat; dispersal limitation and regional speciation can be stronger factors for determining occurrence. Understanding rare plant distribution, habitat, and life history requires developing good inventory and ecological data collection methods. Rare plants distribution is often determined by forest managers solely through the use of electronic databases such as CNDDB RareFind or CNPS’s Electronic Inventory of Rare and Endangered Plants of California. The problem is that these are positive occurrence databases that contain only known locations of species that have been observed and documented. Absence of data in these sources does not mean that a rare species is not present (Jirak 1998). Known occurrence data needs to be used by forest managers in conjunction with inventories and monitoring in order to better manage the forest and not contribute to the further decline of rare plants.

Although the persistence of some species can seemingly mimic weedy behavior, it is important to distinguish pioneer species (early successional) behavior from the weedy behavior of invasive exotics. Pioneer species will have different responses to impacts from disturbance based on their early successional role. Humboldt milk-vetch likely evolved and benefited with fire disturbance events, as indicated by observations of increased seedling expression at burned sites (Golec 2004). Slash is, however detrimental to its establishment and growth. Prior to fire suppression, forest fires were significant disturbance events in generating large forest openings and exposure of mineral soil. Fire history of the redwood forest shows surface fires created frequent disturbances in coast redwood forests of Mendocino County prior to the early 20th century. Average fire frequency varied between 6 and 20 years in one study (Brown and Baxter 2003). With fire suppression timber harvesting has for the most part replaced these prehistoric disturbance events in generating forest openings and exposing mineral soil.

Lastly, invasive exotics are often introduced or proliferate with repeated disturbance. Invasive non-native species are a worldwide threat to biodiversity, and this threat is second only to direct habitat loss and fragmentation. Their effects are severe, complex, and extensive to the environment, and include displacement of native plants and animals. Rare plants are particularly vulnerable to invasive exotics and every rare plant can be readily displaced by invasive exotics such as the early successional species purple pampas grass (*Cortaderia jubata* (Lemoine) Stapf), or the shade tolerant forest understory species English ivy (*Hedera helix* L.). In addition to natural dispersal through wind, water and wildlife, the spread of non-native invasive plants on forest lands are often facilitated by heavy equipment and vehicle traffic, road construction and maintenance, erosion control seeding and mulching, and vegetation and soil disturbance associated with management activities. Some of the problematic non-native and invasive plants associated with redwood forests are bull thistle (*Cirsium vulgare* (Savi) Ten.), purple pampas grass, Scotch broom (*Cytisus scoparius* (L.) Link), French broom (*Genista monspessulana* (L.) L. Johnson), English ivy and Himalayan blackberry (*Rubus discolor* Weihe & Nees).

Conclusions

The concept of "disturbance" as used in forest ecology is subjective, often complex, and oversimplified. Many rare plants are not necessarily benefited by disturbance from forest management activities. Forest management activities may create what appears to be potential habitat; however, census and monitoring data do not always indicate an increase in rare plants. This may be due to unapparent niche requirements, poor viability (such as small populations, or pollination and dispersal limitations), and proliferation of invasive non-native plants.

To ensure the persistence of rare plants, forest managers need to take into account the distribution, habitat, and life history of each individual rare plant in their management area. A great deal of positive effect can take place given care to both the direct and indirect impacts of management activities. With increased knowledge of the biology and ecology of rare species, forest management activities can have a positive effect on rare plant viability and survival. Without this knowledge and the interest in fostering rare plant conservation, forest management activities are sure to continue to play a role in the increase of rare plant extirpation and extinction.

References

Baker, H.G. 1989. **Some aspects of the natural history of seed banks**. In: Leck, M.A.; Parker, V.T.; Simpson, R.L., eds. Ecology of soil seed banks. San Diego, CA: Academic Press; 9-12.

Barneby, R.C. 1957. ***Astragalus agnicidus*, a new locoweed form Humboldt County, California**. Madrono 14(14): 37-40.

Bencie, R. 1995. **A preliminary investigation of the seed bank of *Astragalus agnicidus***. Unpublished paper for The Nature Conservancy of California; 10 p.

Bencie, R. 1997. **Genetic variation and inbreeding depression in the rare California endemic, *Astragalus agnicidus* (Leguminosae)**. Arcata, CA: Humboldt State University; 46 p. M.A. thesis.

Bittman, R. 2003. **Rare plant status review: *Usnea longissima*.** Sacramento, CA: California Department of Fish and Game; 3 p.

Brown, P.; Baxter, B. 2003. **Fire history in coast redwood forests of the Mendocino Coast, California.** Northwest Science 77(2): 147-158.

California Department of Forestry and Fire Protection. 2003. **Timber harvesting plan 1-03-034 MEN**. Santa Rosa, CA.

California Department of Forestry and Fire Protection. 2004. **Timber harvesting plan 1-04-027 MEN**. Santa Rosa, CA.

California Native Plant Society. 2003. **Electronic inventory of rare and endangered plants of California.** Sacramento, CA: California Native Plant Society.

California Natural Diversity Data Base. 2003. **RareFind version 3.0.3** [Computer program]. Sacramento, CA: California Department of Fish and Game.

Enberg, A.W. 1990. **The effects of shading on the emergence, growth, and development of the rare and endangered California plant species, *Astragalus agnicidus***. Arcata, CA: Humboldt State University, Department of Biology; 15 p. Senior project.

Golec, C.T. 2001. **Simpson Timber Company's running-pine assessment and survey**. Eureka, CA: Natural Resources Management Corporation; 32 p.

Golec, C.T. 2004. **Humboldt Milk-vetch (*Astragalus agnicidus*) occurrences notes, Mendocino County**. Unpublished notes on file, California Department of Fish and Game, Fort Bragg, CA.

Golec, C.T. 2004. **Campbell Timberland Management, LLC summary of sensitive and uncommon plants with known occurrence on ownership;** 2001 through 2003. Unpublished notes on file, California Department of Fish and Game, Fort Bragg, CA.

Golec, C.; Labanca, T.; Leppig, G. 2006. **The conservation of sensitive plants on private redwood timberlands in northern California**. In: Standiford, R.B.; Giusti, G.A.; Valachovic, Y.; Zielinkski, W.J., technical coordinators. Proceedings of the Redwood Region Forest Science Symposium. 2004 March 15-17; Rohnert Park, CA. Gen. Tech. Rep. PSW-GTR-194. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture [this volume].

Halpern, C.B.; Spies, T.A. 1995. **Plant species diversity in natural and managed forests of the Pacific Northwest.** Ecological Applications 5(4): 913-934.

Hickman, J.C. 1993. **The Jepson manual: higher plants of California**. Berkeley, CA: University of California Press; 1400 p.

Imper, D. 2004. **The role of disturbance in the ecology of rare plants in northwestern California;** 2002 symposium of the ecology and management of rare plants of northwestern California. Unpublished draft supplied by the author.

Imper, D. Ecologist/Botanist, United States Fish and Wildlife Service at Arcata Office. [Telephone conversation with Clare Golec]. 15 December 2003.

Jepson, W.L. 1925. **A manual of the flowering plants of California**. Berkeley, CA: University of California Press; 1238 p.

Jirak, G.A. 2001. **Impact of timber harvesting on rare, threatened, and endangered native plants.** Fremontia 29(3-4): 38-50.

Jules, E.S. 1997. **Consequences of forest fragmentation for the understory plant, *Trillium ovatum* (Liliaceae)**. In: Proceedings of the conservation and management of native plants and fungi symposium; 1995 November 15-17; Corvallis, OR. Corvallis, OR: Native Plant Society of Oregon; 201-206.

Kaye, T.N.; Meinke, R.; Kagan, J.; Virlakus, S.; Chambers, K.; Zika, P.; Nelson, J. 1997. **Patterns of rarity in the Oregon flora: implications for conservation and management**. In: Proceedings of the conservation and management of native plants and fungi symposium; 1995 November 15-17; Corvallis, OR. Corvallis, OR: Native Plant Society of Oregon; 1-10.

Keon, D.B. 2001. **Factors limiting the distribution of the sensitive lichen *Usnea longissima* in the Oregon coast range; habitat or dispersal?** Corvallis, OR: Oregon State University. M.S. thesis.

Leppig, G. 2004a. **The rare California club-mosses.** Fremontia 32(4): 20-22.

Leppig, G. 2004b. **PALCO rare plant program assessment of 2003 survey and monitoring results and proposed 2004 program recommendations**. Unpublished report, California Department of Fish and Game, Eureka, CA.

Lundby, D.M. 2003. **Monitoring report 2001-2002 Humboldt milk vetch (*Astragalus agnicidus*), Bald Hill THP (#1-01-197 MEN), Baptiste THP (#1-01-416 MEN)**. Fort Bragg, CA: Campbell Timberland Management, LLC; 34 p.

Lundby, D.M. 2003. **The Shack THP (#1-01-062 MEN) monitoring report.** Fort Bragg, CA: Campbell Timberland Management, LLC.; 17 p.

Lundby, D.M.; Gragg, J.E. 1998. **Experimental transplantation of *Campanula californica*, Sandhill Lake Timber Harvest Plan 1-95-505 MEN**. Unpublished manuscript, California Department of Fish and Game, Yountville, CA.

Massicotte, H.B. 1996. **Mycorrhizal landscapes**. http://www.ou.edu/cas/botany-micro/ben/ben145.html

McCune, B.; Geiser, L. 1997. **Macrolichens of the Pacific Northwest**. Corvallis, OR: Oregon State University Press; 386 p.

Molina, Randy; Massicotte, Hugues; Trappe, James M. 1992. **Specificity phenomena in mycorrhizal symbioses: community-ecological consequences and practical applications.** In: Allen, M.F., ed. Mycorrhizal functioning: an integrative plant fungus process. New York: Chapman and Hall; 358-423.

Munz, P.A.; Keck, D.D. 1959. **A California flora and supplement 1968**. Berkeley, CA: University of California Press; 1681 p.

Nauertz, E.A. 1999. **Impact of various silvicultural practices on the abundance and frequency of *Lycopodium* species in northern hardwood forests**. Houghton, MI: Michigan Technological University; 164 p. M.S. thesis.

Nauertz, E.A. 2002. **Lycopodium management**. USDA Forest Service and Colleen Matula, Wisconsin Department of Natural Resources. http://www.extension.umn.edu/specializations/environment/components/lycopodium1.html

Neitlich, P.N.; McCune, B. 1997. **Hotspots of epiphytic lichen diversity in two young managed forests**. Conservation Biology 11(1): 172-182.

Noss, R.F.; Strittholt, J.R.; Heilman, Gerald G.E., Jr.; Frost, P.A.; Sorensen, M. 2000. **Conservation planning in the redwoods region**. In: Noss, Reed F., ed. The redwood forest: history, ecology, and conservation of the coast redwoods. Washington DC: Island Press; 201-228.

Parsons, L. 2003. **Western Erythronium species from seed**. http://www.peak.org/~parsont/rockgard/erythrseed.htm.

Pickart, A.J.; Hiss, A.E.; Enberg, A.W. 1992. **Return from extinction: recovery of the Humboldt milk-vetch**. In: Proceedings of the symposium on biodiversity of northwest California; 1991 October 28-30; Santa Rosa, CA.

Reed, P.B., Jr. 1988. **National list of plant species that occur in wetlands: California (Region 0)**. Biol. Rep. 88 (26.10). Washington DC: USDI Fish and Wildlife Service; 135 p.

Rice, K. 2000. Professor of Agronomy and Range Science, University of California at Davis. [Letter to Mary A. Showers] in Showers, Mary A. 2002. **Report to the Fish and Game Commission on the status of the North Coast Semaphore Grass *Pleuropogon hooverianus***: California Department of Fish and Game; 49 p.

Schmidt, M.G. 1980. **Growing California native plants**. Berkeley, CA: University of California Press; 366 p.

Sholars, T.A. 2004. **Searching for rare plants through niche definition**. In: Proceedings of the 2002 symposium of the ecology and management of rare plants of northwestern California; 2002 February 6-8; Arcata, CA. Calif. Native Plant Society; 73-76.

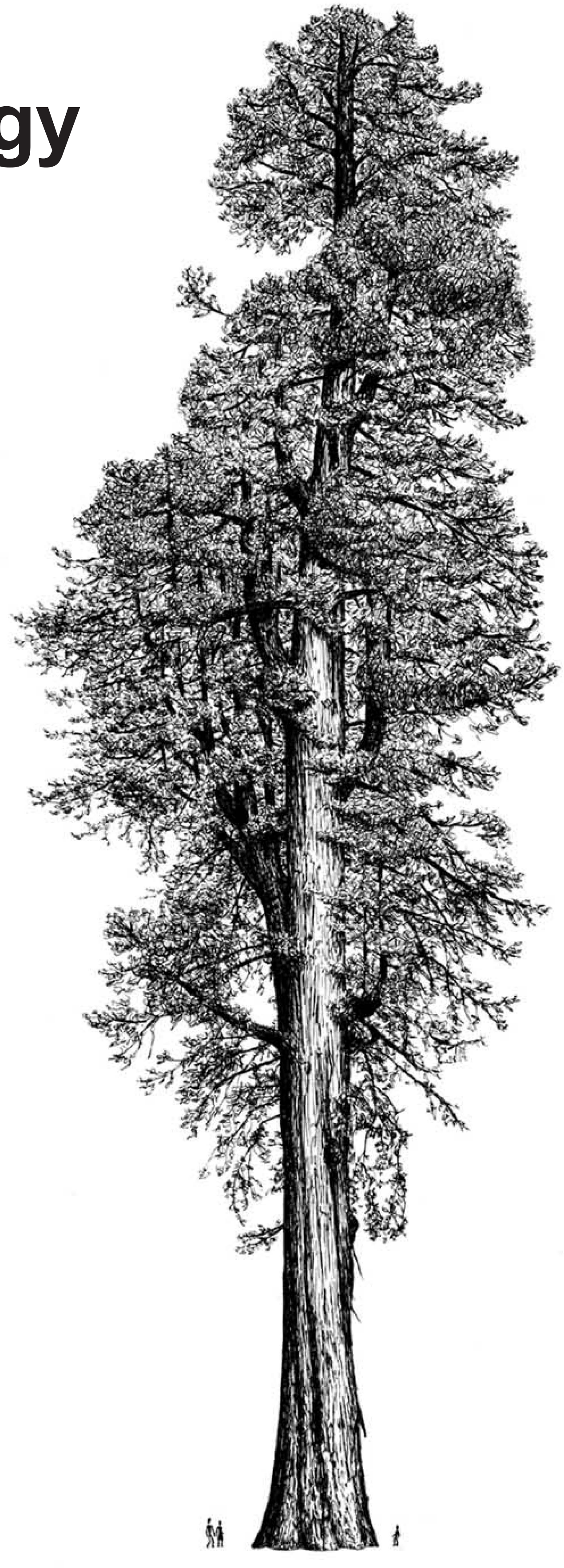
Showers, M.A. 2002. **Report to the Fish and Game Commission on the status of the North Coast Semaphore Grass *Pleuropogon hooverianus***. Sacramento, CA: California Department of Fish and Game; 49 p.

Smith, G.L.; Wheeler, C.R. 1992. **A flora of the vascular plants of Mendocino County, California**. San Francisco, CA: University of San Francisco; 367 p.

Whittier, D.P. 1998. **Germination of spores of the Lycopodiaceae in axenic culture**. American Fern Journal 88(3): 106-113.

SESSION 5

Forest Ecology



Environmental Control of Microbial N Transformations in Redwood Forests[1]

Damon Bradbury and Mary Firestone[2]

Scenarios of global change predict alterations in patterns of temperature, rainfall and oceanic fog impacting continental margins. The response of redwood *(Sequoia sempervirens)* forests to climatic changes will be, in part, predicated on the response of soil N dynamics to the altered environment. Our research focuses on how patterns in climate affect microbial community composition and rates of soil N transformations in redwood forests. Rates of gross N-mineralization are a primary controller of N availability to the plants in these ecosystems and rates of nitrification, denitrification, and nitrous oxide production control important N-loss mechanisms from these episodically wet soils. Rates of all of these N-cycling processes were determined in soils from two northern California redwood forests, incubated at field and lab temperatures under aerobic and anaerobic conditions. Gross rates of nitrification were high in these redwood forest soils, a result contrary to older studies that quantified simple rates of nitrate accumulation. High rates of gross nitrification under anaerobic conditions suggest the possibility of heterotrophic nitrification. Although higher temperature incubations did generally tend to increase the rates of all processes, the impacts of warming on the anaerobic processes of denitrification and dissimilatorynitrate reduction to ammonium (DNRA) suggest unanticipated interactions between temperature and water in controlling these important anaerobic processes. Warmer temperatures increased DNRA more than denitrification suggesting that increasing temperatures may promote nitrogen conservation by DNRA in these wet forests. We are currently exploring the impacts of climate on the abundance, composition, and activity of the nitrifying and denitrifying communities in old-growth redwood forests. The relationship between microbial community composition and rates of soil N transformations may provide insights that help predict the response of the N dynamics in these forests to future changes in temperature and precipitation.

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] University of California at Berkeley, Department of Environmental Science Policy and Management, Ecosystem Sciences Division, 151 Hilgard Hall, Berkeley, CA 94720, (510) 642-3677. email: mkfstone@nature.berkeley.edu

Redwood Trees, Fog Water Subsidies, and the Hydrology of Redwood Forests[1]

Todd Dawson,[2] Stephen Burgess,[3] Kevin Simonin,[2] Emily Limm,[2] and Anthony Ambrose[2]

Fog is a defining feature of the coastal California redwood forest and fog inputs via canopy drip in summer can constitute between 10 to 45 percent of the total water input each year. Furthermore, between four to eight percent of fog-water can be directly absorbed by the tree crowns following heavy fog events. Site-to-site and inter-annual variation in fog inputs is significant and has a marked influence on a host of tree and forest processes. Together, root water uptake from canopy drip and direct fog uptake by foliage has a significant and positive influence on several aspects of redwood tree water relations including the degree of water stress and how water stress determines tree distribution, growth, size as well as stand water balance. How the spatial and temporal variation and dynamics of fog inputs influences hydrological processes from the stand to the region scale is an issue in need of further research; it is a neglected, yet we believe pivotal part of redwood forest ecology and hydrology. Our goal is to use our ecological and physiological knowledge of plant and stand water relations to inform research and eventually management issues linked to the hydrology of the redwood region. We believe that armed with some basic understanding of redwood's water requirements and water use patterns, particularly in relation to the water subsidies provided by fog in summer, that more sound and sustainable water resource management policies could be achieved. As we look to the future of the redwood region, how water resource issues are dealt with must be part of any successful and sustainable management framework. We propose a framework based on the research we are currently involved in.

---

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Department of Integrative Biology, Valley Life Science Building, University of California, Berkeley, CA. email: tdawson@berkeley.edu
[3] School of Plant Biology, University of Western Australia, 35 Stirling Highway, Nedlands WA 6009, Australia.

Status of Vegetation Classification in Redwood Ecosystems[1]

Thomas M. Mahony[2] and John D. Stuart[3]

Abstract

Vegetation classifications, based primarily on physiognomic variability and canopy dominants and derived principally from remotely sensed imagery, have been completed for the entire redwood range (Eyre 1980, Fox 1989). However, systematic, quantitative, floristic-based vegetation classifications in old-growth redwood forests have not been completed for large portions of redwood's range. Quantitative classifications have been completed in portions of the northern (Atzet and Wheeler 1984, Becking 1967, Jimerson and Jones 2000, Lenihan 1986, Mahony 1999a, Mahony and Stuart 2000, Matthews 1986, Mendonca[4]), and extreme southern portion of redwood's range (Becking 1971, Borchert and others 1988). With the exception of very limited sampling in Marin and San Mateo Counties by Keeler-Wolf and others (2003), no quantitative classifications have been published for the central and much of the southern portion of redwood's range. Vegetation classifications are most useful if classification units are spatially represented on maps, but this is often difficult using standard remote sensing techniques. Vegetation modeling is one technique which may allow vegetation classification units to be mapped. Mahony (1999b) developed a predictive model for a portion of the northern redwood range based on a quantitative, floristic-based vegetation classification, discriminant analysis, and a Geographic Information System. Based on field verification, the model was 75 percent accurate.

Key words: old-growth, redwood, vegetation classification, vegetation modeling

Introduction

Redwood5 forests are endemic to a narrow coastal belt, one to fifty kilometers wide, from central California to southwestern Oregon (Fox 1989, Griffin and Critchfield 1972, Roy 1966). Temperature, precipitation, substrate, and other abiotic variables differ throughout redwood's range, producing tremendous diversity in species composition, stand structure, and stand dynamics.

Vegetation classification is a tool used to simplify highly complex natural ecosystems into repeatable, relatively homogeneous vegetation units for the purpose of management and communication. While there are many ways to describe and classify vegetation, systematic, quantitative, floristic-based classifications are most

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Plant Ecologist, Albion Environmental, Inc., 1414 Soquel Avenue, Suite 205, Santa Cruz, CA 95062. email: tmahony@albionenvironmental.com
[3] Professor, Department of Forestry and Watershed Management, Humboldt State University, Arcata, CA 95521. email: jds2@humboldt.edu
[4] Unpublished data on file, Humboldt State University, Arcata, California.
[5] Common names are used throughout the text. Corresponding scientific nomenclature follows Hickman (1993) and is included as Appendix A.

effective for vegetation conservation and management (Sawyer and Keeler-Wolf 1995). Quantitative, floristic-based classifications, such as those used by the California Native Plant Society (Sawyer and Keeler-Wolf 1995) and the California Department of Fish and Game (California Department of Fish and Game 2003), classify vegetation based on the dominant overstory species (series or alliance), in conjunction with the dominant understory species (association).

Despite the importance of classification to understanding and managing redwood ecosystems, large portions of the redwood range have not been quantitatively classified and described. Most redwood forests have been logged (Evarts and Popper 2001, Fox 1989, Sawyer and others 2000). Classification of remaining old-growth redwood is particularly important because old-growth forests are in greater equilibrium with site variables compared to rapidly changing logged forests, and represent conditions generally present prior to large scale human disturbance. Information gleaned from old-growth classification can, in addition to assisting in management and conservation of old-growth forests themselves, provide baseline data essential to restoring logged redwood ecosystems to old-growth conditions.

Classifications Conducted in Redwood Ecosystems

Vegetation classifications, based primarily on physiognomic variability and canopy dominants and derived principally from remotely sensed imagery, have been completed for the entire redwood range (Eyre 1980, Fox 1989). These efforts are crucial to understanding the geographic extent, canopy physiognomy, and ecological dynamics of redwood forests at coarse scales. However, systematic, quantitative, floristic-based vegetation classifications in old-growth redwood forests have not been completed for large portions of redwood's range.

Quantitative, floristic-based classifications have been completed in portions of the northern (Atzet and Wheeler 1984, Becking 1967, Jimerson and Jones 2000, Lenihan 1986, Mahony 1999a, Mahony and Stuart 2000, Matthews 1986, Mendonca[6]), and extreme southern portion of redwood's range (Becking 1971, Borchert and others 1988). With the exception of very limited sampling in Marin and San Mateo Counties by Keeler-Wolf and others (2003), no redwood classifications have been published for the central and much of the southern portion of the range. The three redwood range divisions described here generally follow Sawyer and others (2000).

Based on completed classifications, it is clear that redwood forests are highly variable in terms of stand structure, floristic composition, and ecosystem dynamics. Redwood forests, therefore, are not a uniform vegetation type but a highly diverse assemblage of forest associations that, with the exception of redwood trees themselves, have few floristic and structural attributes in common. This variability underscores the importance of classifying all redwood stands, particularly old-growth stands, in order to develop a complete catalogue of redwood forest associations occurring throughout the range. Moisture availability, represented by abiotic variables such as elevation, topographic position, aspect, slope angle, and coastal proximity, appears to be a dominant factor determining composition of redwood

[6] Unpublished data on file, Humboldt State University, Arcata, California.

forests both within a particular region and throughout the range as a whole.

Northern Range

Northern redwood forests are the most extensively classified redwood forests. Becking (1967) classified redwood into alliances (=series): the Redwood-redwood sorrel alliance and Redwood-sword fern alliance. He indicated that the moist Redwood-redwood sorrel alliance occurred on lower slopes and alluvial flats, and the drier Redwood-sword fern alliance was generally found on middle and upper slopes and ridges.

Lenihan (1986) described three old-growth redwood associations occurring along a moisture gradient in the Little Lost Man Creek Research Natural Area in Redwood National Park, California: Redwood/deer fern, Redwood/little Oregon-grape, and Redwood/madrone. He considered the associations as moist, mesic, and xeric habitats, respectively.

Matthews (1986) described four series, five associations, and one phase in the Bull Creek watershed, Humboldt Redwoods State Park, California: Redwood/redwood sorrel, Redwood-Douglas-fir/salal, Redwood-Douglas-fir/evergreen huckleberry, Redwood-Douglas-fir/madrone, and Douglas-fir/tanoak-madrone. He suggested these associations were arranged along a complex environmental gradient controlled primarily by soil moisture.

Mahony (1999a) and Mahony and Stuart (2000) classified and described six associations within the redwood series throughout the northern range of redwood in Jedediah Smith Redwoods State Park, Del Norte Coast Redwoods State Park, and northern Redwood National Park in northwestern California, and the Siskiyou National Forest in southwestern Oregon: Redwood/sword fern, Redwood-Douglas-fir/rhododendron, Redwood-western hemlock/evergreen huckleberry, Redwood-western hemlock/sword fern, Redwood-western hemlock/salmonberry, and Redwood-red alder/salmonberry. They suggested that moisture was the primary environmental variable controlling the distribution of forest associations. Discriminant analysis revealed that the physiographic factors influencing floristic associations, in decreasing order of importance, were elevation, coastal proximity, and a combination of aspect and topographic position called a Moisture Equivalency Index.

Jimerson and Jones (2000) described ten redwood associations in both old-growth and logged portions of the Headwaters Forest Reserve in central Humboldt County: Redwood-tanoak/evergreen huckleberry-salal, Redwood-tanoak/sword fern, Redwood-Douglas-fir/salal-evergreen huckleberry, Redwood-Douglas-fir/sword fern, Redwood-red alder/salmonberry, Redwood-western hemlock/evergreen huckleberry-salal, Redwood-western hemlock/salmonberry/sword fern, Redwood-western red-cedar/sword fern, Redwood-Sitka spruce/thimbleberry, and Redwood-grand fir/salal/sword fern. Redwood associations in the Reserve were correlated with abiotic factors generally indicative of moisture availability, such as elevation, aspect, slope angle, slope position, and coastal proximity, with elevation the strongest correlated variable.

Mendonca[7] described four old-growth forest associations and two cutover forest associations in the forested portion of Prairie Creek Redwoods State Park in northern Humboldt and southern Del Norte Counties. The old-growth forest associations are: Redwood/sword fern-evergreen huckleberry-redwood sorrel, Redwood-Douglas-fir/tanoak-rhododendron/evergreen huckleberry-sword fern, Redwood-western hem-

[7] Unpublished data on file, Humboldt State University, Arcata, California

lock-Sitka spruce/sword fern, and Red alder-Sitka spruce/salmonberry-sword fern. The associations described for cutover stands are: Sitka spruce/sword fern-salmonberry and Redwood-Sitka spruce/evergreen huckleberry-sword fern. The distribution of associations throughout the park was correlated with logging disturbance and abiotic variables such as elevation, topographic position, aspect, and coastal proximity.

Central Range

Keeler-Wolf and others (2003) described two associations in the redwood alliance (=series) in portions of Marin and San Mateo Counties based on thirteen plots: Redwood/tanoak/evergreen huckleberry and Redwood-Douglas-fir-California bay. The associations occurred throughout a variety of environmental conditions, and most showed signs of past logging disturbance.

Southern Range

Becking (1971) sampled 26 plots in Monterey County, California, which he placed in the Redwood-tanoak-Coulter pine association and the Redwood-tanoak-redwood sorrel association. He suggested that the redwood forest distribution in the area was primarily a result of soil moisture, with the former association occurring on drier sites and the latter on moister sites.

Borchert and others (1988) classified redwood in the southern part of the range in southern Monterey County, California using the USDA Forest Service ecological type classification system. They devised six associations ("ecological types") using biophysical factors such as landform and soil series: Redwood/western bracken-chain fern//Streamsides, Redwood/sword fern-western trillium//Gamboa-Sur, Redwood//Gamboa-Sur, Redwood/common man-root-common vetch//Gamboa-Sur, Redwood-bigleaf maple/California polypody//Gamboa, and Redwood-tanoak/round fruited carex-Douglas iris//Gamboa.

Vegetation Classification Modeling

Vegetation classifications are most useful if classification units are represented on maps, since management and conservation activities can then be spatially correlated and quantified. While vegetation maps exist for many redwood forests, the vegetation mapping units are not necessarily related to classification units. Vegetation maps depict visually distinct groupings of plants at some scale, but a vegetation classification lacks scale and can involve much more complex floristic or structural detail compared to what can be effectively mapped. However, the best maps are presented in the form of a classification (Sawyer and Keeler-Wolf 1995, Sawyer and others 2000), and vice-versa.

Classification units, particularly at the association level, can be difficult to map in redwood forests because: (1) classification unit boundaries are often diffuse and spatially patchy; (2) classification units may be too small to be shown at a designated map resolution; and (3) floristic variability at the series (=alliance) level is often difficult to identify from remotely sensed imagery (Sawyer and others 2000), and virtually impossible to identify at the association level in redwood forests due to dense canopy cover. Since much redwood forest variability is found at the association level, the ability to map associations can be extremely beneficial for redwood conservation and management.

Vegetation modeling is one technique which may allow vegetation classification units to be mapped across the landscape. Predictive vegetation modeling using quantitative vegetation classification and Geographic Information System (GIS) technologies is emerging as an effective method of mapping and predicting vegetation at landscape scales (Brzeziecki and others 1993; Franklin 1995, 1998; Lowell 1991).

Mahony (1999b) developed a prediction model for a portion of the northern redwood range in Jedediah Smith Redwoods State Park in Del Norte County based on a quantitative floristic classification developed by Mahony (1999a) and Mahony and Stuart (2000). The classification described three old-growth redwood forest associations in the park, and discriminant analysis correlated each forest association with abiotic variables, including elevation, coastal proximity, slope position, and aspect. Discriminant analysis is a statistical technique assessing differences between two or more groups and a set of discriminating variables (Klecka 1980). The eigenvalue (expressed in individual percentages relative to total percent) indicates how much discriminatory power each function has.

The prediction model used the following assumptions: (1) all old-growth redwood vegetation in the park consists of one of three classified associations; (2) each association has distinct correlations with the abiotic variables of elevation, coastal proximity, slope, and topographic position/aspect (in the form of a moisture equivalency index (MEI), an index of soil moisture); (3) abiotic variables can be identified on a ten-meter Digital Elevation Model (DEM); and (4) equations can be developed predicting the most likely association occupying each DEM pixel.

Equations were developed that predicted, based on an association's affinity for each abiotic variable derived from discriminant analysis, which pixel in a DEM had the strongest correlation with a particular association. After the equations were run in Arc/Info GRID, the association with the highest score for a particular pixel (the higher the score, the greater the affinity for abiotic variables found in the pixel) was the association to which the pixel was classified. With each pixel classified, a vector map was created depicting predicted associations as polygons.

To determine model accuracy, the map was field verified. Based on 296 ground samples, the map was determined to be 75 percent accurate. Using similar methods, additional models were developed for other portions of Redwood National and State Parks, with similar, though slightly less accurate, results.

While vegetation modeling has limitations, it offers a technique to depict classification units in mapped form. When classification units are mapped, they can be quantified and defined for the purposes of conservation planning and resource management. Prediction models developed over the entire redwood range can help resource managers, scientists, and conservation professionals quantify, compare, and conserve the diversity of redwood ecosystems found throughout the entire range. Since modeling predicts potential vegetation (Brzeziecki and others 1993), it can be used for second-growth management to approximate old-growth forest associations which existed prior to harvesting, and subsequently guide restoration efforts. Finally, since wildlife habitat modeling is essentially identical to vegetation modeling (Franklin 1995), wildlife managers can utilize forest association models to predict suitable wildlife habitat.

Conclusions and Recommendations

Vegetation classification offers important baseline information that can be used to understand, conserve, and manage redwood forests. Quantitative, floristic-based classifications have been completed in portions, most notably the northern part, of redwood's range, but large sections of the range have not been adequately classified and described. Unclassified old-growth redwood forests should be prioritized for quantitative, floristic-based classification in order to develop a complete redwood forest baseline. Classification information can be incorporated into vegetative models to map existing old-growth forest associations, as well as approximate pre-logging conditions in cutover forests to aid in restoration and management.

References

Atzet, T.; Wheeler, D.L. 1984. **Preliminary plant associations of the Siskiyou Mountain Province**. Portland, OR: Pacific Northwest Region, Forest Service, U.S. Department of Agriculture; 315 p.

Becking, R.W. 1967. **The ecology of the coastal redwood forest and the impact of the 1964 floods upon redwood vegetation**. Unpublished Final Report. NSF Grant GB No. 3468; 91 p.

Becking, R.W. 1971. **The ecology of the coastal redwood forest of northern California and the impact of the 1964 flood upon redwood vegetation**. Unpublished Final Report. NSF Grant GB No. 6310; 158 p.

Borchert, M.; Segotta, D.; Purser, M.D. 1988. **Coast redwood ecological types of southern Monterey County, California**. Res. Paper PSW-107. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 27 p.

Brzeziecki, B.; Kienast, F.; Wildi, O. 1993. **A simulated map of the potential natural forest vegetation of Switzerland**. Journal of Vegetation Science 4: 499-508.

California Department of Fish and Game. 2003. **List of California terrestrial natural communities recognized by the California natural diversity database**. Sacramento, CA: Wildlife and Habitat Data Analysis Branch; 77 p.

Dyrness, C.T.; Franklin, J.F.; Maser, C. 1972. **Wheeler Creek Research Natural Area. Supplement No. 1 to Federal research natural areas in Oregon and Washington: a guidebook for scientists and educators**. Portland, OR: Pacific Northwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 16 p.

Evarts, J.; Popper, M., eds. 2001. **Coast redwood: a natural and cultural history**. Los Olivos, CA: Cachuma Press; 228 p.

Eyre, F.H., ed. 1980. **Forest cover types of the United States and Canada**. Washington, DC: Society of American Foresters; 148 p.

Fox, L. 1989. **A classification, map and volume estimate for the coast redwood forest in California.** Interagency agreement number 8CA52849. Sacramento, CA: The Forest and Rangeland Resources Assessment Program, California Department of Forestry and Fire Protection; 37 p.

Franklin, J. 1995. **Predictive vegetation mapping: geographical modeling of biospatial patterns in relation to environmental gradients**. Progress in Physical Geography 19(4): 474-499.

Franklin, J. 1998. **Predicting the distribution of shrub species in southern California from climate and terrain-derived variables**. Journal of Vegetation Science 9: 733-748.

Griffin, G.R.; Critchfield, W.B. 1972. **The distribution of forest trees in California**. Res. Paper PSW-82. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 118 p.

Hickman, J.C., ed. 1993. **The Jepson manual: higher plants of California**. Berkeley: University of California Press; 1400 p.

Jimerson, T.M.; Jones, J.K. 2000. **Vegetation classification and mapping of the Headwaters Forest Reserve**. Unpublished paper supplied by author.

Keeler-Wolf, T.; San, S.; Schindel, M. 2003. **Classification of the vegetation of Point Reyes National Seashore Golden Gate National Recreation Area, Samuel P. Taylor, Mount Tamalpais, and Tomales State Parks, Marin, San Francisco, and San Mateo Counties, California**. Unpublished paper supplied by author.

Klecka, W.R. 1980. **Discriminant analysis**. London: Sage Publications; 71 p.

Lenihan, J.M. 1986. **The forest associations of the Little Lost Man Creek Research Natural Area, Redwood National Park, CA**. Arcata, CA: Humboldt State University; 85 p. M.S. thesis.

Lowell, K. 1991. **Utilizing discriminant function analysis with a geographical information system to model ecological succession spatially**. International Journal of Geographical Information Systems 5(2): 175-191.

Mahony, T.M. 1999a. **Old–growth forest associations in the northern range of redwood**. Arcata, CA: Humboldt State University; 96 p. M.S. thesis.

Mahony, T.M. 1999b. **Modeling old-growth forest associations in Jedediah Smith Redwoods State Park and northern Redwood National Park**. Unpublished paper supplied by author.

Mahony, T.M.; Stuart, J.D. 2000. **Old-growth forest associations in the northern range of coastal redwood**. Madroño 47(1): 5360.

Matthews, S.C. 1986. **Old-growth forest associations of the Bull Creek watershed Humboldt Redwoods State Park, California**. Arcata, CA: Humboldt State University; 93 p. M.S. thesis.

Roy, D.F. 1966. **Silvicultural characteristics of redwood**. Res. Paper PSW-28. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 20 p.

Sawyer, J.O.; Keeler-Wolf, T. 1995. **A manual of California vegetation**. Sacramento, CA: California Native Plant Society; 470 p.

Sawyer, J.O.; Sillett, S.C.; Popenoe, J.H.; LaBanca, A; Sholars, T.; Largent, D.L.; Euphrat, F.; Noss, R.F.; Van Pelt, R. 2000. **Characteristics of redwood forests**. In: Noss, R.F., ed. The redwood forest, history, ecology, and conservation of the coast redwoods. Covelo, CA: Island Press; 339 p.

Appendix A. *Common and scientific names for species mentioned in the text. Nomenclature for scientific names follows Hickman (1993).*

bigleaf maple	*Acer macrophyllum*
California bay	*Umbellularia californica*
California polypody	*Polypodium californicum*
chain fern	*Woodwardia fimbriata*
common man-root	*Marah fabaceus*
common vetch	*Vicia sativa* ssp. *nigra*
Coulter pine	*Pinus coulteri*
deer fern	*Blechnum spicant*
Douglas-fir	*Pseudotsuga menziesii* var. *menziesii*
Douglas iris	*Iris douglasiana*
evergreen huckleberry	*Vaccinium ovatum*
grand fir	*Abies grandis*
little Oregon grape	*Berberis nervosa*
madrone	*Arbutus menziesii*
red alder	*Alnus rubra*
redwood	*Sequoia sempervirens*
redwood sorrel	*Oxalis oregana*
rhododendron	*Rhododendron macrophyllum*
round fruited carex	*Carex globosa*
salal	*Gaultheria shallon*
salmonberry	*Rubus spectabilis*
Sitka spruce	*Picea sitchensis*
sword fern	*Polystichum munitum*
tanoak	*Lithocarpus densiflorus*
thimbleberry	*Rubus parviflorus*
western bracken	*Pteridium aquilinum* var. *pubescens*
western hemlock	*Tsuga heterophylla*
western redcedar	*Thuja plicata*
western trillium	*Trillium ovatum* ssp. *ovatum*

What Was the Role of Fire in Coast Redwood Forests?[1]

Peter M. Brown[2]

Fire has long been recognized as an important disturbance in coast redwood *(Sequoia sempervirens)* forests (Fritz 1931), but the exact nature of historical fire regimes in many areas is uncertain. Coast redwood grows in relatively mesic, often fog-shrouded coastal locations not usually associated with widespread or frequent fires. Past reviews of coast redwood fire ecology have concluded that over much of its range fires were typically infrequent and that effects on forest composition and structure varied depending primarily on fire severity (for example, Sawyer and others 2000). However, a growing body of evidence from fire scars documents that frequent, episodic surface fires were a dominant fire regime in many coast redwood forests, and that loss of surface fires has occurred over the recent century in response to loss of Native American ignition sources, active fire suppression, and other changes brought about by Euro-American settlement and land use (Brown and Baxter 2003; Brown and others 1999; Brown and Swetnam 1994; Finney 1990; Finney and Martin 1989, 1992). This fire-scar evidence is usually poorly preserved as a result of both the nature of the preservation of fire-scar records on coast redwood trees and past harvest practices—especially broadcast burning of slash—that has tended to remove the record from many locations.

In this talk, I described results from fire history reconstructions in coast redwood and coast redwood/Douglas-fir forests at Point Reyes National Seashore, Jackson State Forest in the Mendocino area, and Redwood National Park. Fire chronologies based mainly on fire scar evidence from these locations span the past two to four centuries, and average pre-20th century intervals between fires varied from seven to 20 years (*table 1, fig. 1)*. Although the tree-ring evidence is fragmentary and—as with all historical reconstructions—an imperfect model of past conditions, these data document that coast redwood forests in these areas experienced fire regimes of frequent, recurrent surface fires. Many of the past fires were likely set by humans because of a general lack of lightning in forests along the coast.

Loss of surface fires from coast redwood forests has led to changes in associated ecosystem patterns and processes that are analogous to those in ponderosa pine and other low-elevation forests of the western North America. Frequent surface fires promote open, low-density stand structure by killing a majority of tree regeneration before it has a chance to reach canopy status. Fire-intolerant species, such as grand fir, western hemlock, and hardwoods, would have had less opportunity to establish between fires and likely were not as abundant in pre-fire exclusion forests as they are today. Biomass of shrubs also would have been reduced because of recurrent mortality during episodic burns. Duff and woody litter would have been reduced, and

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Director, Rocky Mountain Tree-Ring Research, 2901 Moore Lane, Ft. Collins, CO 80526. email: pmb@rmtrr.org

diversity and biomass of grass and herbaceous vegetation likely were correspondingly greater. Increases in fuel loads, tree density, canopy coverage, and formation of ladder fuels result in feedbacks to the fire regime, with the result that crown fires replace surface fires as the dominant fire behavior when fires occur. Although fire exclusion on the California Coast occurred later in the 20th century than many other low elevation forests in North America, coast redwood forests are often nearly twice as productive as pine or mixed-conifer forests of the central mountains or Sierra Nevada in California. Because of higher productivity in these relatively warm, wet coastal forests, changes in community composition and fuels structure may be as pronounced as in other forests that experienced earlier loss of surface fire regimes.

Table 1—*Fire frequencies in coast redwood stands at Jackson Demonstration State Forest.*

		Intervals per stand			Intervals per tree	
Site	Period analyzed	MFI ± 1 SD (yr)	WMPI (yr)	Range (yr)	No. of trees	MFI ± 1 SD (yr)
PYG	1700s	-	-	-	-	-
	1750 - 1850	17.4 ± 12.3	15.8	6 - 34	5	24.5 ± 5.9
	1800s	13.2 ± 6.3	12.9	6 - 23	6	18.8 ± 7.7
NSF	1700s	21.8 ± 4.6	22.1	16 - 27	3	21.6 ± 0.1
	1750 - 1850	12.9 ± 6.7	12.4	5 - 23	3	14.4 ± 5.1
	1800s	9.8 ± 4.4	9.6	5 - 19	4	13.0 ± 5.1
TCR	1700s	9.4 ± 7.4	8.3	3 - 28	2	9.9 ± 2.3
	1750 - 1850	9.3 ± 5.3	8.8	3 - 19	3	12.4 ± 0.9
	1800s	10.0 ± 5.2	9.6	4 - 19	3	11.1 ± 0.7
HLT	1700s	7.1 ± 4.9	6.5	3 - 18	8	11.6 ± 6.2
	1750 - 1850	6.0 ± 3.3	5.7	2 - 15	10	13.0 ± 4.9
	1800s	8.6 ± 4.6	8.1	2 - 17	14	14.5 ± 2.6
WTF	1700s	-	-	-	2	14.7 ± 3.3
	1750 - 1850	10.0 ± 3.4	10.1	5 - 15	5	12.7 ± 1.5
	1800s	13.4 ± 2.4	13.6	10 - 17	8	14.9 ± 2.3
CHG	1700s	10.8 ± 3.3	10.9	6 - 16	8	12.2 ± 2.0
	1750 -1850	11.6 ± 4.7	11.5	6 - 21	8	13.7 ± 2.0
	1800s	9.8 ± 3.1	9.8	7 - 14	7	14.1 ± 3.7
DRG	1700s	14.0 ± 5.1	14.0	10 - 22	3	15.0 ± 4.3
	1750 - 1850	15.2 ± 6.2	15.1	9 - 23	4	17.9 ± 3.2
	1800s	15.5 ± 5.0	15.6	9 - 22	5	24.1 ± 7.0

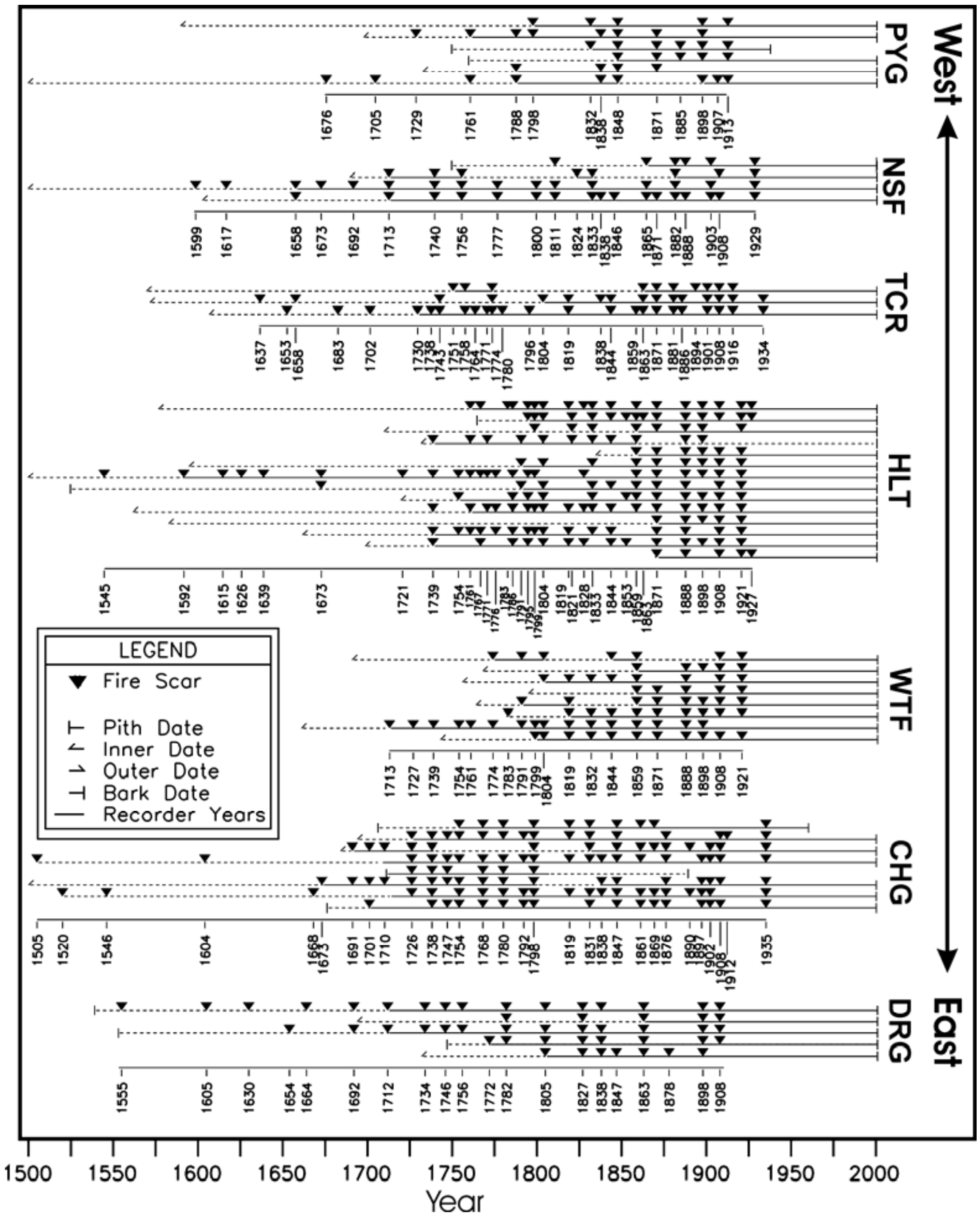


Figure 1—Fire chronologies along an ocean (top) to inland (bottom) gradient at Jackson Demonstration State Forest on the Mendocino Coast (Brown and Baxter 2003). All samples are from coast redwood. Time spans of individual trees are represented by horizontal lines with inverted triangles marking dates of fire scars. Pith or bark dates refer to pith or bark present on cross sections while inside or outside rings refer to cross sections with unknown numbers of rings to pith or bark. Inside dates on several trees extend earlier than 1500. Dashed segments on trees are decayed segments where fire scars are likely missing from the fire-scar record. These segments were not used to calculate point (tree) fire frequencies in *table 1*.

The probable role of Native-Americans in fire ignitions leads to questions concerning the appropriate approach to management of coast redwood ecosystems. In the absence of human ignitions, it is likely that fires would not have been as common, and historical forest composition and structure may have been quite different under such altered fire regimes. Historical data are meant to provide restoration ecologists and land managers with descriptions of ecosystem conditions over long time scales, regardless of whether they were influenced by human or natural factors (for example, Landres and others 1999). How these data are used in management decisions is then as much a societal as it is an ecological question. Do we attempt to restore these forests to historical conditions, or let them return naturally to a fire regime that would have likely been present in the absence of Native American influence?

In my experience, I have found fire-scar records in coast redwood forests to be rare because of poor preservation. But in those locations where fire-scar records have been compiled, fire histories document these forests are not burning today nearly as often as they did in the past. Recognition of changes caused by loss of surface fires has resulted in efforts to restore historical processes in many low-elevation forests throughout western North America. Fire histories reconstructed by these studies provide both guidelines and justification for ecological restoration efforts in coast redwood forests throughout much of its range.

References

Brown, P.M.; Baxter, W.T. 2003. **Fire history in coast redwood forests of the Mendocino Coast.** Northwest Science 77: 147-158.

Brown, P.M.; Kaye, M.W.; Buckley, D. 1999. **Fire history in Douglas-fir and coast redwood forests at Point Reyes National Seashore, California.** Northwest Science 73: 205-216.

Brown, P.M.; Swetnam, T.W. 1994. **A cross-dated fire history from a stand of coast redwood near Redwood National Park, California.** Canadian Journal of Forest Research 24: 21-31.

Finney, M.A. 1990. **Fire history from the redwood forests of Bolinas Ridge and Kent Lake Basin in the Marin Municipal Water District.** Vegetation and Fire Management Baseline Studies: The Marin Municipal Water District and the Marin County Open Space District (Northridge Lands), Marin County, California. Unpublished report on file at the Marin Municipal Water District.

Finney, M.A.; Martin, R.E. 1989. **Fire history in a *Sequoia sempervirens* forest at Salt Point State Park, California.** Canadian Journal of Forest Research 19: 1451-1457.

Finney, M.A.; Martin, R.E. 1992. Short **fire intervals recorded by redwoods at Annadel State Park, California.** Madroño 39: 251-262.

Fritz, E. 1931. **The role of fire in the redwood region.** Journal of Forestry 29: 939-950.

Landres, P.B.; Morgan, P.; Swanson, F.J. 1999. **Overview of natural variability concepts in managing ecological systems.** Ecological Applications 9: 1179-1188.

Sawyer, J.O., Sillett, S.C.; Libby, W.J.; Dawson, T.E.; Popenoe, J.H.; Largent, D.L.; Van Pelt, R.; Veirs, Jr., S.D.; Noss, R.F.; Thornburgh, D.A.; Del Tredici, P. 2000. **Redwood trees, communities, and ecosystems: A closer look.** In: Noss, R.F., ed. The Redwood Forest; Washington, DC: Island Press; 81-118.

Fire History in Coast Redwood Stands in San Mateo County Parks and Jasper Ridge, Santa Cruz Mountains[1]

Scott L. Stephens[2] and Danny L. Fry[3]

Abstract

Fire regimes in coast redwood forests in the northeastern Santa Cruz Mountains were determined by ring counts from 46 coast redwood stumps and live trees. The earliest recorded fire from two live samples was in 1615 and the last fire recorded was in 1884, although samples were not crossdated. For all sites combined, the mean fire return interval (FRI) was 12.0 years; the median FRI was 10 years. There was a significant difference in mean FRI between the four sampled sites. Past fire scars occurred most frequently in the latewood portion of the annual ring or during the dormant period. It is probable that the number of fires recorded in coast redwood trees is a subset of those fires that burned in adjacent grasslands and oak savannahs. The Ohlone and early immigrants were probably the primary source of ignitions in this region.

Key words: fire return interval, seasonality

Introduction

Evidence of past fires is common in California's coast redwood (*Sequoia sempervirens* (D. Don) Endl.) forests. Redwood trees and stumps commonly exhibit fire scars in their annual growth rings, charred bark, and burned-out basal cavities. Recent research has documented the ecological role of fire in coast redwood forests (Brown and Baxter 2003; Brown and others 1999; Brown and Swetnam 1994; Finney and Martin 1989, 1992; Jacobs and others 1985; Stuart 1987).

The objective of this study is to determine the fire history of four coast redwood stands in the northeastern Santa Cruz Mountains of California. This is only the second fire history study done using fire scars in the southern portion of the coast redwood range, the first (Greenlee 1983) was a small study that analyzed samples from two stumps to estimate past fire frequency.

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2]Assistant Professor, Ecosystems Sciences Division, Dept. of Environmental Science, Policy, and Management, 137 Mulford Hall MC#3114, University of California, Berkeley, CA 94720-3114. email: stephens@nature.berkeley.edu

[3]Staff Research Associate, Ecosystems Sciences Division, Dept. of Environmental Science, Policy, and Management, 137 Mulford Hall MC#3114, University of California, Berkeley, CA 94720-3114. email: dfry@nature.berkeley.edu

Methods

Fire scars were collected from the east-side of the northern Santa Cruz Mountains in San Mateo County, California. Three locations were included in this study, Stanford Universities Jasper Ridge Biological Preserve (Jasper Ridge), and two San Mateo County Parks, Huddart and Wunderlich. The three areas were reconnoitered to determine where clusters with fire-scarred materials were located. Each cluster that contained a minimum of seven fire scarred samples over an area ≤5 ha was selected for sampling. Partial cross sections were cut with a chainsaw from all fire-scarred snags, downed logs, and live trees with visible fire scars within the four fire scar sampling areas.

To determine the FRI's, annual rings between successive fire scars were counted for each sample and summarized for samples within each cluster. The position within the ring in which a scar occurred was noted as EE (early earlywood), ME (middle earlywood), LE (late earlywood), LW (latewood), D (dormant), or U (undetermined) to serve as an estimate of the season of fire occurrence (Dieterich and Swetnam 1984).

Results and Discussion

Degradation of remnant materials from decay and post-harvest fires severely limited available fire scars in this region. FRI's were found to be frequent for the period of record (~1500s to 1800s) (*table 1*). For all sites combined, mean FRI was 12.0 years (SEM = 0.5); the median FRI was 10 years. There was a significant difference in mean FRI between the four sites (K-W test statistic = 15.2, p = 0.002) but not between aspects (*table 1*). The earliest recorded fire from two live samples was in 1615 and the last fire recorded by both samples was in 1884 based from ring counts. Despite having several fire scar years coincident between the two samples, we do not consider these dates to be absolute.

Table 1—*Fire history statistics for four coast redwood forests in San Mateo County. Mean values in a column followed by the same letter are not significantly different (p<0.05). SE = standard error of the mean.*

	N samples	General aspect	Average FRI (SE)	Median FRI
Huddart 1	13	NW-NE	13.1^{b} (0.9)	11
Huddart 2	8	NE-S	16.1^{b} (1.9)	11
Wunderlich	13	NE-S	9.3^{a} (0.4)	9
Jasper	12	NW	14.1^{b} (1.4)	10

Comparison of fire frequency in coast redwood forests is confounded by several factors including differing methodologies (ring-count, crossdating, sprout aging), differences in the size of the sampled areas, and if MFI's are computed from intervals between fire scars or the actual differences in calendar dates between scars.

The season of fire occurrence was determined for 50.3 percent of the fire scars. Past fire scars occurred most frequently in the latewood portion of the annual ring or during the dormant period (99 percent). Early growing season (earlywood) fires were very rare (1.7 percent). Precise estimates of the seasonality of past fires in coast redwood forests requires additional cambia phenology studies to characterize tree-ring growth within a year at different locations and elevations (Caprio and Swetnam 1995).

Lightning ignited fires are relatively rare in the coast redwood region because of modest topography and moist conditions that usually accompany lightning storms. The most common ignitions in this forest type were from Native Americans (Brown and Baxter 2003; Brown and others 1999; Brown and Swetnam 1994; Finney and Martin 1989, 1992; Fritz 1931; Stuart 1987). The Ohlone (Costanoan) Indians have lived in this region for thousands of years and they used fire as a management tool (Lewis 1973). Native Americans burned these areas for diverse purposes including increasing the efficiency of food gathering, reducing acorn eating insects, to clear areas for travel, to produce high quality cordage materials, and to increase food production (Blackburn and Anderson 1993, Brown 2001, Lewis 1973). Early ranchers and farmers also burned these areas during the 19th and early 20th centuries.

The first recorded Spanish expedition entered this region on November 6, 1769, when the first Portola expedition camped in Portola Valley. Shortly after this expedition the number of European immigrants increased because of abundant natural resources and easy access from the San Francisco Bay.

Continued development of old-growth and young-growth coast redwood parklands toward prehistoric conditions may be dependent of a fire regime where prescribed burning substitutes for the now-absent aboriginal ignitions.

References

Blackburn, T.; Anderson, M.K. 1993. **Before the wilderness: environmental management by native Californians.** Ballena Press Anthropological Paper No. 40. Menlo Park, CA: Ballena Press; 476 p.

Brown, A.K., ed. 2001. **A description of distant roads, original journals of the first expedition into California, 1769-1770 by Juan Crespí.** Edited and translated by A.K. Brown. San Diego: San Diego State University Press; 890 p.

Brown, P.M.; Baxter, W.T. 1995. **Fire history in coast redwood forests on the Mendocino Coast, California.** Northwest Science 77: 147-158.

Brown, P.M.; Kaye, M.W.; Buckley, D. 1999. **Fire history in Douglas-fir and coast redwood forests at Point Reyes National Seashore, California.** Northwest Science 73: 205-216.

Brown, P.M.; Swetnam, T.W. 1994. **A cross-dated fire history from coast redwood near Redwood National Park, California.** Canadian Journal of Forest Research 24: 21-31.

Caprio, A.C.; Swetnam, T.W. 1995. **Historic fire regimes along an elevational gradient on the west slope of the Sierra Nevada.** In: Brown, J.K.; Mutch, R.W.; Spoon, C.W.; Wakimoto, R.H., technical coordinators. Proceedings of the symposium on fire in wilderness and park management. Gen. Tech. Rep. INT-320. Ogden, UT: Intermountain Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 173-179.

Dieterich, J.H.; Swetnam, T.W. 1984. **Dendrochronology of a fire-scarred ponderosa pine.** Forest Science 30: 238-247.

Finney, M.A.; Martin, R.E. 1989. **Fire history in a *Sequoia sempervirens* forest at Salt Point State Park, California.** Canadian Journal of Forest Research 19: 1451-1457.

Finney, M.A.; Martin, R.E. 1992. **Short-fire intervals recorded by redwoods at Annadel State Park.** Madroño 39: 251-262.

Fritz, E. 1931. **The role of fire in the redwood region.** Journal of Forestry 29: 939-950.

Greenlee, J.M. 1983. **Vegetation, fire history, and fire potential of Big Basin Redwoods State Park, California.** Santa Cruz, CA: University of California; 167 p. Ph.D. dissertation.

Jacobs, D.F.; Cole, D.W.; McBride, J.R. 1985. **Fire history and perpetuation of natural coast redwood ecosystems.** Journal of Forestry 83: 494-497.

Lewis, H.T. 1973. **Patterns of Indian burning in California: ecology and ethnohistory.** Ballena Press Anthropological Paper No. 1. Ramona, CA: Ballena Press.

Stuart, J.D. 1987. **Fire history of an old-growth forest of *Sequoia sempervirens* (Taxodiaceae) in Humboldt Redwoods State Park, California.** Madroño 34: 128-141.

Progression and Behavior of the Canoe Fire in Coast Redwood[1]

Hugh Scanlon[2]

Abstract

Lightning caused fires occur in coast redwood forests, but large fires have been rare since the 1930s. Coast redwood (*Sequoia sempervirens*) is considered fire resistant. In 2003, the Canoe Fire, a lightning fire started in an old-growth redwood stand in Humboldt Redwoods State Park, burned 5,554 hectares (13,774 acres) before it was contained. Fuel characteristics and weather contributed to fire progression. Topography, access, fuel loading and management considerations affected fire suppression tactics and effectiveness. From the old-growth redwood stand, fire progressed into young-growth, producing significant changes in fire behavior and effects. Old-growth redwood stands should be managed to anticipate wildfire impacts.

Key words: fire, fire behavior, redwood

Introduction

On September 3, 2003, a major lightning event over northwestern California ignited numerous fires. The Canoe fire, starting in Humboldt Redwoods State Park, became the largest recorded fire in old-growth coast redwood *(Sequoia sempervirens)* since the beginning of the fire suppression era. The fire proved difficult to contain due to weather, poor access, fuel conditions, other fire activity, and park management considerations. The Canoe fire presents a case study of fire progression and behavior in mature and young-growth coast redwood.

Coast redwood is considered a fire resistant species, but not dependent on fire for regeneration. Since implementation of fire suppression policies in the 1930s, large fires in redwood have been rare *(fig. 1)*. The last recorded large fires in coast redwood were Comptche in 1931 at 13,064 hectares (32,283 ac), Will Creek in 1945 at 11,672 hectares (28,841 ac), Strong Mountain in 1950 at 7,522 hectares (18,588 ac), and A-Line in 1936 at 5,583 hectares (13,797 ac) (California Department of Forestry and Fire Protection 2003). These fires occurred primarily in young-growth redwood. The Canoe fire in 2003 started in old-growth redwood and burned 5,554 hectares (13,774 acres), becoming the fifth largest fire on record in this vegetation type.

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California

[2] Battalion Chief, California Department of Forestry and Fire Protection, Humboldt, Del Norte Unit, 118 Fortuna Blvd., Fortuna, CA 95540. email: Hugh.Scanlon@fire.ca.gov

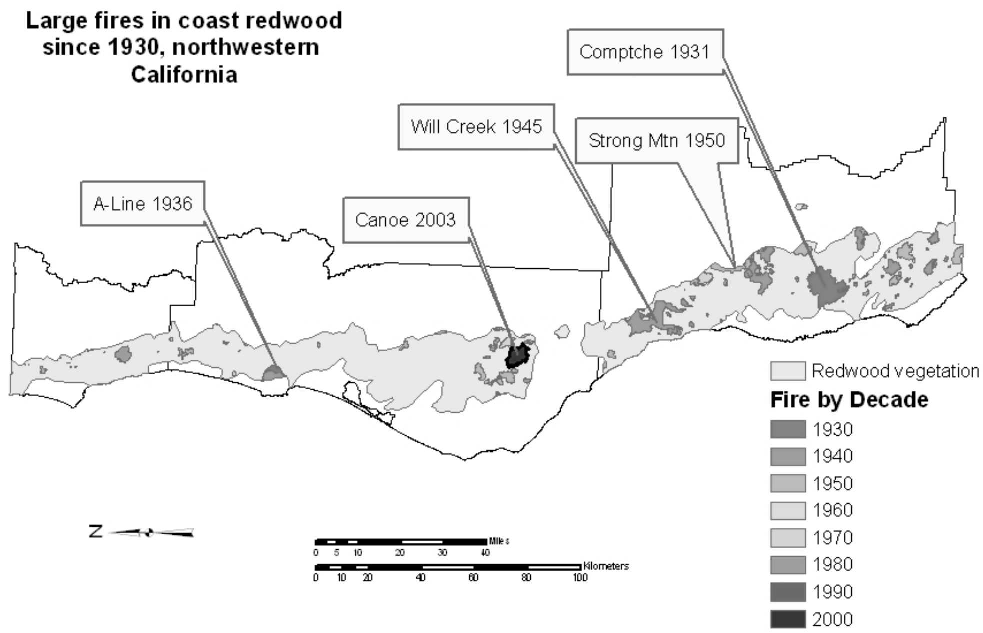


Figure 1—Large fire locations in coast redwood since 1930.

Before the era of fire suppression, coast redwood forests burned often. Mean fire return interval in the pre-settlement period for redwood in Humboldt Redwoods State Park was measured as 24.6 years, with an estimated fire cycle ranging from 26.2 to 51.6 years (Stuart 1987). Fires burning in old-growth redwood produced mortality in approximately two percent of trees over four-foot in diameter (Fritz 1932).

Research on fire behavior in coast redwood has been limited. Young-growth redwood is resistant to being killed in low intensity prescribed burning, but increased mortality was likely when large amounts of fuel were present and burned (Finney and Martin 1993). At Redwood National Park, Nives (1989) found fuel moisture, influenced by elevation and topographic position, a determining factor in fire spread rates, with fuel depth and load affecting the intensity and duration of the fires.

Methods

This is a qualitative evaluation of the fire progression and behavior, primarily based upon fire information found in the documentation of daily firefighting operations for the Canoe fire. The old-growth redwood areas burned were generally inaccessible and precluded using quantitative methods during active burning. Weather information was obtained by reviewing forecasts, data records from Remote Automated Weather Stations (RAWS) near the fire, and in discussion with meteorologists that analyzed weather conditions. Fuel information was gathered from limited sampling in the area and fuel moisture modeling in the Weather Information Management System (WIMS).

Results

Weather

The occurrence of lightning in northwestern California during fire season is infrequent, but not considered rare. The years 1987, 1990, 1994, 1999, 2002, and 2003 experienced major lightning fires in this area.

By late August 2003, northwestern California had warm and very dry conditions, provided by strong high pressure over the western states and a surface thermal trough along the coast. Dry, offshore winds reduced daily minimum relative humidity regularly into the lower to mid teens to set up a critical fire weather pattern for northern California. The Keetch-Byram Drought Index forecast for Eel River Camp was 763.

Early on September 3, a band of cumulonimbus rapidly formed and generated numerous lightning strikes in Lake and Mendocino counties. As the band tracked northwest through and exited western Humboldt, the county was peppered with lightning strikes. Sixty-four new fires were reported. Rainfall ranged from zero precipitation at Eel River Camp and Cooskie Mountain to 0.08 inches at the Alderpoint and Honeydew RAWS. After the Canoe fire started, a series of weak systems and strong highs affected the fire area. Morning inversions were common.

Vegetation, Terrain and Fuel Condition

The Canoe fire started approximately mid-slope in the Canoe Creek watershed of Humboldt Redwoods State Park *(fig. 2)*. This is an area of old-growth redwood, west of the south fork of the Eel River. The area had been designated state wilderness in 2001.

Figure 2—Canoe fire origin and area vegetation.

Humboldt Redwoods State Park was established in 1920, through land acquisition by the Save the Redwoods League. This area is primarily old-growth redwood, but some areas of pre-acquisition selection harvest are scattered in this area. Other areas were in young-growth timber, with few residual old-growth present. The area had some ridge roads. Old skid trails were few, and had reverted to heavy vegetation. Other roads had become trails within the park. The area where the Canoe fire started had no ground access, except by off-trail travel.

Live fuel information forecast by the WIMS using the Eel River Camp RAWS predicted live fuel moisture at 70 percent for September 2. On August 13, live fuel moisture in the Miranda area reached 100.1 percent in manzanita, drier than the average of 108 percent for early August for the years 1995 to 2002. On October 9, live fuel moisture had dipped to 78.6 percent in manzanita, well below the seasonal average of 88.6 percent. Dead fuel moisture for the 10-hour size class dipped to one percent on September 2, and then rising rapidly after the thunderstorm passed. The daily 10-hour fuel moisture lows did not again reach below four percent until September 13. The large dead fuel moisture ranged from 11 to 14 percent during the fire. No quantitative analysis of fuel loading had been done in this area. The area had a heavy fuel load, and no past fires were documented for this area.

Sequence of Events

The morning of September 3, a lightning storm tracked in from the southeast of Humboldt County, moving northwest and out to the ocean near the mouth of the Klamath River. This started the Canoe fire, which was detected later that afternoon. For this area, the Humboldt–Del Norte Unit of the California Department of Forestry and Fire Protection (CDF) is responsible for wildland fire suppression.

Initially, fire behavior was estimated from aerial reconnaissance, fire expansion, and ground-based lookouts. Light smoke and flame lengths of 15 to 30 centimeters (six inches to one ft) were observed, with no torching of trees. The rate of spread remained slow, up to about six m per hour (20 ft/hr) for surface fire during the peak burning periods.

During the first few days, the weather was cool, relative humidity high, and winds were light. Fire progression was slow in the first two weeks *(fig. 3* and *4)*. From its ignition on September 3 until fire crews made access on September 10, the only fire suppression activity came from water drops by helicopter. This effort was limited to a few hours per day.

Access to the Canoe fire was subject to several constraints. It had to be sufficient for supporting the firefighters with food, water, equipment, and emergency evacuation. The fire was located in a state wilderness area, which also limited options. While not subject to the strict constraints of federal wilderness, the State Park needed to agree on suppression strategy if mechanical equipment was used or large trees felled.

Fire crews unsuccessfully attempted creating two different hiking routes into the fire. These routes would take several hours to hike and were not sufficient for firefighter safety and support. On September 9, a bulldozer was walked in along a ridgeline for several miles to construct a helispot near the fire. Weather had become overcast with light rain, halting fire advance and moderating fire behavior. Crews began trail construction to the fire from the helispot. On September 10, seven days after it started, ground resources arrived at the Canoe fire, now at 21 hectares (51 ac).

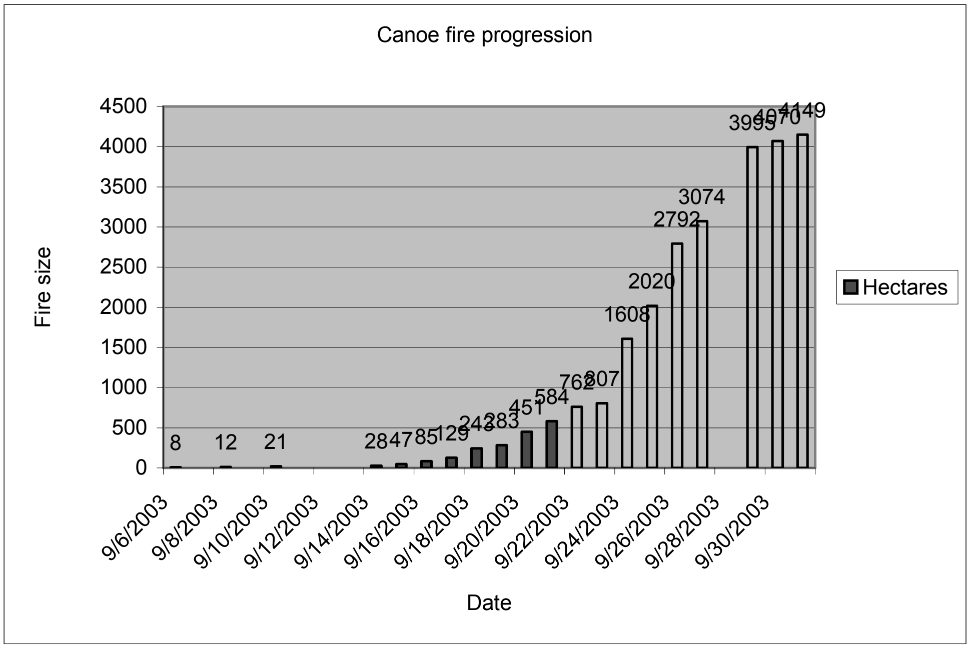


Figure 3—Canoe fire area growth.

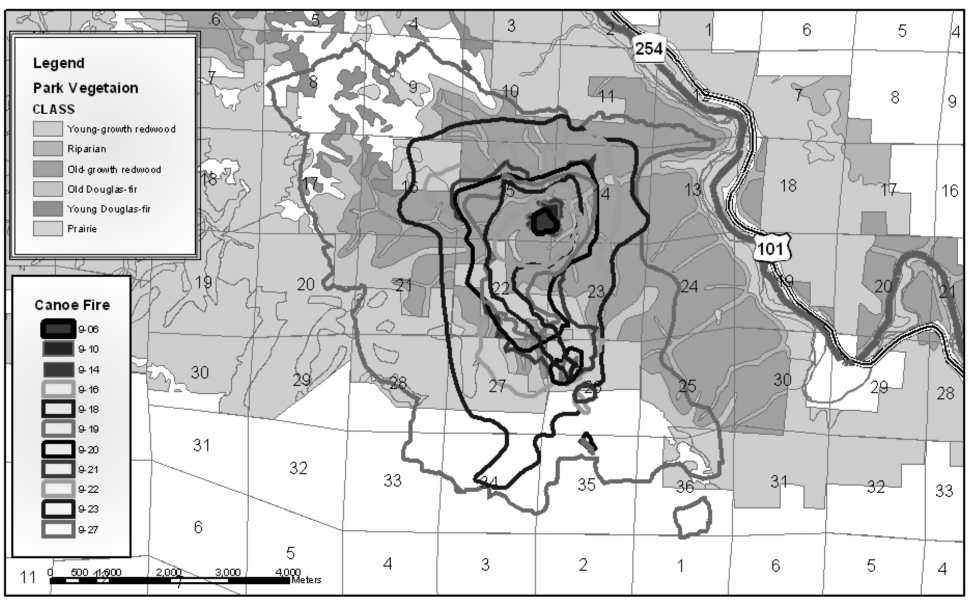


Figure 4—Canoe fire perimeters.

Red flag warnings were in effect for high winds and low humidity on September 11, 12, and 13. Fire growth remained slow, increasing to 28 hectares (69 ac) by September 14. Fire behavior was described as flame lengths of 15 to 30 centimeters (six inches to one ft) and a spread rate of 4.9 m per hour (16 ft/hr). However, active burning periods were occurring both in the afternoon and in the early morning hours. As inversions settled into the Eel River valley, a thermal belt of warm, dry air influenced the fire on the mid and upper slopes, increasing fire activity at night. This was a common occurrence until the fire was contained.

The old-growth area had many standing dead trees prone to falling as the fire burned. The fire moved slowly, but had a long residency time as it consumed deep fuel layers. Given the heights of these trees and the long burn out times, working near the active fireline posed a significant risk to firefighters.

On September 16, several spot fires crossed the west flank control line. Short range spotting was becoming the main factor in fire growth. Crews were unable to establish control lines around all the spot fires before nightfall. After dark, the snags posed too great a hazard to continue work. Control line construction directly on the fire edge was no longer an option on the Canoe fire.

Suppression strategy changed on September 17 to indirect control methods, using the Grasshopper Trail, Grasshopper and Greig Roads, and fire line construction on the southern boundary of the park. The fire progressed southward up Canoe Creek past the helispot, actively burning in young-growth redwood stands by September 18.

On September 19, the Canoe fire had burned 283 hectares (700 ac), growing by 34 hectares (85 ac) in five hours. Relative humidity dropped into the teens and gusty northeast winds were forecast for the fire area. As the fire remained on the lower slopes and in dense forest, wind effects were negligible. But on ridgelines, swirling winds caused spot fires well ahead of the main fire.

The fire growth came from a combination of slow expansion of the main fire, firefighters conducting burnouts along the southern constructed fire line, and spot fires crossing the southern control line. The State Park boundary/control line followed the ridge and swirling winds pumped embers onto private land, creating numerous spot fires. By September 20, acreage growth was more influenced by suppression activity than the fire itself.

The fire had now established in young-growth redwood areas (estimated age 70 years). Fire behavior in these areas was substantially different than observed in old-growth redwood. In both areas, fire would climb well up the tree boles. In old-growth, some of this burning material would fall and establish new fire just ahead of the flaming front. This form of short range spotting was common. In young-growth redwood, the crown of individual trees burning (torching) lofted embers farther distances and was source of short, medium, and long-range spot fires.

Discussion

Fuel characteristics and weather contributed to fire progression. The fire was weather caused, but the fire advance was fuel driven.

The Canoe fire displayed different fire behavior in old-growth redwood than observed in young-growth areas. Observed fire front advance in old-growth areas exhibited flame lengths of up to 0.3 m (one foot) and very slow rates of spread, up to 6 m per hour (20 ft/hr). Adjusting for slope and wind factors, these characteristics are considerably lower than predicted by the BEHAVE model for the most closely matched standard fuel model, Fuel Model 10 (Anderson 1982, Andrews 1986). Predicted flame lengths of 1.5 m (five ft) and maximum rates of spread of 152 m per hour (500 ft/hr) were common from the model. Heavy fuel loads produced long duration fire intensity within the burn. While torching of the tall, living trees was rare, standing dead trees were common and readily burned. These became aerial fuels, providing embers that advanced the fire through a short range spotting effect.

Upon entering young-growth redwood areas, Fuel Model 10 more closely predicted the actual fire behavior observed, but still over-predicted the rate of spread. In both young-growth and old-growth redwood stands, the wind effect on surface fire was minor. The canopy reduced the wind to approximately 0.1 times the forecasted 6.1-meter (20 ft) wind speed, reducing the advance of the fire front. Nor did surface wind from diurnal effects appear to be a factor in the spread rate.

Fire commonly would climb into the canopy in dead branches, moss, and lichens in both vegetation types, producing falling embers. Torching trees were more common in young-growth redwood. These effects produced substantial fire advance from ember ignited spot fires, some reported as far as one-mile ahead of the main fire. Spot fires were common and predictable downwind from ridges.

Topography and access were important factors in the Canoe fire. Only limited logging had occurred on the area's moderate to steep slopes before park designation. Using old skid routes and ridgelines, bulldozer access to the fire area was feasible, as was demonstrated by the eventual construction of a helispot. Further use of bulldozers for fire suppression was not implemented at that time, given management constraints.

Because the Canoe fire started in a state wilderness area, managers implemented a Minimum Impact Suppression Tactics (MIST) approach that affected fire suppression effectiveness. The balancing of resource impacts from operations against the impacts of wildfire was a delicate issue. Using the Eel River and the nearest road system for control was possible on the north, east, and west and would increase the 12-hectare (30 ac) fire to 5,000 plus hectares. But on the southern flank, no logical firebreak was present within the park. Keeping the fire on the State Park was a declared objective. Direct suppression was selected as the best of several poor options.

Low impact overland access was attempted via several routes. It took several days to determine that this was not feasible given firefighter safety issues. Escalating the impact, bulldozer access (blade up) was made to create a helispot. Fire crews were inserted, access made, and direct fire line construction started. Firefighter safety issues resulted in abandoning this plan, as spot fires started that could not be safely contained. Twelve days after the Canoe fire started, the road system and park boundary became the targeted control points. Critical fire weather conditions were now also affecting the fire area.

After constructing and improving the control lines on the north, west, and south fire flanks, firing operations were implemented to create a wide area of burned fuel before the main fire reached these control lines. Along the southern ridge control line, spot fires were large and numerous, requiring new control lines well onto private land. On the western flank, the fire crossed the control line near Grasshopper Lookout before firing operations could be completed. The eastern flank, between the Eel River and the main fire, was intentionally burned using helicopter dropped incendiaries on a 230 meter by 305 meter (75 ft by 100 ft) grid.

Within the old-growth stands, canopy loss from direct burning was rare. However, the long duration burning of surface fuels is likely to have killed the cambium layer of trees in all size classes. The two-percent mortality reported by Fritz in the early 1930s represented old-growth stands that likely had experienced occasional, low intensity fire. The Canoe fire area has been subject to years of fire exclusion. The larger dead fuels that contribute to long duration burn times would be in greater abundance than in areas with frequent fire cycles. This produces a higher intensity and duration of burn that may likely produce greater mortality than found in the earlier study.

Younger stands adjacent to the old-growth areas burned with greater intensity, spread rates, and flame lengths. Both old-growth and young-growth fuel complexes were resistant to control, but improved access in young-growth stands increased suppression effectiveness.

The long duration burning produced smoke impacts that lasted for weeks. The North Coast Unified Air Quality Management District issued daily health advisories for southern Humboldt County. Measured peak levels of PM10 from smoke in Myers Flat often exceeded 800 $\mu gm/m^3$. Some schools near the fire area closed and many residents left the area due to poor air quality (Torzynski, personal communication). While a relatively short-term impact, the smoke impacts from wildfire, or prescribed burning, must be a management consideration.

Conclusions

Fire behavior prediction models over-predicted flame lengths and spread rates for old-growth redwood stands. The duration and intensity of burning lasted many days beyond the initial passage of the fire front. Inability to access the Canoe fire with ground-based fire suppression resources while it was still small was the main factor allowing the fire to grow. Embers falling a short distance ahead of the fire front were the primary means of fire advance. Trees and snags fell at a high rate, presenting a greater risk to firefighters than from fire spread rates or flame lengths. While direct damage to old-growth tree crowns was rare, the high intensity of the surface fire, due to fuel accumulation during an extended fire-free period, is likely to produce higher mortality. Regional smoke impacts from wildfire in this old-growth stand were substantial. Parks must be managed to anticipate and mitigate impacts that fire will have on vegetation, habitat, watersheds, and neighboring property. Prescribed burning should be applied to decrease the fire-free interval, reducing fuel accumulation and potential damage to habitat. Fire management access must be improved by planning approved routes, helispot construction, and maintenance. Parks must incorporate fuels management adjacent to private land to reduce potential future wildfire risk and impacts.

References

Anderson, H.E. 1982. **Aids to determining fuel models for estimating fire behavior.** Gen. Tech. Rep. GTR-INT-122. Ogden, UT: Intermountain Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 22 p.

Andrews, P.L. 1986. **BEHAVE: fire behavior prediction and fuel modeling system: BURN subsystem, part 1.** Gen. Tech. Rep. GTR-INT-194. Ogden, UT: Intermountain Research Station, Forest Service, U.S. Department of Agriculture; 130 p.

California Department of Forestry and Fire Protection. 2003. Wildfire perimeters. [GIS shapefile]. Forest Resource Assessment Program, P.O. Box 944246, Sacramento, CA 94244.

Finney, M.A.; Martin, R.E. 1993. **Modeling effects of prescribed fire on young-growth coast redwood trees.** Canadian Journal of Forest Research 23: 1125-1135.

Fritz, E. 1932. **The role of fire in the redwood region.** Berkeley, CA: University of California Press; 23 p.

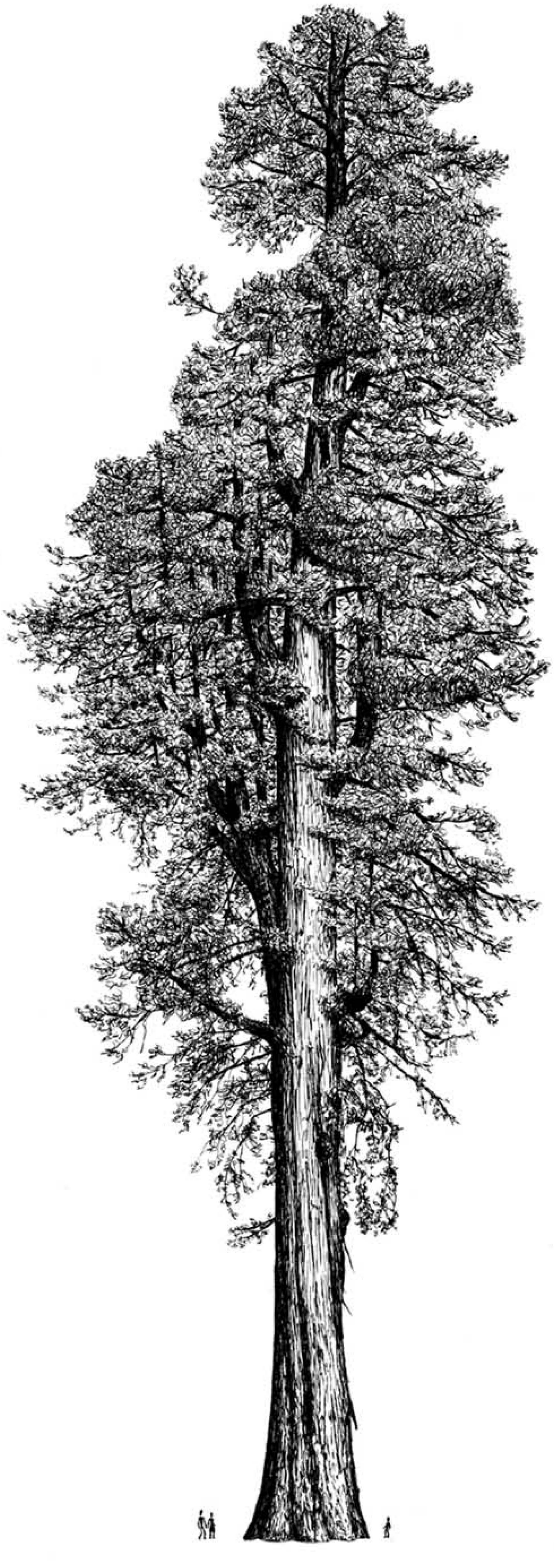
Nives, S.L. 1989. **Fire behavior on the forest floor in coastal redwood forests, Redwood National Park.** Arcata, CA: Humboldt State University; 73 p. Master's thesis.

Stuart, J.D. 1987. **Fire history of an old-growth forest of *Sequoia sempervirens* (Taxodiaceae) in Humboldt Redwoods State Park, California.** Madroño 34(2): 128-151.

Torzynski, R., North Coast Unified Air Quality Management District, 2300 Myrtle Avenue, Eureka, CA 95501. [personal communication].

SESSION 6

Wildlife and Fisheries II



Abundance and Habitat Associations of Dusky-Footed Woodrats in Managed Redwood and Douglas-fir Forests[1]

Keith A. Hamm,[2] Lowell V. Diller,[2] and Kevin D. Hughes[3]

Simpson Resource Company (formerly Simpson Timber) initiated studies on dusky-footed woodrats *(Neotoma fuscipes)* in 1992 on its approximate 450,000-acre ownership in Humboldt and Del Norte Counties, California. This land base is comprised of second and third growth forests primarily managed under an even-aged (clearcut harvest) silviculture technique. Interest in abundance and habitat associations of woodrats was driven by its importance as a primary prey item for the federally threatened northern spotted owl *(Strix occidentalis caurina)*. Simpson's studies of spotted owls have shown that woodrats comprise approximately 45 percent frequency and 70 percent biomass of prey consumed by spotted owls.

Research projects on dusky-footed woodrats have consisted of two master's thesis projects at Humboldt State University and one "in-house" study. In the 1992 to 1993 master's study, we live-trapped woodrats on 1.2 ha grids located in 24 forested stands from four age classes of redwood *(Sequoia sempervirens)*/Douglas-fir *(Pseudotsuga menziesii)* forest. Age classes were: five to nine years, 10 to 20 years, 21 to 60 years, and 61 to 80 years. We live-trapped each stand for five nights with Tomahawk (model #201) traps. In the 1999 "in-house" study, we live-trapped woodrats in 15 redwood/Douglas-fir stands ranging from young regeneration nine to 15 years old to mature second growth forest 50 to 70 years old that had varying levels of commercial thinning harvest. Thinning existed on a continuum of basal area removed, but for the purposes of sampling we placed stands into light, medium and heavy thinning categories. Vegetation was measured in 0.04 ha circular plots within trapping grids. Akaike's Information Criterion (AIC) was used to identify the top models predicting woodrat occurrence. During 2000 and 2001, an HSU graduate student sampled 29 stands of Douglas-fir/tanoak *(Lithocarpus densiflorus)* forests for woodrats through the use of live-trapping techniques. Stands were stratified into four age classes: five to 20; 21 to 40; 41 to 60; 61 plus years. Two transects were randomly located within each stand and 25 Tomahawk live traps were placed at 15 m intervals along each transect. Trapping was conducted for five nights. Captured woodrats were marked and released. Vegetation was measured in 0.04 ha circular plots along trap lines. The top models for predicting woodrat occurrence were ranked according to AIC values. In addition, this study compared woodrat house centered vegetation plots and randomly chosen plots to investigate the influence of habitat variables on nest site (house) selection.

During 1992 to 1993 we found woodrats were most abundant in young stands

---

1 An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

2 Simpson Resource Company, Korbel, CA 95550, (707) 668-4437. email: khamm@simpson.com

3 Dept. of Biology, Montana Tech University, Butte, MT 59701.

from five to 20 years of age. Density estimates were ≥31 woodrats/ha in stands five to 20 years old and ≤2 woodrats/ha in stands 21 to 80 years old. In 1999, woodrats trapped in clearcut areas <15 years of age were found in similar abundance to clear-cuts trapped in 1992 to 1993. Abundance in clearcuts was significantly greater than thinned stands ($\chi^2 = 12.54$, $P = 0.006$). In thinned stands, woodrats were associated with increasing understory cover, increasing amounts of redwood shrub cover, and decreasing amounts of Pacific rhododendron *(Rhododendron macrophylum)* and salal *(Gaultheria shallon)* cover. Woodrats were negatively associated with conifer stems >45.7 cm dbh. A Poisson regression indicated that woodrats began responding to thinning when stand basal area approached a lower limit of 28 m^2/ha. During 2000 to 2001, 207 different woodrats were captured among the four age classes of Douglas-fir forest. Woodrat abundance differed among the four age classes of Douglas-fir forest ($\chi^2 = 13.27$, $P = 0.004$) and woodrat abundance was negatively related to stand age ($r_s = -0.68$). The top model predicting woodrat abundance indicated a negative association with stand age, a positive association with shrub hardwood cover and a negative association with percent Douglas-fir in the shrub layer. The top model predicting woodrat house occurrence showed positive associations with ground cover of tanoak, percent tanoak in the shrub layer and density of understory tanoak.

All three studies indicate that in the redwood/Douglas-fir zone of Simpson's ownership, woodrats are in greatest abundance in young stands <20 years of age. Use of uneven-aged silviculture techniques such as commercial thinning or selection is not likely to enhance woodrat abundance because these practices generally encourage the proliferation of shade tolerant understory species that are not palatable forage for woodrats. Silviculture practices that promote a dense and diverse shrub layer of heliophilic species that are more palatable should promote woodrat abundance. However, woodrats also require suitable substrate in the form of redwood or tanoak stump sprouts, logs, and other down material for construction of their houses.

Because woodrats are the primary prey species of spotted owls *(Strix occidentalis caurina)* in northern California, forest management practices that influence woodrat abundance have implications to management of populations of threatened spotted owls. Thinning of mature stands is not likely to enhance the primary prey base for spotted owls in this region. The management strategies for threatened populations of spotted owls must take into consideration the habitat needs of the species itself and that of its primary prey.

Individual Legacy Trees Influence Vertebrate Wildlife Diversity in Commercial Forests[1,2]

M. J. Mazurek[3] and William J. Zielinski[4]

Key words: basal hollows, bats, small mammals, and birds, biodiversity, biological legacy, forest management, legacy tree, managed forests, northwestern California, redwood, Sequoia sempervirens, *wildlife communities*

Old-growth forests provide important structural habitat elements for many species of wildlife. These forests, however, are rare where lands are managed for timber. In commercial forests, large and old trees sometimes exist only as widely-dispersed residual or legacy trees. Legacy trees are old trees that have been spared during harvest or have survived stand-replacing natural disturbances. We define legacy trees as having achieved near-maximum size and age, which is significantly larger and older than the average trees on the landscape. This distinguishes them from other 'residual' trees, which may also have been spared from harvest but are not always larger and older than the average trees in the landscape. The value of individual legacy trees to wildlife has received little attention by land managers or researchers within the coast redwood *(Sequoia sempervirens)* region where 95 percent of the landscape is intensively managed for timber production.

We investigated the use of individual legacy old-growth redwood trees by wildlife and compared this use to randomly selected commercially-mature trees. This research was conducted during 2001 and 2002 in Mendocino County, California. At each legacy/control tree pair we sampled for bats using electronic bat detectors, for small mammals using live traps, for large mammals using remote sensor cameras, and for birds using time-constrained observation surveys. Legacy old-growth trees containing basal hollows were equipped with 'guano traps'; monthly guano weight was used as an index of roosting by bats. The diversity and richness of wildlife species recorded at legacy trees was significantly greater than at control trees (Shannon index = 2.81 vs. 2.32; species = 38 vs. 24, respectively). The index of bat activity was significantly greater at legacy trees compared to control trees ($F_{1,\ 45.7}$ = 17.66, P < 0.0001). There was a significantly greater number of birds observed (t = 16.6, P < 0.0001) and time spent (t = 4.05, P = 0.0004) at legacy trees compared to control trees. Woodpeckers, nuthatches, and some swallows were observed only at legacy trees. There was also twice as much foraging activity by birds at legacy trees

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] This article was published in *Forest Ecology and Management* 193 (2004) 321-334, and can be viewed in full at: http://www.sciencedirect.com/science/journal/03781127
[3] Wildlife Biologist, USDA Forest Service, Pacific Southwest Research Station, 1700 Bayview Dr., Arcata, CA 95521. email: mmazurek@fs.fed.us
[4] Research Ecologist, USDA Forest Service, Pacific Southwest Research Station, 1700 Bayview Dr., Arcata, CA 95521. email: bzielinski@fs.fed.us

compared to control trees. We found no statistical differences between legacy and control trees in the numbers of small mammals captured (*t* = 0.5, P = 0.62) or in the number of species photographed using remote cameras (*S* = 37.5, P = 0.10). Every basal hollow contained bat guano and genetic methods confirmed use by four species of bats. Vaux's swifts, pygmy nuthatches *(Sitta pygmaea)*, violet-green swallows *(Tachycineta thalassina)*, and the long-legged bat *(Myotis volans)* reproduced in legacy trees.

As measured by species richness, species diversity, and use by a number of different taxa, legacy trees appear to add significant habitat value to managed redwood forests. The use of legacy trees by wildlife was demonstrated by evidence of their nesting, roosting and resting; behaviors which were not observed at control trees. This value is probably related to the structural complexity offered by legacy trees. The presence of a basal hollow, which only occur in legacy trees, was the feature that appeared to add the greatest habitat value to legacy trees and, therefore, to commercial forest stands. Basal hollows were used by every taxa sampled, but appear to be particularly important to bats and birds. Because roost and nest availability can limit the populations of birds and bats, basal hollows may play a critical role in the redwood region if they provide roost and nest sites in forests that are otherwise deficient.

Our traditional view of conservation reserves is of large protected areas. However, few landscapes provide us with the opportunity to preserve large tracts of land and we must consider conserving biodiversity within the matrix of multiple use lands. Given the fragmented nature of mature forests in the redwood region, remnant patches of old-growth and individual legacy trees may function as 'mini-reserves' that promote species conservation and ecosystem function. Therefore, legacy trees may provide some measure of habitat connectivity ('stepping stones') to larger more contiguous tracts of old-growth forests. Because of their rarity in commercial forests, the first step in the management of legacy trees is to determine their locations and protect them from logging or from physical degradation of the site. Because legacy redwoods with basal hollows are even rarer, locating and protecting these should be the highest priority. In addition, the circumstances that lead to their genesis will be difficult to recreate, especially on commercial timberland. Hollows form by repeated exposure of the base of trees to fire, and because most fires on private land are suppressed, prescribed fire would need to be repeatedly applied to trees that would be designated as 'future legacies' and which would be excluded from harvest in perpetuity. Even without management to encourage basal hollows we suggest that managers plan for the recruitment of trees that are destined to become legacies. Although we do not believe that any one tree will protect a species, we do believe that the cumulative effects of the retention, and recruitment, of legacy and residual trees in commercial forest lands will yield important benefits to vertebrate wildlife and other species of plants and animals that are associated with biological legacies.

The results of our study beg us to consider habitat at a spatial scale that is smaller than that of habitat patches or remnant stands; we conclude that *individual trees* can have very important values to wildlife. More research would be helpful, however, to specify the level of individual tree retention required to maintain biodiversity in managed lands. It would help to know, for example, whether the fitness of individual species, and the diversity of wildlife communities, is greater in landscapes in which legacy trees are common compared to landscapes with very few legacy trees. It is possible that because legacy trees are rare—despite their apparent values to wildlife— that they do not affect wildlife diversity or productivity over large areas. It would also advance our knowledge to determine whether legacy trees in legacy-rich landscapes can function to maintain connectivity between protected stands of mature and old-growth forests. If so, the landscape context will be an important component of managing residual legacy trees and planning their recruitment across landscapes. For now, however, this study makes clear that protecting legacy trees will protect important habitat features that receive disproportionate use by many wildlife species. The protection and management of these trees can enhance wildlife conservation on lands where the opportunities to do so can be limited.

The Relationship Between the Understory Shrub Component of Coastal Forests and the Conservation of Forest Carnivores[1]

Keith M. Slauson[2] and William J. Zielinski[2]

The physical structure of vegetation is an important predictor of habitat for wildlife species. The coastal forests of the Redwood region are highly productive, supporting structurally-diverse forest habitats. The major elements of structural diversity in these forests include trees, shrubs, and herbaceous plants, which together create three-dimensional complexity. In the forests of the northern Redwood region, dense, continuous shrub layers were common understory structural elements in mature forests (Sawyer and others 2000). However, within the last 60 to 80 years, most of these forests have been logged and subsequently managed on short rotations (for example, 60 years) to maximize the production of wood. This has resulted in a reduction in the complexity of shrub and herb layers in these forests due to a combination of detrimental factors (for example, mechanical damage, burning, herbiciding, competition for light with densely stocked stands, fragmentation by roads). We investigated the importance of shrub cover to three species of mesocarnivores, the American marten *(Martes americana)*, fisher *(M. pennanti)*, and gray fox *(Urocyon cinereoargenteus)*.

To determine whether the distributions of mesocarnivores have changed since large scale changes in forest structure have occurred in the Redwood region, we compared the historical distributions of each of the three mesocarnivore species to their contemporary distributions in Northwestern California. For historical information we used the records from Grinnell and others (1937), which covered the period from 1919 to 1924 in California. To determine contemporary distribution we conducted surveys using two, five, and 10 km systematic grids, with two track plates at each two km grid point and six track plates at each five and 10 km grid point; each station was surveyed for 16 consecutive days (see Zielinski and others 2000). From 1995 to 2002, we sampled 493 sample units, composed of 1,920 individual stations, representing 30,720 survey days.

American martens were historically found within coastal forests (redwood and Douglas-fir types) from near the Oregon border to northwestern Sonoma county; most records (20 of 24) occur within ≤25 km of the coast. Contemporary detections of martens were limited to a single population, occupying an area equivalent to less than five percent of its historical range. Fishers were historically found in more interior forests with nearly all records (50 of 51) occurring ≥25 km from the coast. However contemporary detections of fishers occurred commonly in both interior and near coast forests with 30 of 76 (39 percent) stations <25 km and 68 of 377 (18

---

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] USDA Forest Service, Pacific Southwest Research Station, Redwood Sciences Lab, 1700 Bayview Drive, Arcata, CA 95521, (707)825-2931. email: kslauson@fs.fed.us and bzielinski@fs.fed.us

percent) >25 km detecting fishers. Most historical records of gray foxes occurred in interior forests (78 of 100), but clusters of records occurred in near coast areas with naturally mixed habitats (for example, forest and oak woodland or forest and estuarine) such as the Humboldt bay vicinity. Contemporary detections of gray foxes also occurred commonly in both coastal and interior forests, with 14 of 50 (28 percent) sites <25 km and 38 of 177 (21 percent) >25 km detecting gray foxes. Thus, by comparing historical and contemporary information, the distribution of the marten has dramatically declined in near coast forests while the fisher and gray fox have maintained their interior distributions and now appear to have expanded their distributions in coastal forests.

In an investigation of habitat use by the remnant population of martens, Slauson (2003) found that stands of old growth conifers with dense, spatially extensive shrub cover were most often selected. This population occupies the largest remnant area of late-successional dominated coastal forest within its historical range. Fishers and gray foxes were rarely detected within this region, despite being commonly detected in adjacent areas that are either more interior, or in logged coastal forest landscapes. Martens show the strongest selection for stands with >80 percent shrub cover and select against stands with <60 percent shrub cover. Both fishers and gray foxes show selection against stands with shrub cover >80 percent and use sites with <60 percent in proportion to availability.

The shrub layers used by martens in coastal forests are typically dominated by salal *(Gautheria shallon)*, rhododendron *(Rhododendron macrophyllum)*, evergreen huckleberry *(Vaccinium ovatum)*, and huckleberry oak *(Quercus vaccinifolia)*. Overall in unlogged sites, mean shrub cover naturally declines with increasing distance from the coast *(table 1)*. At 20 km from the coast shrub cover is at the lower end of shrub cover used by martens and begins to enter the range used more by the fisher and gray fox. However, in logged sites shrub cover is significantly reduced in the five to 20 km region that naturally supports denser shrub cover. This has created conditions in the near coast region that now appear more suitable for use by the fisher and gray fox and less unsuitable for the marten.

Table 1—*Mean shrub cover (SE) for unlogged and logged stands and their distance from the coast.*

	Distance from coast (km)							
	5	**10**	**15**	**20**	**25**	**30**	**35**	**>35**
Percent unlogged	92	86	77	59	66	53	54	34
(n = 584)	(1.7)	(2.0)	(2.6)	(3.8)	(3.2)	(3.2)	(7.0)	(1.9)
Percent logged	55	57	50	46	60	72	34	44
(n = 218)	(6.1)	(5.0)	(5.3)	(4.4)	(6.6)	(6.8)	(13.9)	(7.4)

The distributions of several mesocarnivore species have changed over the last 80 years, perhaps in response to the changes in understory forest structure we have described here. Two species, the fisher and the gray fox, now appear more common in redwood forest types than in the early 1900s. Both of these species typically occupy forest types interior to the redwood region where understory shrub densities are naturally lower and are absent or rarely detected within coastal forest habitats with dense, spatially-extensive shrub cover. The American marten has disappeared from >95 percent of its historical range within the redwood region, in part due to over-trapping (Zielinski and others 2001). It currently only occurs in coastal forest habitats with dense, spatially-extensive shrub cover. The smaller-bodied marten may be able to more effectively forage within the dense shrub layers than fishers or gray foxes. Dense, spatially-extensive understory shrub cover may play an important role in structuring the mesocarnivore community in coastal forests of the Redwood region. The cover of mature shrub communities may be critical to the restoration of the American marten across this region. Strategic restoration and management efforts will be necessary to return this natural structural element where it has been lost.

References

Grinnell, J.; Dixon, J.S.; Linsdale, J.M. 1937. **Fur-bearing mammals of California**. Vol. 1. Berkeley, CA: University of California Press; 375 p.

Sawyer, J.O.; Silett, S.C.; Popenoe, J.H.; LaBanca, A.; Sholars, T.; Largent, D.L.; Euphrat, F.; Noss, R.F.; Van Pelt, R. 2000. **Characteristics of redwood forests**. In: Noss, R.F., ed. The redwood forest: history, ecology, and conservation of the coast redwoods. Covelo, CA: Island Press; 339 p.

Slauson, K.M. 2003. **Habitat selection by a remnant population of American martens in coastal northwestern California**. Corvallis, OR: Oregon State University. M.S. Thesis. *Available at* http://www.rsl.psw.fs.fed.us/pubs/WILD90S.html

Zielinski, W.J.; Truex, R.L.; Campbell, L.A.; Carroll, C.R.; Schlexer, F.V. 2000. **Systematic surveys as a basis for the conservation of carnivores in California forests**. Pacific Southwest Research Station, Redwood Sciences Lab, Forest Service, U.S. Department of Agriculture, Redding, CA. Unpublished report; 23 p. *Available at* http://www.rsl.psw.fs.fed.us/pubs/WILD90S.html

Zielinski, W.J.; Slauson, K.M.; Carroll, C.R.; Kent, C.J.; Kudrna, D.K. 2001. **Status of American marten populations in the coastal forests of the Pacific states**. Journal of Mammalogy 82(2): 478-490. *Available at* http://www.rsl.psw.fs.fed.us/pubs/WILD90S.html

Fisher *(Martes pennanti)* Use of a Managed Forest in Coastal Northwest California[1]

Joel Thompson,[2] Lowell Diller,[2] Richard Golightly,[3] and Richard Klug[4]

A sooted track plate survey was conducted for two seasons during winter, spring and summer of 1994 and 1995 to investigate fisher distribution across a managed landscape in Humboldt and Del Norte Counties, California. Forty survey segments were established throughout the region with each segment consisting of six sooted track plates (stations) at one-km intervals. Habitat characteristics were measured at 238 track-plate stations and in the stands where they were placed. Habitat attributes associated with fisher detection sites were compared to non-detection sites at both the station and stand level. Vegetation type was the only variable to be selected by the logistic procedure to predict fisher occurrence at the stand level. Fishers were detected more often in stands dominated by Douglas-fir *(Pseudotsuga menziesii)* or a mixture of Douglas-fir and coast redwood *(Sequoia sempervirens)* than in stands dominated only by redwood. We found no relationship between fisher detections and stand age, canopy cover, or topographic position. A forward stepwise logistic procedure indicated that presence of fishers at the station level was best predicted by increasing elevation, greater volume of logs, less basal area of conifer 52 to 90 cm, more moderate slopes and greater distance to the coast.

During 1996 to 1997 a telemetry project was conducted to specifically look at vegetation and structural characteristics of fisher den and rest sites. Twenty-four individuals (10 male, 14 female) were captured during the study. Twelve individuals (six male, six female) were fitted with radio collars and followed to determine resting and denning sites. A total of nine dens were found for five of six females outfitted with radio transmitters. These consisted of four natal and five maternal dens. The dens were located in four "highly decadent" live hardwoods, one "sound" hardwood and four conifer snags. Natal dens were all in cavities. Two were located in tanoaks *(Lithocarpus densiflorus)*, one chinquapin *(Chrysolepis chrysophylla)* and one Douglas-fir snag. Mean diameter at breast height (dbh) was 76.5 cm (SD = 15.6, range 62.5 to 95.3 cm). Maternal dens were also all in cavities. Three of the cavities appeared to be created by pileated woodpeckers *(Dryocopos pileatus)* and two appeared to have been created by old fire scars. The cavities were in two tanoaks, two Douglas fir snags and one western red cedar (*Thuja plicata*) snag, with a mean dbh of 112.0 cm (sd = 45.8, range 62.5 to 184.4). A total of 35 rest sites were located in seven different tree species, however, 3 species accounted for over 80 percent of all rest sites. These were western hemlock *(Tsuga heterophylla)*, western red cedar and Douglas-fir. The most common structures used as rest sites were platforms formed by mistletoe deformities, debris accumulations and lateral branches. These comprised

---

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Simpson Resource Company, P.O. Box 61, Korbel, CQ 95550, (707) 668-4438. email: jthoms @simpson.com
[3] Humboldt State University, Dept. of Wildlife, Aracata, CA 95521.
[4] Fruit Growers Supply Company, Hilt, CA 96044.

roughly 65 percent of all rest sites. Cavities were used for resting approximately 20 percent of the time. The mean dbh of trees containing rest sites was 84.6 cm with a range from 22.4 to 175.0 cm.

Beginning in 2002 a mark-sight study, using remote cameras to identify uniquely marked individuals, was initiated to estimate the density of fisher on two 100 km^2 study sites located in Humboldt County, northwestern California. Both study sites were located in the Douglas-fir/redwood transition zone. Fieldwork for this project has been completed but only preliminary summary data are presented here. Thirty-six fishers were captured on the two study sites (19 females, 17 males) over the two year period. Twenty-nine of 36 fishers captured were adults. Fourteen of 17 adult females appeared to be reproductive, with 11 lactating at the time of capture and three which were not lactating but later located at dens. The status of the remaining three adult females was not determined due to difficulties with transmitters and the inability to successfully locate them on the ground. During the resighting portion of the study, a grid with 1.6 km spacing between points was overlaid on each study site, creating 44 camera stations in each area. Twenty-two cameras were deployed in each study area (every other station) and checked once weekly for a two week period. Cameras were then moved to the adjacent stations and checked for another two week period, thereby sampling all 44 stations every four weeks. This was considered one "sighting period". During the 2002 season, cameras were monitored for three sighting periods from June through August, with fishers detected on 148 occasions. In 2003, cameras were monitored for 4 sighting periods from May-August, with fisher detections on 224 occasions. Ratios of marked to unmarked individuals were 1:2.1 and 1:1.6 during 2002 for the Bald Hills and NF Mad River sites respectively. In 2003, ratios were 1:0.52 and 1:0.38 for the Bald Hills and NF Mad River sites respectively. Crude population estimates, based only on the ratio of marked:unmarked fisher sightings, resulted in estimates of 22 fishers on the Bald Hills site in 2002 and 18 fishers in 2003. Estimates for the NF Mad River site were 21 fishers in 2002 and 11 fishers in 2003. In combination with the previous two studies mentioned above, it appears that fishers are relatively abundant and well distributed throughout the managed forests of extreme northwestern California.

Salmonid Communities in the South Fork of Caspar Creek, 1967 to 1969 and 1993 to 2003[1]

Bradley E. Valentine,[2] Richard A. Macedo,[2] and Tracie Hughes[2,3]

Abstract

Demand for wood products and advances in logging technology post-World War II resulted in timber harvesting that extensively modified streams on the North Coast of California. To assess the resulting impacts to salmonid populations, the Department of Fish and Game conducted studies at widely spaced sites throughout the redwood region during the 1960s. In order to compare current salmonid communities with those documented earlier, we resumed investigations on the South Fork, Caspar Creek in 1993. During the last decade, total salmonid density was within the range observed during the 1960s, albeit generally in the lower half of that prior range. Biomass has remained consistent between the time periods. The salmonid communities shifted from ones in which coho salmon and steelhead were both well represented to ones that were greatly dominated by one species or the other. Stream surface area and substrate quality between the two time periods are similar. Explaining the cause of the different salmonid community patterns between time periods are problematic but is probably related to climatic differences in ocean conditions and precipitation patterns. Because there was a single, early summer pretreatment assessment of salmonids, the long-term effects of the 1960s era timber operations can not be determined.

Key words: coho, dominance, salmonid community, steelhead, timber harvest

Introduction

Rapidly increasing construction activity at the end of World War II created heavy demand for wood products that dramatically increased timber harvest in California (Anonymous 1988). At the same time, tractor yarding and truck hauling replaced steam-railroad systems as the primary mode of yarding and log transport. These advances also enhanced accessibility to timber in more remote and rugged areas (Sawyer and others 2000). Finally, increased lumber values encouraged mills to improve technology and products, thereby increasing the range of merchantable timber materials and reducing post-harvest retention of live trees, snags, and downed logs.

Changes in timber harvest operations promoted severe and widespread impacts to watercourses. Many studies were initiated to evaluate the consequences of such operations on aquatic habitat and fishery values. Most studies had small spatial scales, but some were at a watershed scale (for example, Moring and Lantz 1975, Scrivener and Andersen 1984). In California, initial efforts to evaluate these impacts

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] California Department of Fish & Game, Central Coast Region, 7329 Silverado Trail, Napa, CA 94558.

[3] Tracie Hughes is now Tracie Nelson.

were both extensive (Burns 1972) and intensive (Ziemer 1998). The studies in total have generally found impacts to salmonids mediated through changes in solar radiation and temperature; reductions in channel complexity due to modified large woody debris loading and increased delivery of coarse sediment; and modifications in productivity through fining of the spawning beds and changed stability.

Anadromous salmonid populations across the west coast have been declining for some time, possibly due to a combination of both freshwater and marine conditions. For example, summarizing information mainly taken from Brown and others (1994), the NMFS (Weitkamp and others 1995) estimated that coho salmon *(Oncorhynchus kisutch)* escapement in California ranged between 200,000 and 500,000 in the 1940s, but had declined to an average of about 31,000 from 1987 to 1991, and that they were apparently extirpated from 46 percent of the streams with historic records. The National Marine Fisheries Service (Weitkamp and others 1995) also noted that traditional assessment of salmon populations had focused on the number of harvestable or reproductive adults but that data on other life history stages (for example, freshwater smolt production) can be used as supplemental indicator of abundance. In response to the mounting evidence of declining salmonid stocks, the National Marine Fisheries Service listed several Evolutionarily Significant Units of anadromous salmonids including coho and steelhead *(O. mykiss)* in California and the Pacific Northwest under the federal Endangered Species Act.

Little long-term trend information has been collected on juvenile salmonids. Yet, assuming juvenile populations are habitat limited (Burns 1971), trends in their numbers can provide insights into freshwater habitat conditions, a life stage which is an integral but not necessarily the most limiting portion of their life history. Motivated by this tenant and the availability of historic data (Burns 1970, 1972) against which to compare, a monitoring program was activated on the Little North Fork of the Noyo in 1992 (since assumed by Campbell Timberland Management) and the South Fork of Caspar Creek in 1993. This articles reports on the juvenile salmonid populations in the South Fork of Caspar Creek.

Methods

Burns' Era

Methods used during the 1967 to 1969 are more fully described in Burns (1970, 1971, 1972). Because this work was extensive, varied between watercourses, and was not always specified by study site, exact methods used at any one site are speculative. For instance, Burns used both mark-recapture and depletion methods to estimate populations in his study sites, so the method applied at any site must be inferred based on conditions described in his articles.

For spawning gravel analyses, Burns (1970) collected 20 samples mid-stream near the head of a riffle using a 15.24 cm diameter and 15.24 cm deep core 'McNeil' sampler. Samples were passed through sieves with 26.6, 3.327, and 0.833 mm openings and the fine materials were collected in a pan. Water displacement was used to determine volume of each screened sample. Residual sediments from the pan were measured in graduated cylinders after settling for 10 minutes.

To assess salmonid numbers, Burns sampled with a backpack electrofisher in June and October of each year from 1967 through 1969. He separated age classes by length-frequency methods and estimated population sizes probably with mark-recapture methods. To convert population estimates to density estimates, Burns measured stream dimensions using cross sections at probably 101 transects (average of one transect every 30 m). Since Burns reports the area of his study reaches but not the individual measures, we back-calculated average width during his studies from his length and surface area summary data (Burns 1972). Burns' study reach was a continuous reach that ranged from 2,723 to 3,091 m between samples. The varied reach lengths likely reflected differences in available habitat between water years. Burns' efforts spanned from a single pre-impact sampling (June 1967), to immediately after clearing and constructing the roads (October 1967), then through two years of continued road maintenance (June and October 1968 and 1969).

Recent Methods

Since 1993, we have sampled biota in four index reaches each approximately 100 m in length. Index reaches were selected non-randomly to meet two goals: to span Burns' study site and to sample distinctly different habitat conditions. Stream dimensions (wetted width and three equidistant water depths) were recorded at cross-stream transects every five or ten m within the sampled reach. Depths and locations of all pools were also measured. We weighed all collected vertebrates using an electronic scale (±0.1 g) calibrated at each pass. We measured fork-length (±1 mm) of salmonids, total length (±1 mm) of other fish taxa, and snout-vent length (±5 mm) of Pacific giant salamanders *(Dicamptodon tenebrosus)*. We estimated population size in each index reach using multiple-pass, depletion methods (Carle and Strub 1978). We used Price (1982) to determining the number of passes required for a total salmonid population estimate with a less than a ten percent error. Bulk sediment samples were collected and processed according to Valentine (1995), essentially identically to Burns (1970).

Results

From the pre-road construction effort in June of 1967 to the first autumn, Burns (1971) noted an approximately 85 percent decrease in both coho and steelhead density *(fig. 1)*. In each subsequent year, density declined from the June sampling effort to the October effort, but by much smaller amounts. Both coho and steelhead showed positive October trends for the subsequent two years, suggesting that habitat conditions were recovering. However, a comparison between coho trends in North Fork of Caspar Creek, a control watercourse, and the South Fork suggested otherwise. In the South Fork, Burns found that coho populations were 2.3 times as dense in October 1969 than in October 1967. But in the North Fork, coho density increased by a factor of 15 over the same period. During each of Burns' sampling periods, coho comprised a significant proportion (range 27 to 40 percent) of total salmonid density. During the 1993 through 2003 period, total salmonid density (range 0.33-0.96 fish/m^2) fluctuated just below to within the lower 1/2 of the range of Burns' October estimates (0.52-1.33 fish/m^2, *fig. 1*). With the exception of 1996, 2002, and 2003, coho were rare or absent during our study. Coho were nearly absent for three consecutive years (1998 to 2000); encompassing a complete cohort cycle. However, coho dominated (59 to 88 percent) the salmonid population in the final two years (2002 and 2003), and production of steelhead was nil in 2002.

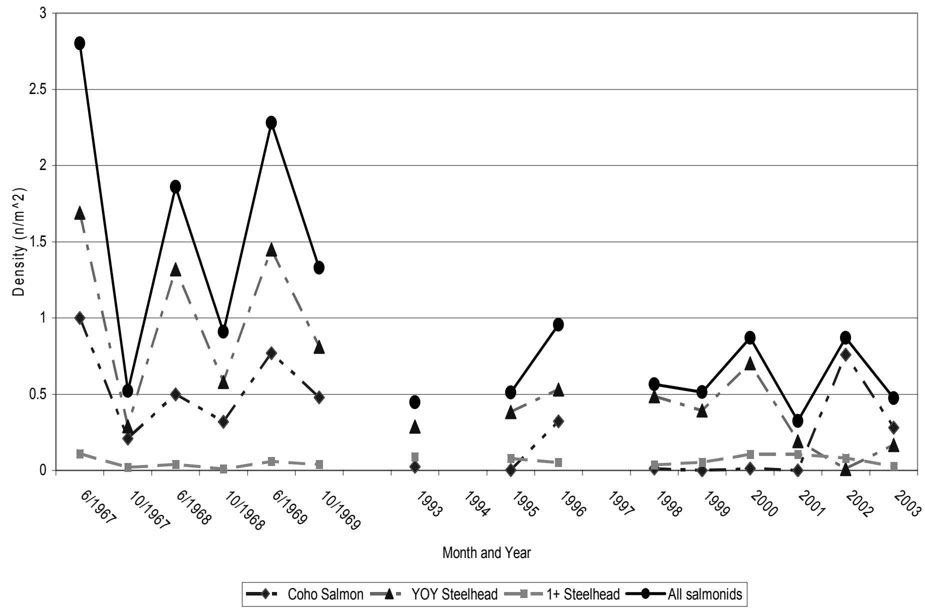


Figure 1—Density (#/m2) of salmonids in the South Fork of Caspar Creek, 1967 to 1969 and 1993 to 2003.

Biomass *(fig. 2)* shows similar changes as does density, although generally not as severe. By June of 1969, total salmonid and steelhead biomass were very similar to the pre-road building timber harvest (June 1967), while changes in coho and young of the year steelhead biomass suggested recovery. Relative coho biomass changes in the North Fork and the South Fork also suggested differential recovery. Coho salmon biomass in October 1969 for the South Fork was 1.3 times that in October 1967, while coho biomass in the North Fork increased by a factor of five over the same time. Like density, biomass declined dramatically during the first summer. Total salmonid biomass from 1993 to 2003 (range 15.71 to 31.15 kg/ha) was similar to that observed by Burns during his October samples (range 14.92 to 30.38 kg/ha). Total salmonid biomass reflected the species composition: steelhead accounted for most of the salmonid biomass prior to 2002. However, in three years (1996, 2002, and 2003), coho biomass exceeded even the values observed during the October samples from 1967 to 1969.

Substrate particle size indices varied widely *(fig. 3)*. The greatest within-year change was observed in 1967, when percent fines <0.8 mm increased from 20.6 percent to 34.2 percent, presumably in response to road construction activity. Within subsequent years, differences between June to October substrate values were negligible. Percent of fines <0.8 mm were marginally lower in the first spring after road-building than they were prior to harvest, suggesting that the fines had flushed during the preceding winter. However, percentage of fines <0.8 mm increased in June of 1969, apparently in response to stream bank erosion, erosion on road sidecast, and slides. Percentages of particles <3.3 mm showed parallel changes to that of the fines <0.8 mm category. Samples taken in 1993 were within the 1967 to 1969 range, suggesting spawning gravel quality was similar to the earlier samples and thus could not explain density and community structure differences between the periods.

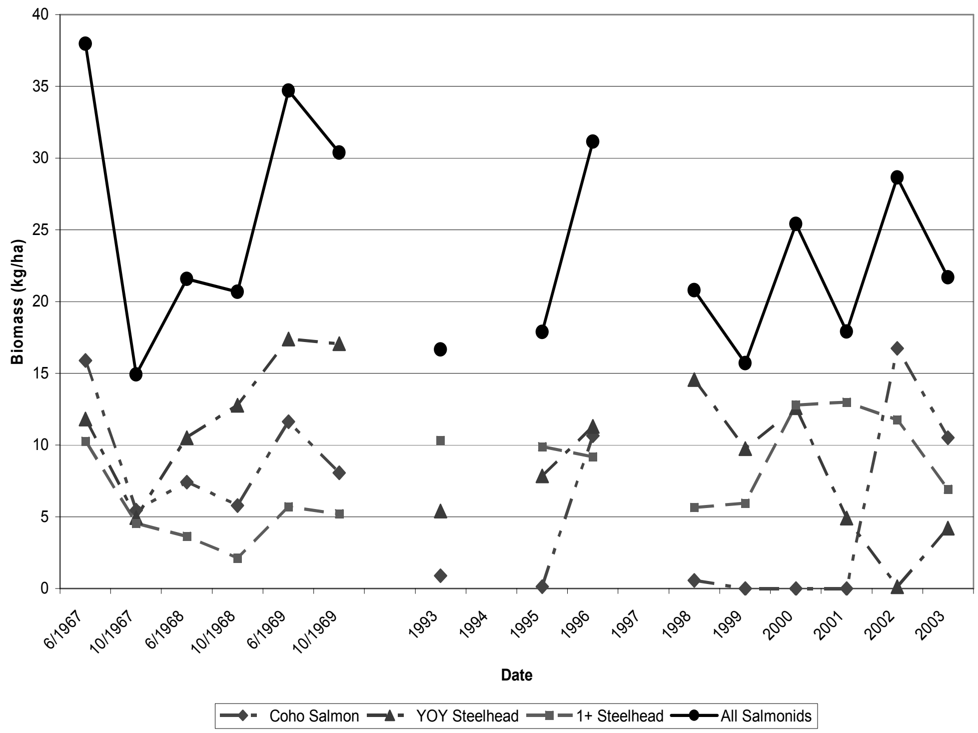


Figure 2—Biomass (kg/ha) of salmonids in the South Fork of Caspar Creek, 1967 to 1969 and 1993 to 2003.

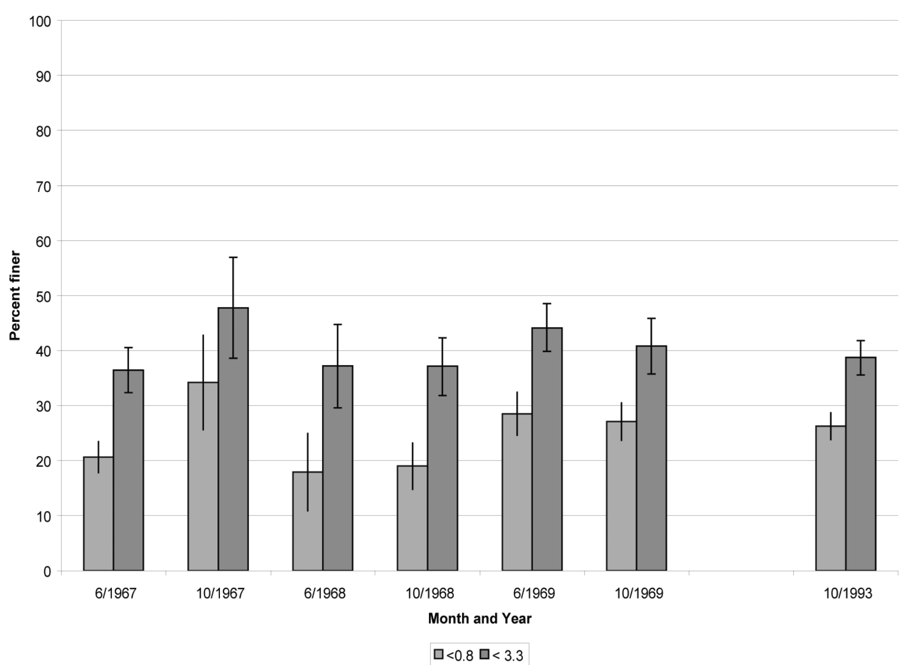


Figure 3—Mean (and 95 percent CI) condition of substrate in the South Fork of Caspar Creek expressed as the percentage of a sample smaller than two particle size criteria.

Average width during the 1967 to 1969 autumn samples generally declined *(fig. 4)*, suggesting a deepening and narrowing of the wetted channel. When evaluating all sampling periods, average width during the two study periods were similar. But considering autumn samples only, the wetted channel during 1993 to 2003 tended to be slightly wider (175.9 cm ± 8.9 95 percent CI) than it was during 1967 to 1989 (152.5 cm ± 19.1 95 percent CI). This suggest that any deepening and narrowing that began immediately after the initial impacts did not continue, and may have slightly reversed after years of processing the land-management delivered sediment. However, the data were not collected in such a means to enable statistical comparison.

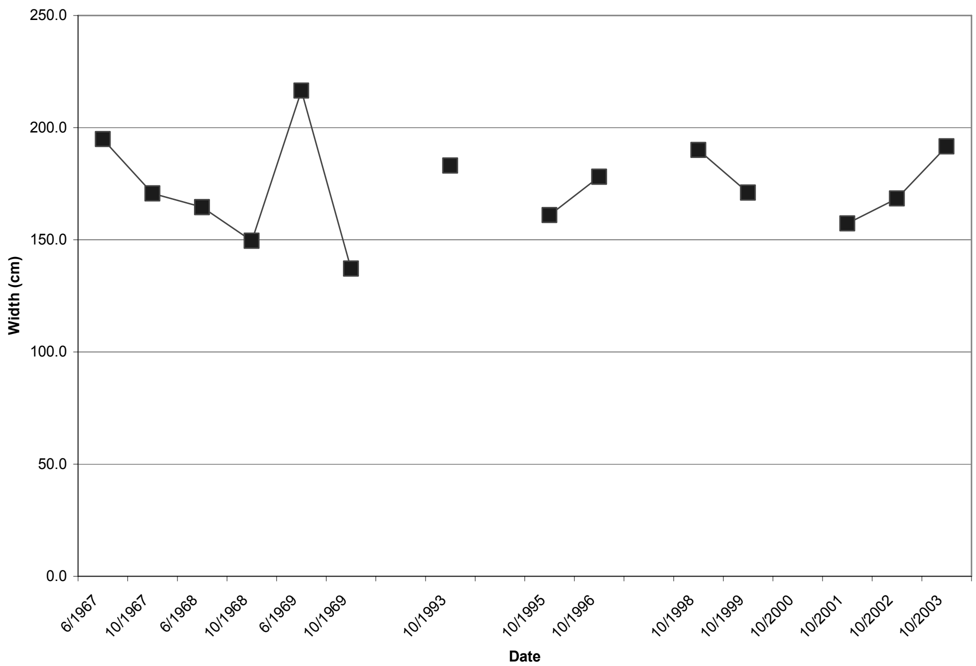


Figure 4—Average width (cm) of South Fork Caspar Creek.

Discussion and Conclusions

There are two primary differences apparent when comparing Burns' salmonid communities with ours. First, the autumn total salmonid density during the 1993 to 2003 period was slightly more stable and averaged lower but still within the lower part of the range reported by Burns (1972). Secondly, during Burns study the densities of steelhead and coho were relatively more even than during the last decade when one species has always dominated the salmonid community. The shift to strong species dominance appeared in the South Fork in 1989 (Nakamoto 1988), who found in sampling from 1986 through 1995 similar total densities and biomass as we found. These patterns may have resulted from the land management practices and subsequent habitat recovery, or temporal differences in variability of natural drivers of salmonid density.

The between-year October increases in total salmonid density observed by Burns may reflect short term recovery processes in the physical habitat as streamflows processed the disturbed channels. It may also reflect reductions in predator pressure, climatic and streamflow changes, or increases in nutrient availability. Timber management near streams can result in increased solar radiation, higher invertebrate densities and improved forage conditions for some salmonid species provided that that water temperature regimes remain within tolerable ranges (Murphy and others 1986). Burns (1972) reported that solar radiation increased an average of 98 Langley's/day, and that water temperature increased as much as 11.1 °C. After the right of way clearing and road building, urea was spread with the grass seed sown for erosion control. Burns (1972) reported that, within a two year period following road construction, benthos in the South Fork increased 370 over that which existed prior to road construction. All these factors enhanced available food base and likely enhanced salmonid growth. These land management changes to the food-base and channel should have become relative stabile with time as the channels and stream-side zones recovered. Beyond the land management induced changes and subsequent recovery, variation in natural conditions may subsequently drive variation in density of juvenile salmonids. Marine conditions (Beamish and others 2004) can affect the number of adults available to return. Differential flow patterns can affect the strength and success of the spawning run both between and within years (Lisle and Lewis 1992).

The simplification of salmonid communities expressed by the greater dominance of one species in the South Fork could reflect other factors. Reeves and others (1993) found that basins subjected to high levels of timber harvest are more likely to be dominated by a single species, and attributed that change to habitat simplification. In the South Fork, the same mechanism may play a role in the single species domination we observed.

Whether predation might differentially and variably affect the salmonid species abundance in Caspar Creek is unknown. As top carnivore of Pacific Northwest aquatic communities, giant salamanders may exert a controlling influence on populations of salmonid especially through predation on alevins and juveniles. During the last decade, Pacific giant salamanders have been generally as numerous as the salmonids. Unfortunately, Burns did not report on salamander density or biomass, thus eliminating our ability to use temporal differences to assess their possible community structuring role.

A weir was constructed on the South Fork in 1962 (Anonymous 1970). Over the years, the weir's fish ladder deteriorated and may have been a migration barrier under some flow conditions. Repairs completed in 1998 may have improved coho access upstream of the weir and into our study sites. However, this does not fully explain recent increases in South Fork coho salmon populations: the ladder still is not fully functional, Nakamoto (1998) recorded strong coho density up through 1989, and a similar species shift has been noted in the Little North Fork of the Noyo River,[4] a stream that does not have a similar migration barrier.

Despite the similarity in total salmonid biomass between 1967 to 1969 and 1993 to 2003, population densities were generally lower during the last decade. This contradiction could be explained by density dependant interactions among and between salmonids species (Chapman 1966). If a watercourse's carrying capacity declines across the summer period in varying but non-linear ways, salmonid populations may pass through a bottleneck and then enter a period where survivors experience reduced density-dependent pressure and enhanced growth. The contradiction could also be explained by the enhanced food resources observed by

[4] Unpublished data, David Wright, Campbell Timberland Management.

Burns (1972) being diminished as the adjacent forest regrows. Gregory and others (1987) noted that timber harvest enhanced nutrient levels persist for ten to 20 years followed by a decline in primary production during the following 40 to 80 years of deciduous forest stage.

There has been little change in wetted width and no change in channel substrate quality. As measured on the South Fork, these parameters provide little insight into the changes we observed in salmonid community structure. Bulldozer-assisted right-of-way clearing, road building, and stream channel clearing notably simplified the South Fork channel. Evidence persists today where the channel's cross section appears to be about one bulldozer blade wide in many locations (Valentine, pers. observ.). Recovery of the channel has been slow, as embedded LWD is gradually being exposed and off-channel LWD recruitment is initiating. The two decade lull in habitat change is consistent with Knopp (1993), who found that recovery from timber harvest impacts to watercourse metrics may be slow. He found various stream aquatic habitat indices were similar between recently managed (timber harvest) landscapes and those that had a 40 year break in management. However, he found that the same metrics differed significantly between unharvested basins and those which had been subjected to past management.

The fact that total salmonid density fluctuated somewhat less during 1993 to 2003 than during the samples of 1967 to 1969 is somewhat surprising based on general stream flow and climatic conditions. According to the Palmer Drought Severity Index (NCDC 2005), Burns experienced relatively stable and near normal climatic conditions before and during his studies. On the other hand, our sampling efforts included, and were preceded by years of widely varying and prolonged weather and stream flows conditions. The effects of storm flows on salmonid reproduction is specific to timing and magnitude and is mediated through controlling access to spawning areas and dynamics of redd scour (Lisle and Lewis 1992).

Late summer and early fall is a common period for conducting juvenile salmonid surveys. At this time of year, populations have experienced the stress of low flow and high temperature of summer, and have therefore provided biologists an insight into existing habitat quality. This sampling period will not capture another critical period that may limiting populations—the high runoff and turbidity conditions that accompany winter storm runoff (Nickelson and others 1992). Numerically and by biomass, 1+ steelhead were never dominant, but during the last decade, their density has been slightly greater than during the post impact period of Burns study. Since these fish have over-wintered prior to our sampling period, this suggests some improvement recently in winter habitat. Unfortunately, Burns (1972) only had one sampling period (June 1967) that preceded the treatments he evaluated. The June 1967 sample period does not provide much insight into late summer salmonid populations under unmanaged conditions. A single point for examining parameters with substantial variability leaves us with scarce "pre-treatment" data against which to assess the long-term impacts of timber harvest

Acknowledgments

This study would not have been possible without the strong encouragement of Jackson Demonstration State Forest Manager Marc Jamison. Support was also provided by Carl Wilcox of the Central Coast Region of the Department of Fish and

Game. Over the years, many people have assisted in field data collection—attempting to acknowledge them all would lead to unintended oversights, so we thank them all.

References

Anonymous. 1970. **Timber harvest and logging plan for the South Fork of the Caspar Creek watershed.** Unpublished Report.

Anonymous. 1988. **California's forests and rangelands: Growing conflict over changing uses.** California Department of Forestry and Fire Protection, Forest and Rangeland Resources.

Beamish, R.J.; Sweeting, R.M.; Neville, C.M. 2004**. Improvement of Juvenile Pacific Salmon Production in a Regional Ecosystem after the 1998 Climatic Regime Shift.** Trans. Am. Fish. Soc.133: 1163-1175.

Brown, L.R; Moyle, P.B.; Yoshiyama, R.M. 1994. **Historical decline and current status of Coho Salmon in California.** N. Am. J. Fish. Manage. 14: 237-261.

Burns, J.W. 1970. **Spawning bed sedimentation studies in Northern California Streams.** Calif. Fish & Game 56(4): 253-270.

Burns, J.W. 1971. **The Carrying Capacity for Juvenile Salmonids in Some Northern California Streams.** Calif. Fish & Game 57(1): 44-57.

Burns, J.W. 1972. **Some effects of logging and associated road construction on northern California streams.** Trans. Am. Fish. Soc. 101(1): 1-16.

Carle, F.L.; Strub, M.R.. 1978. **A new method for estimating population size from removal data.** Biometrics 34: 621-630.

Chapman, D.W. 1966. **Food and space as regulators of salmonid populations in streams.** Am. Nat. 100: 345-357.

Gregory, W.V.; Lamberti, G.A.; Erman, D.C.; Koski, K.V.; Murphy, M.L.; Sedell, J.R. 1987. **Influence of forest practices on aquatic production.** In: Salo, E.O.; Cundy, T.W., eds., Streamside Management: Forestry and Fishery Interactions. Institute of Forest Resources. Univ. of Washington, Contrib. 57. Seattle, WA; 233-255.

Knopp, C. 1993. **Testing Indices of Cold Water Fish Habitat.** Final Report to the North Coast Regional Water Quality Control Board. Santa Rosa, CA. 55 p.

Lisle, T.E.; Lewis,J. 1992. **Effects of sediment transport o survival of salmonid embryos in a natural stream: A simulation approach.** Can. J. Fish. Aquat. Sci. 49: 2337-2344.

Moring, J.R.; Lantz, R.L. 1975. **The Alsea Watershed Study: Effects of logging on the aquatic resources of three headwater streams of the Alsea River, Oregon. Part I—Biological Studies.** Oregon Dept. Fish Wildl. Fish Res. Rept. No. 9.; 66 p.

Murphy, M.L.; Heifetz, J.; Johnson, S.W.; Koski, K.V.; Thedinga, J.F. 1986. **Effects of clear-cut logging with and without buffer strips on juvenile salmonids in Alaskan streams**. Can. J. Fish. Aquat. Sci. 43: 1521-1533.

Nakamoto, R.M. 1998. **Effects of timber harvest on aquatic vertebrates and habitat in the North Fork Caspar Creek**. In: Ziemer, R.R., technical coordinator. Proceedings of the conference on coastal watersheds: The Caspar Creek Story. Gen. Tech. Rep. PSW-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 87-96.

National Climate Data Center (NCDC). 2005. S*ee* http://lwf.ncdc.noaa.gov/oa/climate/onlineprod/drought/xmgrg3.html

Nickelson, T.E.; Rodgers, J.D.; Johnson, S.L.; Solazzi, M.F. 1992. **Seasonal changes in habitat use by juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams.** Can. J. Fish. Aquat. Sci. 49(4): 783-789.

Price, D.G. 1982. **A fishery resource sampling methodology for small streams.** Pacific Gas & Electric Co., Dept. Eng. Res. Rep. 420-81.141; 49 p.

Reeves, G.H.; Everest, F.H.; Sedell, J.R. 1993. **Diversity of Juvenile Anadromous Salmonid Assemblages in Coastal Oregon Basins with Different Levels of Timber Harvest.** Trans. Amer. Fish. Soc. 122: 309-317.

Sawyer, J.O.; Gray, J.; West, G.J.; Thornburgh, D.A.; Noss, R.F.; Engbeck, Jr. J.G.; Marcot, B.G.; Raymond, R. 2000. **History of Redwood and Redwood Forests,** Chapter 2. In: Noss, R.F., ed. The Redwood Forest. Covelo, CA: Island Press; 339 p.

Scrivener, J.C.; Andersen, B.C. 1984. **Logging impacts and some mechanisms that determine the size of spring and summer populations of coho salmon fry (*Oncorhynchus kisutch*) in Carnation Creek, British Columbia.** Can. J. Fish Aquat. Sci. 41: 1097-1105.

Valentine, B.E. 1995. **Stream substrate quality for salmonids: Guidelines for sampling, processing and analysis.** In: Duncan-Vaughn, J., ed. Proceedings of the thirteenth annual Salmonid Restoration Federation Conference; 76-79.

Weitkamp, L.A.; Wainwright, T.C.; Bryant, G.J.; Milner, J.B.; Teel, D.J.; Kope, R.G.; Waples, R.S. 1995. **Status review of coho salmon from Washington, Oregon, and California.** U.S. Dep. Commer., NOAA Tech. Memo. NMFS-NMFSC-24; 258 p.

Ziemer, R.R., technical coordinator. 1998. **Proceedings of the conference on coastal watersheds: The Caspar Creek Story**. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture, Gen. Tech. Rep. PSW-168; 149 p.

Amphibians as Indicators of Headwater Processes in Northern California Forests: What Are They Telling Us and Why Should We Listen?[1]

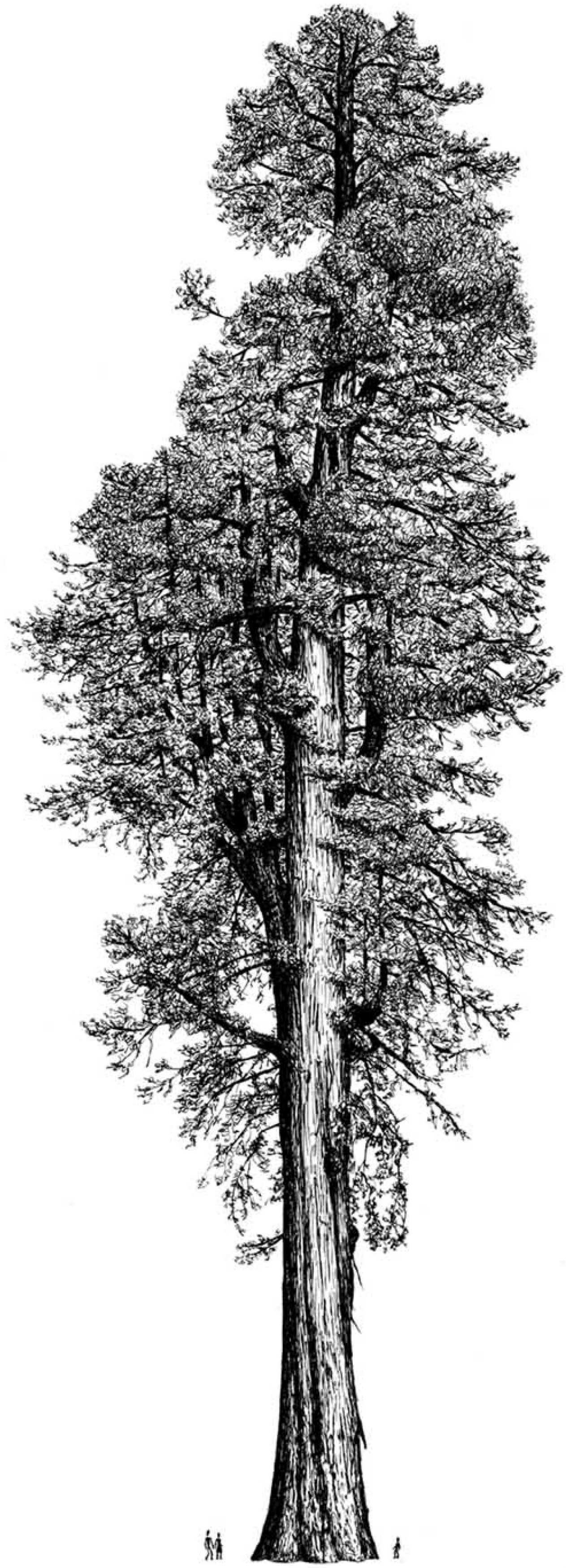
Hartwell H. Welsh, Jr.[2]

Amphibians are recognized for their sensitivity to environmental perturbations and for the advantages they offer as metrics of environmental health. Our research on headwater amphibians in the Klamath-Siskiyou and North Coast Bioregions spans nearly twenty years. Clear biotic patterns have emerged from our combination of broadscal retrospective studies and finer scale natural experiments. These patterns indicate that the species composition and relative abundances of headwater-adapted amphibians, respond directly to changes in headwater processes that determine fine sediment loads, water temperatures, and the forming of stream channel habitats and sorting of streambed substrates that are influenced by amounts of large woody debris. I review five studies that demonstrate the connections between amphibian diversity and abundance and key headwater processes. The notion that monitoring biota as metrics of changes in headwater stream processes is too difficult because of extreme background noise is a fallacy. Amphibians respond directly to perturbations in these processes, and thresholds for negative responses by these species can be determined. While responses of anadromous species are difficult to link directly with changes in headwater processes because of the challenge of separating out the influences of time at sea, headwater amphibians are resident, long-lived, and numerous in healthy streams. Consequently they can provide a direct metric of stream health and an excellent surrogate for salmonids because they occupy a higher position in the stream continuum such that when headwater processes are managed to promote thriving amphibian populations, the downstream fish and their habitats will also likely thrive.

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] USDA Forest Service, Pacific Southwest Experiment Station, Redwood Sciences Laboratory, 1700 Bayview Drive, Arcata, CA 95521, (707) 825-2956. email: Amphnhwelsh@fs.fed.us

SESSION 7

Silviculture



Modeling Coast Redwood Variable Retention Management Regimes[1]

John-Pascal Berrill[2] and Kevin O'Hara[3]

Abstract

Variable retention is a flexible silvicultural system that provides forest managers with an alternative to clearcutting. While much of the standing volume is removed in one harvesting operation, residual stems are retained to provide structural complexity and wildlife habitat functions, or to accrue volume before removal during subsequent stand entries. The residual overstory trees and the new understory cohort will have different growth rates, and should therefore be modeled as distinct stand components. We used the redwood MASAM (multiaged stocking assessment model) to predict the growth of overstory and understory cohorts within pure coast redwood *(Sequoia sempervirens)* stands managed under dispersed variable retention management regimes. A range of overstory densities were simulated, and compared with an even-aged stand in terms of stand growth, and the growth and yield of each stand component. Results showed that overstory density had a minor influence on stand volume production, and a major influence on volume increment within overstory and understory cohorts. As overstory density increased, less growing space was available to the understory cohort. Too few data were available to model a combination of redwood and Douglas-fir. The redwood MASAM does not consider the spatial arrangement of stems, preventing comparison of aggregated and dispersed variable retention.

Key words: coast redwood, growth and yield, leaf area, variable retention

Introduction

The variable retention harvest system is a flexible silvicultural method that offers some continuity of forest ecosystem processes and structural diversity. It may represent a viable alternative to clearcutting in young coast redwood forests of north coastal California. The objective of this study was to predict coast redwood stand growth and yield under variable retention management. The redwood MASAM (multiaged stocking assessment model) was used to demonstrate the influence of density, one of the major variables that define a variable retention harvest system, on growth and yield.

The variable retention harvest system has been defined as "an approach to harvesting based on the retention of structural elements or biological legacies (trees, snags, logs, and so forth) from the harvested stand for integration into the new stand

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Division of Ecosystem Sciences, Dept. of Environmental Science, Policy and Management, 137 Mulford Hall, University of California, Berkeley, CA 94720-3114, (510) 643-2025. email: pberrill@nature.berkeley.edu

[3] Division of Ecosystem Sciences, Dept. of Environmental Science, Policy and Management, 137 Mulford Hall, University of California, Berkeley, CA 94720-3114, (510) 642-2127. email: ohara@nature.berkeley.edu

to achieve various ecological objectives…the major variables in the variable retention harvest system are types, densities, and spatial arrangement of retained structures; aggregated retention is the retention of structures or biological legacies as (typically) small, intact forest patches within the harvest unit; dispersed retention is the retention of structures or biological legacies in a dispersed or uniform pattern" (Helms 1998). The residual overstory trees and the new understory cohort initiated by the partial harvest treatment will occupy different amounts of growing space, grow at different rates, and should therefore be modeled as distinct stand components.

The multiaged stocking assessment model for coast redwood stands (redwood MASAM) was developed with data collected from young coast redwood (*Sequoia sempervirens* (D. Don) Endl.) and Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirbel) Franco) trees on Jackson Demonstration State Forest, Mendocino County, north coastal California (Berrill 2003). The model was developed with breast-height increment core data consisting of age (20 to 100 years), diameter growth and sapwood area data. Tree leaf area, the key explanatory variable, was predicted from sapwood area, and regressed against tree volume increment and basal area data (Berrill 2003).

The redwood MASAM model predicts volume increment and tree size of distinct stand components (species within canopy strata) in multiaged stands. Growth and yield is predicted over one even-aged rotation or one cutting cycle under multiaged management. The model user must define the density of each species within each canopy strata, and the growing space (as represented by projected leaf area) allocated to, or occupied by each stand component at harvest. All or part of each stand component is harvested at the end of the cutting cycle (the time between partial harvests under multiaged management), when stand leaf area index (LAI) returns to the user-defined maximum. Volume increment is predicted from average tree leaf area in each cohort, which is assumed to increase linearly over the cutting cycle *(fig. 1)*.

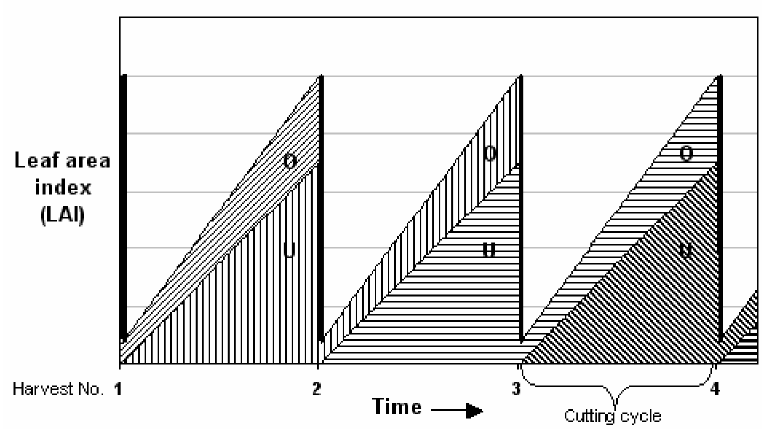


Figure 1—Schematic diagram of stand dynamics under variable retention management, showing theoretical overstory (O) and understory (U) leaf area index (LAI) over time between successive partial harvests. Harvesting involves removal of all overstory stems, and understory stems not needed to replace overstory. Each cohort has unique shading, showing that some understory stems are retained during partial harvesting at the end of each cutting cycle, becoming overstory stems over the next cutting cycle. This example depicts approximately 10 percent retention. Note: LAI increase assumed linear over the cutting cycle, however tree size and volume growth may not be linear.

O'Hara and others (2003) used the ponderosa pine MASAM to model multiaged ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) stand growth, including management for pre-settlement structures with numerous large overstory trees and sparse regeneration. The main advantages of the MASAM approach are that (a) growth of each stand component is predicted separately in accordance with the different growth rates and growing space availability of each component, and that (b) growth projections are constrained within the biologically realistic limits of maximum stand LAI.

The redwood MASAM does not consider the spatial arrangement of stems, preventing comparison of aggregated and dispersed variable retention. In this study, the redwood MASAM was used to evaluate the influence of overstory density on growth and yield under dispersed variable retention management, using tree size, leaf area and LAI data from Jackson Demonstration State Forest as user-defined model inputs.

Methods

Growth and yield under the variable retention harvest system was simulated using the redwood MASAM and data from sample plots on JDSF. Partial cutting in a pure even-aged redwood stand leaving up to 40 trees per acre (tpa) was simulated. A schematic diagram depicts the even-aged redwood stand before partial harvest, and early and late in the cutting cycle under variable retention management *(fig. 2)*. Clearcutting at age 50 was also simulated for comparison. These simulations demonstrated the influence of overstory density under variable retention management on a 50-year even-aged rotation or variable retention cutting cycle. Diameter and age data from overstory trees in 46 sample plots on JDSF gave indicative estimates of dominant redwood tree size and age for each simulation.

The influence of overstory density was demonstrated by simulating a range of overstory densities while holding constant the harvest age and amount of growing space allocated to overstory and understory cohorts. This test assumed that trees would grow at the same rate when allocated the same amount of growing space, as represented by leaf area at the end of the cutting cycle.

The MASAM model was used to allocate growing space to emergent overstory trees at each density level tested (0, 5, 10, 20, 30, 40 tpa). The overstory trees were allocated enough growing space (represented by leaf area) at the end of the cutting cycle to attain 44 inches dbh. The remaining growing space was allocated to the understory cohort. The density of this cohort was adjusted until MASAM predictions showed that an average diameter of 24 inches could be attained once the allocated growing space was completely occupied at the end of the 50-year cutting cycle. Stand density index (SDI) was calculated as the sum of cohort SDIs obtained from MASAM predictions of average tree size at the end of the cutting cycle and the user-defined cohort density. Cohort SDI (SDI_C) was calculated for cohort *i* such that $SDI_{Ci} = D_{Ci}\,(Q_{Ci}/10)^{1.6}$ where D_{Ci} was the density and Q_{Ci} the quadratic mean diameter for cohort *i* (O'Hara and Valappil 1999).

The MASAM model predicted average cohort and stand volume increment over the cutting cycle at each level of overstory density tested. Harvest volume was calculated from harvested density and average stem volume at harvest for each cohort. Stem volume was predicted from a one-foot stump height to a five-inch diameter top using a volume equation for young redwood (Wensel and Krumland 1983) and coefficients for JDSF (Griffen 1986) assuming a 100 ft site index (50 year base age; Wensel and Krumland 1986).

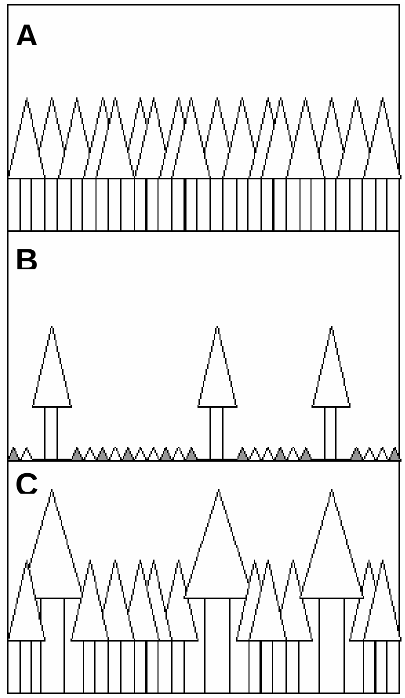


Figure 2—Schematic diagrams of even-aged stand (A) converted to multiaged stand managed under variable retention, early (B) and late (C) in the cutting cycle. Shading denotes removal of unwanted regeneration during pre-commercial thinning (PCT) of new understory cohort initiated after partial harvest removed most overstory stems (B). Crowns expand between successive partial harvests to re-occupy available growing space (C). Stems retained during partial harvesting have more leaf area and thus occupy more growing space per stem over the cutting cycle than trees in the understory cohort.

Results

The linear regression fitted to maximum tree diameter data indicated that widely-spaced emergent redwood trees could attain an average diameter of approximately 44 inches by age 100 years *(fig. 3)*. The data suggested that a second cohort growing beneath the emergent overstory trees could attain approximately 24 inches diameter by age 50, provided that these trees were thinned (for example, *fig. 2b*) to give each tree adequate growing space to maintain dominance or codominance within the cohort until harvest *(fig. 3)*.

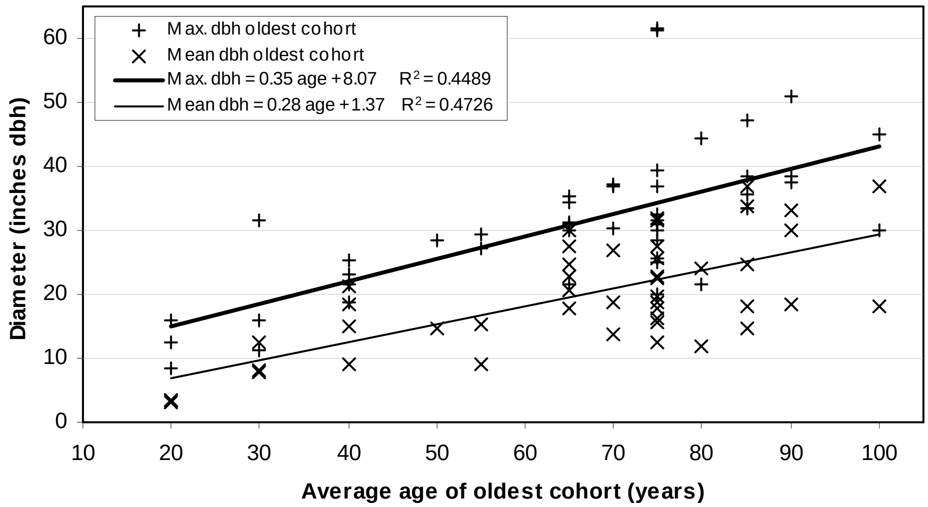


Figure 3—Average and maximum diameter (inches dbh), and average breast-height age (years) of the oldest cohort in 46 sample plots on JDSF. Note: density of oldest cohort ranged from 100-2600 tpa (age 20 to 40 yrs) and 18 to 600 tpa (age 50 to 100 yrs).

When more overstory trees were retained during partial harvesting of the even-aged redwood stand at age 50, less growing space was available to the understory cohort initiated by harvesting. Across the range of overstory densities tested, precommercial thinning in the understory cohort reduced density to leave the same amount of growing space for each stem, in other words, higher overstory densities resulted in lower densities in the understory cohort, while stand-level SDI remained approximately equal *(table 1)*.

Table 1—*Component and stand-level density and predicted stand density index (SDI) for variable retention management regimes with a 50-year cutting cycle. Stand harvested at end of cutting cycle (ECC) once LAI attained 14 ft^2/ft^2, yielding overstory stems with average 44-inch dbh and 7600 ft^2/tree leaf area, and 24-inch dbh stems in understory cohort with 2750 ft2/tree leaf area at harvest. Dominant heights of 150 ft and 100 ft at harvest for overstory stems and understory cohort respectively assuming 100-ft site index (dominant height at breast-height age 50 years).*

Density (trees per acre)			**Harvest SDI (ECC)**		
Overstory	Understory	Stand	Overstory	Understory	Stand
0	225	225	0	910	910
5	210	215	53	853	906
10	200	210	107	798	905
20	170	190	213	683	897
30	140	170	320	569	889
40	110	150	427	454	881

Predictions of cohort volume increment over the cutting cycle showed that higher overstory densities resulted in marginally greater stand volume increments, higher overstory cohort increment and lower volume increment in the understory cohort. The same trends were more pronounced when harvest volumes were calculated for each cohort across the range of densities tested *(fig. 4)*. Overstory cohort harvest volume increased relatively more than volume increment because among fewer stems present in the understory, more were retained as overstory trees. At the end of the 50-year cutting cycle, the large overstory tree crowns occupied approximately three times as much growing space (in terms of leaf area) as individual crowns of understory trees.

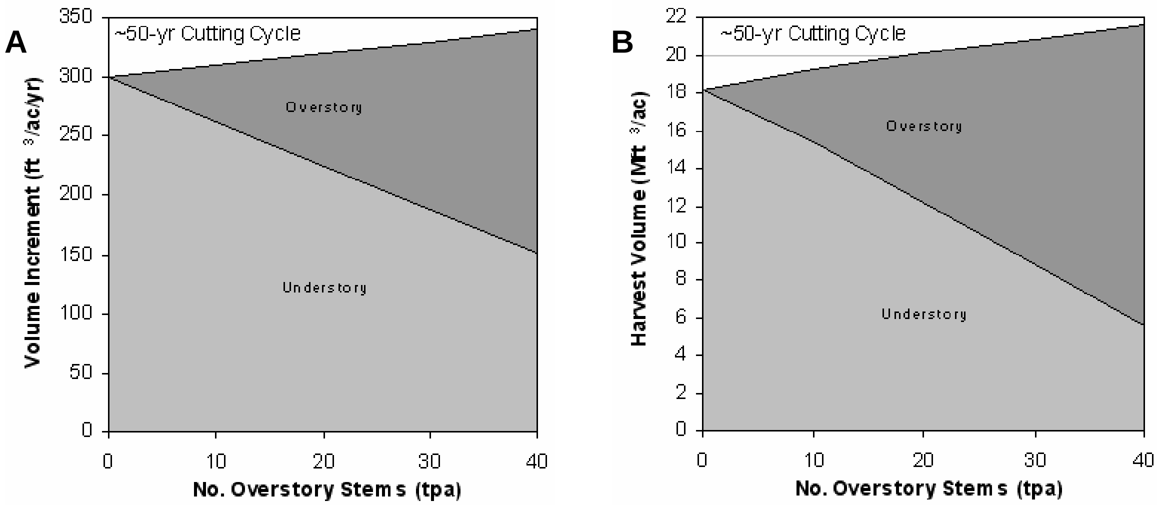


Figure 4—Redwood stand component (overstory and understory cohorts) (A) average annual volume increment in cubic feet per acre and (B) approximate harvest volume (ft^3/acre÷1000) under variable retention management on a 50-year cutting cycle. Site index = 100ft. Maximum stand LAI = 14 ft^2/ft^2. Average tree dimensions at end of cutting cycle, for (*i*) overstory trees: 44-inch dbh, 150ft height, projected leaf area = 7600 ft^2/tree; and (*ii*) understory trees: 24-inch dbh, 100ft height, leaf area = 2750 ft^2/tree.

The emergent tree volume increment model in MASAM predicted lower growth efficiency (volume increment per unit of leaf area) than the main canopy model used to predict the volume increment of the understory cohort with smaller branches and crowns. However, overstory volume increment over the cutting cycle was still greater per unit of occupied growing space at the end of the cutting cycle because the overstory trees had considerable leaf area (and thus produced considerable volume) at the beginning of the cutting cycle. Conversely, the new understory cohort initiated by harvesting was assumed to have no leaf area at the beginning of the cutting cycle, and was therefore predicted to produce little volume early in the cutting cycle.

Discussion

Coast redwood tree and stand volume growth were positively correlated with tree leaf area and stand LAI respectively on Jackson Demonstration State Forest (Berrill 2003). The relationship between leaf area and volume production explains greater productivity predicted at higher overstory densities in coast redwood stands under variable retention management. By retaining trees (and therefore leaf area) at the end of the cutting cycle, some level of volume production is sustained continuously over successive cutting cycles *(fig. 1)*. By contrast, even-aged stand volume production ceases when all leaf area is removed during clearcutting and resumes slowly during the next rotation as stand leaf area develops. This difference explains the lower stand volume production predicted under even-aged management than under variable retention, where the productivity advantage of maintaining some continuity of growing space occupancy and canopy cover increases as more trees are retained at harvest.

Dagley and O'Hara (2003) compiled density data from old growth redwood stands, reporting overstory densities ranging from about 20 to 150 tpa. These large old growth stems in the overstory probably occupied most available growing space. If, for example, 25 trees on one acre of old growth forest collectively carried $15ft^2/ft^2$ of leaf area, the average crown would have 26,136 ft^2 of projected leaf area. Similarly, if 50 trees collectively carried $15ft^2/ft^2$ of leaf area, the average crown would have 13,068 ft^2 of projected leaf area, although the larger stems probably have much more. The largest leaf area estimate for second growth trees on JDSF was 12,050 ft^2. It follows that around 55 trees of this size would also occupy most available growing space on any given acre if the upper limit of available growing space was assumed to be $15ft^2/ft^2$ of leaf area (a stand LAI of 15). When overstory trees occupy most available growing space, the growth of understory cohorts would be restricted. Thus, care must be paid to design structures with adequate end-of-cutting cycle growing space for all cohorts. If harvesting was delayed, or ultimately if residual overstory trees were retained over successive cutting cycles, leaf area in subordinate cohorts would be ceded to the overstory as a result of competition for limited growing space.

The redwood MASAM was formulated to predict growth and yield in pure stands and mixtures of redwood and Douglas-fir. However, the sampling strategy used for MASAM model development did not capture many Douglas-fir trees. Thus, while the regressions within the Douglas-fir component of the model were correctly estimated given the data at hand, there is much uncertainty around the estimates leading to uncertainty in predictions of growth and yield. It must be assigned a high priority for validation and revision once more data are obtained. The results of the simulations presented herein depended largely on accurate estimates of average tree size at harvest *(fig. 3)*. Too few data were available to obtain reliable estimates of average Douglas-fir tree size at harvest. MASAM predictions are also sensitive to changes in stand LAI at harvest. In this study, maximum LAI was defined as $14ft^2/ft^2$, slightly below the largest estimate ($14.9ft^2/ft^2$) obtained in 42 sample plots on Jackson Demonstration State Forest (Berrill 2003). More information on residual overstory tree mortality and maximum LAI within species mixtures is also required before mixed stand development can be simulated with any confidence. The redwood MASAM model could also be improved by adding components for other tree species in coast redwood forests, for example, tanoak *(Lithocarpus densiflorus)*.

The redwood MASAM is a stand component-level model that does not explicitly consider the spatial arrangement of stems. The results presented herein assumed a dispersed form of variable retention. Aggregated groups of residual overstory trees should occupy less growing space per stem at harvest, leaving more growing space for the understory cohort. This effect could theoretically be modeled with MASAM by reducing growing space allocation to overstory stems that will be crowded and competing for growing space within aggregated groups, but would have no scientific basis without information on the spatial patterns and growing space occupancy of aggregated groups of trees. More data from groups of residual overstory stems are needed to quantify this response, which likely differs between interior and edge tree positions, or as a function of proximity to the group edge. In theory, stands managed under aggregated retention should produce less volume from overstory stems and more from the understory cohort than stands managed under dispersed retention, for a given density and cutting cycle length.

The influence of density in the overstory cohort on stand component volume increment and harvest volume will depend on the size, condition and spatial arrangement of the overstory stems. Smaller overstory stems and tree crowns will produce less volume, but occupy less growing space, leaving more growing space for the understory cohort. More data are needed to understand growth efficiency among stump sprouts sharing root systems, within groups of aggregated retention trees, and within species mixtures. The influence of the amount and spatial arrangement of overstory leaf area on growth of trees in the understory cohort must also be assessed over the entire cutting cycle under variable retention management.

Conclusion

The redwood MASAM simulations showed that overstory density was predicted to have a minor influence on stand volume production, and a major influence on volume increment of the overstory and understory cohorts. As overstory density increased, less growing space was available to the understory cohort. Additionally, fewer stems were harvested from the understory cohort since more stems were retained as overstory trees over the next cutting cycle. The flexibility of the MASAM approach allowed simulation of differing levels of growing space occupancy by overstory and understory cohorts under a range of dispersed variable retention management regimes. The user-defined maximum stand LAI constrained model projections within realistic biological limits. More data are required before growth and yield under aggregated variable retention, and within species mixtures can be predicted with any confidence.

Acknowledgements

The redwood MASAM model was developed with data collected from even-aged and multiaged stands on Jackson Demonstration State Forest, funded by California Department of Forestry and Fire Protection (CDF) Research Grant No. 8CA99255. The authors gratefully acknowledge the support and advice of CDF staff, especially Bill Baxter and Marc Jameson.

References

Berrill, J.P. 2003. **Predicting multi-aged coast redwood stand growth and yield using leaf area allocation.** Berkeley, CA: University of California. M.S. thesis.

Dagley, C.M.; O'Hara, K.L. 2003. **Potential for old forest restoration and development of restoration tools in coast redwood: a literature review and synthesis.** Report prepared for the Save the Redwoods League.

Griffen, J. 1986. **Fall-and-buck study results: volume equations for trees on Jackson State Forest.** JDSF Newsletter 22: 1-3.

Helms, J.A., ed. 1998. **The dictionary of forestry.** Bethesda, MD: The Society of American Foresters; 210 p.

O'Hara, K.L.; Valappil, N.I. 1999. **Masam—a flexible stand diversity management model for meeting diverse structural objectives in multi-aged stands.** Forest Ecology and Management 118: 57-71.

O'Hara, K.L.; Valappil, N.I.; Nagel, L.M. 2003. **Stocking control procedures for multi-aged ponderosa pine stands in the inland northwest.** Western Journal of Applied Forestry 18(1): 5-14.

Wensel, L.C.; Krumland, B. 1983. **Volume and taper relationships for redwood Douglas fir, and other conifers in California's north coast.** Bulletin 1907. Berkeley, CA: University of California, Division of Agriculture and Natural Resources, Agricultural Experiment Station; 39 p.

Wensel, L.C.; Krumland, B. 1986. **A site index system for redwood and Douglas-fir in California's north coast forest.** Hilgardia 54(8): 1-14.

Holter Ridge Thinning Study, Redwood National Park: Preliminary Results of a 25-Year Retrospective[1]

Andrew J. Chittick[2] and Christopher R. Keyes[3]

Abstract

Redwood National Park is comprised of large areas of overstocked stands resulting from harvest of the old-growth stands in the late 1940s to the 1970s. The Holter Ridge Thinning Study was initiated in 1978 to address this problem and examine the effects that thinning to varying spacing would have on forest development. Densities following thinning in 1979 ranged from 150 to 790 stems/acre with the controls ranging from 1170 to 3410 stems/acre. Mortality from 1979 to 2003 showed a positive relationship to the number of stems/acre post thinning. The number of redwood sprouts was positively related to the number of redwoods thinned. The percent cover of herbaceous and shrub species showed a negative exponential response to stand density while the change in percent cover from 1984 to 2003 for the shrubs also had a negative response to stand density. Stand structure differed between the control plots and the thinned plots. The control plots showed no stratification of the canopy, but the thinned groups showed stratification into an upper-canopy composed of redwood and Douglas-fir and a lower-canopy of redwood and tanoak. After twenty-five years the thinned stands show the initial signs of a mature forest while the controls still exhibit intense competition.

Key words: density, old-growth, redwood, restoration, second-growth, structure, thinning

Introduction

Upland second-growth forests located within Redwood National Park exhibit a high potential value for restoration of old-growth associated characteristics. When Redwood National Park was expanded in 1978, over 51,000 acres out of 106,000 acres was cutover second-growth forest. These resulting second-growth forests now contain an extremely high density of trees, disproportionate amounts of Douglas-fir and tanoak, stagnated growth and development, and a homogeneous vertical and horizontal structure. Understory vegetation consisting of herbaceous, shrub, and tree species is generally excluded from dense second-growth forests within Redwood National Park with similar negative impacts on wildlife. The length of time until these second-growth forests re-establish new cohorts and become multi-age stands could be several hundred years. That silvicultural treatments can alter a stand's successional pathway to increase near-term old-growth-associated characteristics has gained increasing attention in recent years.

---

[1] This paper was presented at Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rhonert Park, California.

[2]Graduate Student, Dept. of Forestry and Watershed Management, Humboldt State University, P.O. Box 4761, Arcata, CA 95518. email: andy@mattole.org

[3]Assistant Professor of Forestry, Dept. of Forestry and Watershed Management, Humboldt State University, 1 Harpst St. Arcata, CA 95521. email: christopher.keyes@humboldt.edu

The Holter Ridge Thinning Study, located within Redwood National Park, is an experiment initiated in 1978 to determine what effects a low thinning with varying levels of stand densities has on the growth and composition of overstory trees (Veirs 1986). Its purpose is to determine how restoration thinning can alter a successional pathway to more closely resemble the pre-existing vegetation of old-growth forests. The study consists of a 200 acre, ~50-year old stand of second-growth coastal redwood (*Sequoia sempervirens* [D. Don.] Endl.)/Douglas-fir (*Pseudotsuga menziesii* [Mirbel] Franco)/tanoak (*Lithocarpus densiflorus* [Hook and Arn.] Redh.). The stand was harvested in 1954 using the seed tree method; an average of 1 redwood seed tree per acre was left and the stand regenerated from natural seeding (Veirs 1986). Pre-treatment stand densities averaged more than 1000 trees/acre, with some plots having 3000 trees/acre, according to Veirs' (1986) initial observations in 1978. Second-growth redwood/Douglas-fir ratios were also observed to be 1:1 on more xeric sites and 12:1 on mesic sites. Old-growth stands nearby were found to be predominantly redwood with densities ranging from 10 to 35 trees/acre for redwood and one to four trees/acre for Douglas-fir—considerably different than the second-growth that is now present (Veirs 1986). Other associated tree species present include western hemlock (*Tsuga heterophylla* [Raf.] Sarg.) and madrone (*Arbutus menziesii* Pursh.), and understory shrubs consisting of salal (*Gaultheria shallon* Pursh.), rhododendron (*Rhododendron macrophyllum* D. Don), evergreen huckleberry (*Vaccinium ovatum* Pursh.), red huckleberry (*Vaccinium parvifolium* Sm.), and swordfern (*Polystichum munitum* [Kaulf.] C. Presl) (Veirs 1986).

The coastal redwood's longevity and ability to persist in shade, combined with the relative shade intolerance of Douglas-fir and fire exclusion, indicates that Douglas-fir can be expected to decrease in dominance over time (Oliver and Larson 1996, Roy 1966). Without the use of silvicultural activities or disturbance, the redwood/Douglas-fir ratio of second-growth forests and associated structure may eventually approach those of old-growth stands over thousands of years. However, the development of multi-age structures associated with old-growth redwood forests will be stagnated in the near-term as a result of decreased growth owing to high densities and disturbance exclusion (Franklin and others 2002). In stands with low initial densities, characteristics similar to a maturing forest may be achieved at a younger age than the dense stands (Franklin and others 2002). The primary limiting factor for understory vegetation in coast range forests is the amount of light reaching the forest floor which thinning can directly alter (Bailey 1996). Thinning of dense stands has been shown to increase species richness, diversity, density, and cover of understory vegetation (Bailey and others 1998, Bailey and Tappeiner 1998, Thysell and Carey 2001). The research indicates that understory vegetation possesses the ability to persist in severe shade following canopy closure and flourish upon opening of the canopy (Halpern 1988, Halpern and Franklin 1990, Halpern and Spies 1995).

Methods

This study is located in Redwood National Park on Holter Ridge in the headwaters of Lost Man Creek, a tributary to Prairie Creek, principally in the area known as the Holter Ridge Thinning Study. Map coordinates are Sections 20 and 21, T11N, R2E, Humboldt Meridian, Tectah Creek, California Quadrangle (approximately 41º, 18'N, 123º 57'W).

The thinning study established by Veirs (1986) consists of three treatments and a control with each treatment divided into two parts depending on conifer spacing (10 ft to 12 ft and 16 ft to 18 ft), treatment of hardwoods (10 ft to 12 ft with hardwoods cut or included in spacing), or treatment of the slash (10 ft to 12 ft with slash lopped or not lopped). Actual densities following thinning range from 150 to 790 stems/acre (17 ft to 7.5 ft spacing) in the thinned units with 1170 to 3410 stems/acre in the controls. Diameter cut limits were 18 inch dbh for redwood sprouts, 10 inch for free standing redwoods, and 12 inch for Douglas-fir. In all units the numbers of redwood stump sprouts were to be thinned to 30 to 50 percent of the dominant sprouts.

Four 1/10-acre circular plots were systematically established for each treatment subunit including the control for a total of 28 plots. The plots were arranged along two pre-determined bearing lines with two plots randomly located on that line. Each tree was measured for: species, diameter at breast height, diameter at one foot above ground (stump height), crown width in each cardinal direction, height to base of live crown, total tree height, and notice of any damage or anomaly. Trees cut during thinning were measured for stump diameter and regressions were used to reconstruct the stand prior to thinning. From each plot center two photographs were taken in each of four cardinal directions. Re-measurements were conducted in 1984 following the same methods and included percent cover of the understory vegetation by species.

In 2003, using stem maps, azimuth, and distance, each tree was relocated and measured for: species, diameter at breast height, height to base of live crown, total tree height, and notice of any damage or anomaly. For understory vegetation percent cover by species was ocularly estimated into modified Braun-Blanquet cover classes with percent covers of: 0.001 to 0.01 percent, 0.01 to 0.1 percent, 0.1 to one percent, one to five percent, five to 25 percent, 25 to 50 percent, 50 to 75 percent, and 75 to 100 percent. Any seedling/sapling regenerated since thinning was counted by species with a seedling defined as a tree below 4.5 inches in height and a sapling as over 4.5 inches.

Regression analysis was used to predict the relationship of the independent variable in question to density immediately following thinning (stems/acre). In certain cases the relationship fitted best with an exponential curve or logarithmic transformation.

Large discrepancies existed between the initial intended treatments and the resulting densities following treatment. In order to observe differences in stand structure between densities I created three groupings based on density: Control, Mid-Density, and Low-Density.

Results

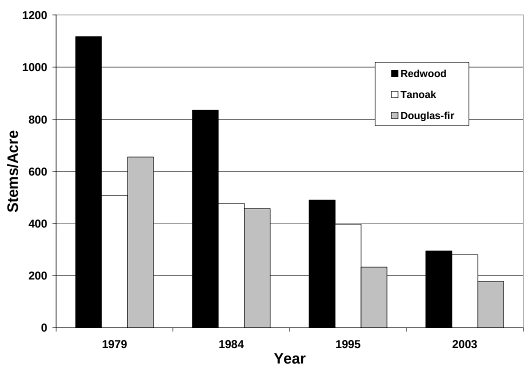
Several aspects of stand description were used to analyze the development of the stands over time: mortality, sprouting, understory vegetation, and stand structure. Densities for 1979 and 2003 for all stems and stems >4 inch dbh for control, mid-density, and low-density plots are given in *table 1*.

Table 1—*Stems/acre for control, mid-density, and low-density by year and diameter.*

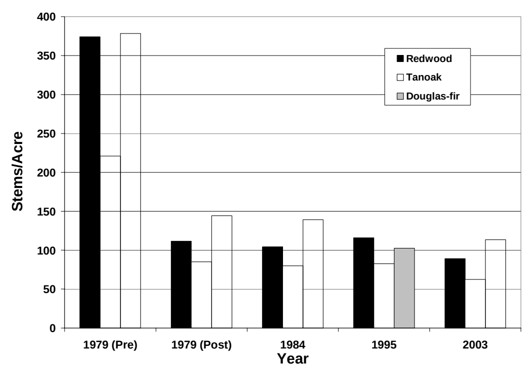
	Stems/Acre (1979)	Stems/Acre >4 inch dbh (1979)	Stems/Acre (2003)	Stems/Acre >4 inch dbh (2003)
Control	2050	552	870	560
Mid-Density	475	302	350	313
Low-Density	200	167	200	183

Mortality

The change in species composition in the control plots is marked by a large reduction in the number redwoods and Douglas-fir relative to tanoak *(fig. 1a)*. In 1979 redwood had 1110 stems/acre where Douglas-fir and tanoak had 655 and 507 stems/acre respectively, but by 2003 they had 295, 178, and 280 stems/acre respectively. The change in species composition for the low and mid-density plots is given in *figure 1b*. While redwood and Douglas-fir were equal in density prior to thinning, immediately following thinning redwood was slightly less than Douglas-fir (112 and 144 stems/acre respectively). This species composition in the thinned stands was then maintained throughout the study period. The number of stems/acre post thinning accounted for 63 percent of the variation in total mortality *(fig. 2)*.



a)



b)

Figure 1—Change in stems/acre by species for (a) Control plots and (b) Low- and Mid-Density plots.

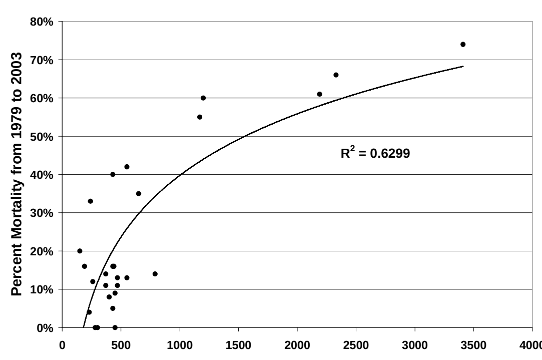


Figure 2—Regression of percent mortality for all species on all plots from 1979 to 2003 and density (stems/acre) using a logarithmic transformation.

Sprouting

The sprouting response of redwood from thinning ranges from zero to 3160 sprouts/acre while the number of redwoods thinned ranges from zero to 970 stems/acre. The largest ratio of the number of sprouts to number of redwoods thinned was 5:1. The number of redwoods thinned/acre accounted for 45 percent of the variation in the number of sprouts/acre *(fig. 3)*. The sapling to seedling ratio had a median value of 2:1 and a range from 5.2:1 to 0.4:1. Average height of saplings was estimated at eight to 12 ft.

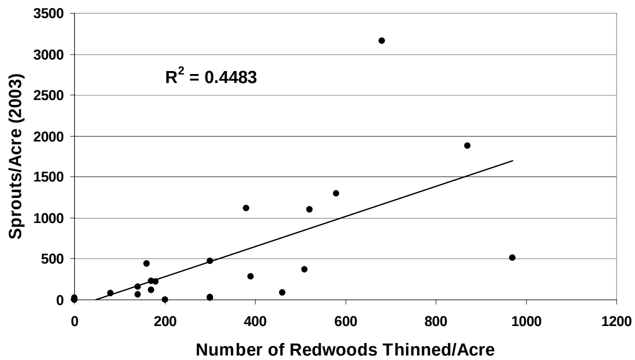


Figure 3—Regression of the number of redwoods thinned/acre in 1979 and the number of sprouts/acre in 2003.

Understory Vegetation

Percent cover of understory herbaceous and shrub species when compared to stems per acre post-thinning yielded an R^2 = 0.77 *(fig. 4)*. Herbaceous and shrub vegetation on the control plots ranged from zero to 20 percent cover but all of the thinned plots had at least 45 percent cover with a high of 95 percent cover. The percent change in the shrub vegetation from 1984 to 2003 is shown in *figure 5*, with the number of stems/acre post thinning accounting for 51 percent of the variation. For the control plots there was either no change or a decrease over the 25-year period while the thinned plots showed increases of up to 75 percent from 1984 levels.

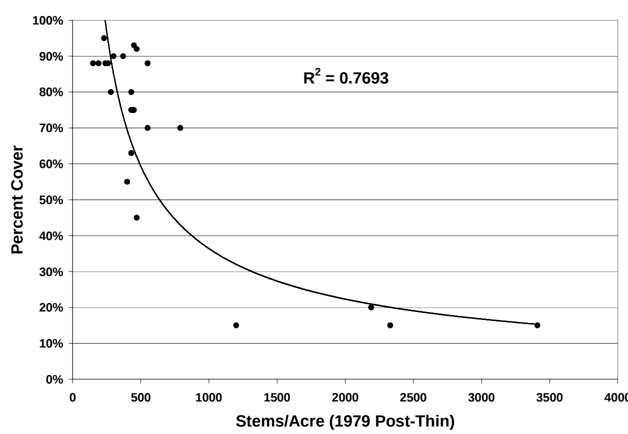


Figure 4—Regression of percent cover of herbaceous and shrub species in 2003 and density (stems/acre) with an exponential transformation.

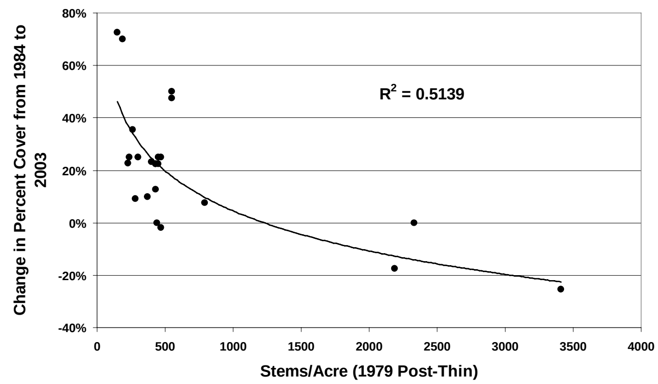


Figure 5—Regression of change in percent cover of shrubs from 1984 to 2003 and density (stems/acre) with a logarithmic transformation.

Stand Structure

Diameter distributions for all densities are given in *figure 6*. The Control plots in 1979 *(a)* exhibit a reverse-J distribution but in 2003 *(d)* there are a large number of trees in the <10 inch dbh classes with relatively few dominants. This is seen in the height distribution for 2003 *(fig. 7a)* where there are a large number of tanoak and redwood that comprise the suppressed/intermediate classes and the dominant/co-dominant classes are mostly Douglas-fir. There is no clear stratification by height class for the Controls.

The diameter distribution for the Mid-Density plots immediately following thinning *(fig. 6b)* show a gradation of redwood and tanoak in the lower classes to redwood and Douglas-fir in the larger classes. In 2003 *(e)* they show a similar but more pronounced stratification by species with Douglas-fir having a normal distribution in the upper diameter classes and a greater range than the control. The height classes in *figure 7b* show two distinct strata: a lower-canopy strata consisting of tanoak and redwood ranging from 10 feet to 70 feet in height and an upper-canopy strata with the majority of those being Douglas-fir followed by redwood.

The Low-Density plots show a single-modal distribution following thinning *(fig. 6c)* but are bi-modal in 2003 *(f)*. The first mode consists mostly of tanoak and redwood with significant portions of in-growth of sprouts into those diameter classes. The second mode is comprised of Douglas-fir and redwood and consists of dominant/co-dominants. The height distribution *(fig. 7c)* shows a strong component of Douglas-fir and redwood dominants in the 80 feet to 120 feet range with tanoak and redwood dominating the 10 feet to 60 feet strata.

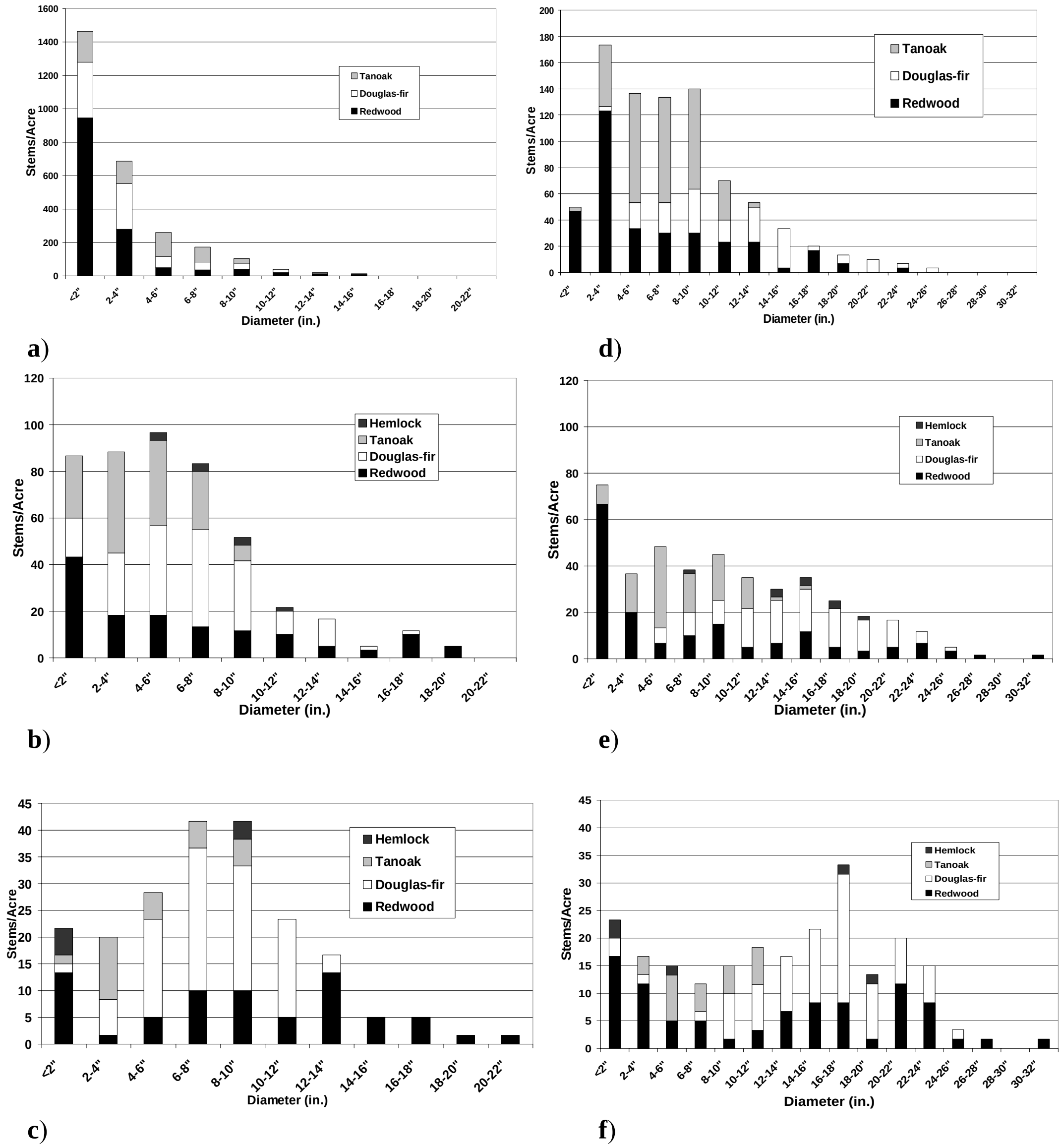


Figure 6—Diameter distributions in 1979 (post-thinning): (a) Control (b) Mid-Density (c) Low-Density and in 2003: (d) Control (e) Mid-Density (f) Low-Density.

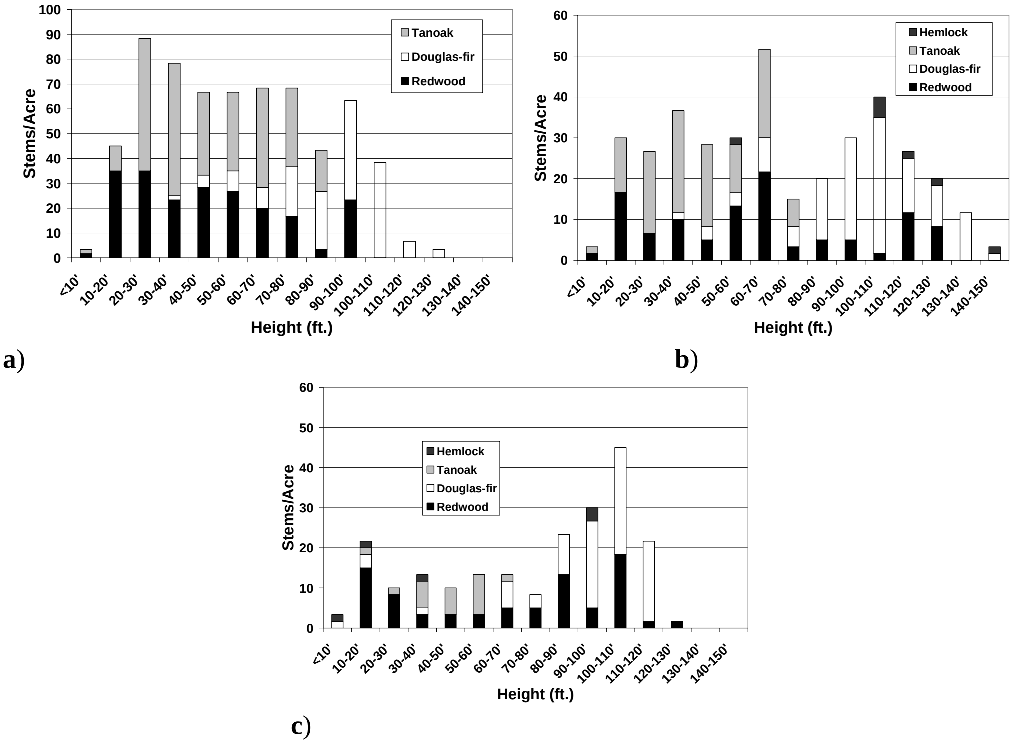


Figure 7—Height distributions in 2003: (a) Control plots (b) Mid-Density plots (c) Low-Density plots.

Discussion

Mortality

The reduction in the numbers of redwoods relative to Douglas-fir and tanoak in the control plots from 1979 to 2003 is due partly to the large number of redwoods comprising the <2 inch dbh class. In the thinned plots, species composition remains unchanged from 1979 to 2003. One of the stated goals in the thinning study was to increase the relative abundance of redwood; the thinning did not accomplish the goal as redwood decreased relative to Douglas-fir and tanoak immediately following the thinning.

Sprouting

Sprouting response of redwood from thinning indicates that there is a high possibility for multiple sprout shoots to persist in the understory with little growth in a moderately-dense overstory canopy. The only observed redwood sprouts to make it into the >4 inch dbh class were on the plots with both large gaps in the canopy and a small number of sprouts competing with each other; otherwise the sprouts had minimal height growth due to competition from the overstory canopy or among shrubs and other sprouts.

Understory Vegetation

Given the exponential nature of the understory vegetation in its relationship to density of overstory trees, it follows that the changes in cover over time should show a similar response. The control stands have fully passed into competitive exclusion of the understory and are expected to continue for the next 50 to 100 years (Franklin and others 2002). Currently, there are still shrubs present in the controls (~20 percent cover) that can possibly expand into any gaps that open in the canopy. In the thinned stands the understory vegetation has received enough light to expand and maintain itself. Canopy closure is not yet complete in most stands and the understory should persist indefinitely.

Stand Structure

The Control plots show a canopy without any definite strata. The upper canopies (>80 feet) of the Mid- and Low-Density stands have similar height distributions and species composition consisting of Douglas-fir and redwood. The lower canopies (<70 feet) differ between the two densities in the numbers of redwood and tanoak comprising those strata (Mid-Density has 2.5 times the number of stems as the Low-Density). The Low-Density appears to have a simple structure consisting of an overstory composed of dominant trees with very few lower canopy trees. In contrast, the Mid-Density is dominated by mid- and lower-canopy trees that provide for a more diverse structure. As these stands progress in the near future, little ingrowth into the lower canopy is expected to occur until sufficient gaps in the canopy open. Stratification will continue to increase between the upper and lower canopies with redwood and tanoak in the lower strata and redwood and Douglas-fir in the upper strata. Overall, the diameter distribution at age 50 that most closely approximates that of the reverse-J distribution is that of the Mid-Density stands.

Conclusions

Twenty-five years after treatment, the thinned stands exhibit reduced density-dependent mortality compared to the controls, high numbers of sprouting redwoods that remain in the understory, a highly significant increase in the growth of understory vegetation, and a stratification of the thinned plots into two distinct canopy layers. Stand development in the controls remains in competitive exclusion and will so for the next 50 to 100 years while the thinned stands have moved into the maturation stage. The Holter Ridge Thinning Study achieved some of its goal of accelerating stand development towards simulating old-growth characteristics. The use of a low-thinning at the lowest densities resulted in a simple structure not characteristic of old-growth forests. At higher densities the structure was closer to old-growth but had densities considerably higher than the old-growth. The thinning did not significantly alter the species composition towards redwood.

References

Bailey, J.D. 1996. **Effects of stand density reduction on structural development in western Oregon Douglas-fir forests—a reconstruction study.** Corvallis, OR: Oregon State University. Ph.D. dissertation.

Bailey, J.D.; Mayrsohn, C.; Doescher, P.S.; St. Pierre, E.; Tappeiner, J.C. 1998. **Understory vegetation in old and young Douglas-fir forests of western Oregon.** Forest Ecology and Management 112: 289-302.

Bailey, J.D.; Tappeiner, J.C. 1998. **Effects of thinning on structural development in 40- to 100-year-old Douglas-fir stands in western Oregon.** Forest Ecology and Management 108: 99-113.

Franklin, J.F.; Spies, T.A.; Van Pelt, R.; Carey, A.B.; Thornburgh, D.A.; Berg, D.R.; Lindenmayer, D.B.; Harmon, M.E.; Keeton, W.S.; Shaw, D.C.; Bible, K.; Chen, J. 2002. **Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example.** Forest Ecology and Management 155: 399-423.

Halpern, C.B. 1998. **Early successional pathways and the resistance and resilience of forest communities.** Ecology 69: 1703-1715.

Halpern, C.B.; Franklin, J.F. 1990. **Physiognomic development of *Pseudotsuga* forests in relation to initial structure and disturbance intensity.** Journal of Vegetation Science 1: 475-482.

Halpern, C.B.; Spies, T.A. 1995. **Plant species diversity in natural and managed forests of the Pacific Northwest.** Ecological Applications 5(4): 913-934.

Oliver, C.D.; Larson, B.C. 1996. **Forest stand dynamics.** New York: McGraw-Hill.

Roy, D.F. 1966. **Silvical characteristics of redwood.** Res. Paper PSW-RP-28. Berkeley, CA: Pacific Southwest Range and Experiment Station, Forest Service, U.S. Department of Agriculture.

Tappeiner, J.C.; MacDonald, P.M. 1984. **Development of tanoak understories in conifer stands.** Canadian Journal of Forest Research 14: 271-277.

Thysell, D.R.; Carey, A.B. 2001. **Manipulation of density of *Pseudotsuga menziesii* canopies: preliminary effects on understory vegetation.** Canadian Journal of Forest Research 31: 1513-1525.

Veirs, S.D. 1986. **Redwood second growth forest stand rehabilitation study, Redwood National Park: Evaluation of 1978-79 thinning experiments.** Unpublished draft supplied by author. Redwood National Park, Arcata, CA; 34 p.

Managing Second-Growth Forests in the Redwood Region for Accelerated Development of Marbled Murrelet Nesting Habitat[1]

Jerry F. Franklin,[2] Andrew B. Carey,[3] Steven P. Courtney,[4] John M. Marzluff,[5] Martin G. Raphael,[6] John C. Tappeiner,[7] and Dale A. Thornburgh[8]

The Marbled Murrelet is a threatened seabird that utilizes old-growth forest stands and coniferous trees with very large branches as their nesting habitat. Suitable nesting habitat is currently one of the limiting factors for the species in the Coast Redwood region. Extensive second-growth forests exist within the region that will eventually provide habitat, but natural development of second-growth into suitable trees and branch systems may take several centuries. Consequently, there is interest in the potential of managing some of these second-growth stands so as to accelerate development of suitable nesting habitat.

We conducted a scientific review of issues associated with Marbled Murrelet nesting habitat, including the character of nesting habitat, the level and importance of murrelet egg and chick predation, patterns of forest development in the redwood region, and silvicultural principles for accelerating development of late-successional structural characteristics in second-growth forests, including creation of suitable nesting habitat.

Our review found that there is strong scientific evidence that (1) Nesting habitat is an important limiting factor for murrelet reproduction and (2) predation on eggs and chicks is an important factor limiting nest success. Hence, efforts to accelerate development of suitable nesting habitat in second-growth forests are warranted but must be carried out in a fashion that does not increaes rates of nest predation (in other words, densities or effectiveness of predator populations).

We concluded that there were excellent opportunities to accelerate the development of murrelet nesting habitat in highly productive second growth forests of the Coast Redwood region and developed a general silvicultural prescription to guide such efforts. In well-developed stands (for example, 50 to 70 years of age) we recommend a combination of (1) uniform thinning from below to reduce overall stand densities and (2) additional release of selected potential nest trees to stimulate

---

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] College of Forest Resources, University of WA, Seattle, WA. email: jff@u.washington.edu
[3] USDA Forest Service, Pacific NW Research Station, Olympia, WA.
[4] Sustainable Ecosystems Institute, Portland, OR.
[5] College of Forest Resources, University of Washington, Seattle, WA.
[6] USDA Forest Service, Pac NW Research Station, Olympia, WA.
[7] School of Forestry, Oregon State University, Corvallis, OR.
[8] School of Natural Resources, Humboldt State University, Arcata, CA.

the development of existing branches and initiation of new epicormic branch systems. This prescription should be implemented as uniformly as possible to avoid creating structural and compositional diversity and food sources that might attract additional predators. A topic that needs further exploration is treatment of crowns to stimulate development of retirated trunks and associated vertical branches or “arms,” since these structures seem to have particular value as nesting habitat.

Details of the proposed silvicultural prescription will be provided in the talk. It must, of course, be adapted to specific stand conditions, so significant variability in the prescription is to be expected both within and between stands.

Restoring Complexity to Industrially Managed Timberlands: The Mill Creek Interim Management Recommendations and Early Restoration Thinning Treatments[1]

Dan Porter,[2] Valerie Gizinski,[3] Ruskin Hartley,[4] and Sharon Hendrix Kramer[5]

Abstract

The Mill Creek Property was a commercial timberland acquired by the State of California to protect and restore local and regional ecological values and provide opportunities for compatible recreation. Interim Management Recommendations (IMR) were developed to guide protection, restoration, and public access of the Property until the California Department of Parks and Recreation (DPR) develops a General Plan. Using existing data as well as public and professional input, the IMR planning process identified management alternatives for forest restoration and other priority issues. Recommendations were based on spatial analysis of potential risks and benefits to resources.

The IMR identified 5,680 hectares (14,000 acres) of overly dense young coniferous stands needing restorative thinning to accelerate the development of late-successional forest characteristics and avoid the unnatural growth trajectories established by plantation–style forest management. Using public and private funds, a pilot project was designed and implemented to experimentally thin approximately 41 hectares (100 acres). A variable density thinning (VDT) prescription was used to lower tree densities and is expected to accelerate growth, increase stand level heterogeneity and adjust tree species composition. Tree growth, wildlife habitat and wildlife use are monitored against unthinned control areas using permanent plots. Results will inform the development of future prescriptions designed to restore late-successional forest characteristics.

Key words: California, forests, restoration, salmonids, silviculture, state parks

Introduction

Following over 100 years of timber extraction (Madej and others 1986), the Mill Creek Property (hereafter referred to as Mill Creek) located in Del Norte County, California, was purchased and transferred to the California State Park System on June

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Forest Ecologist. Save-the-Redwoods League, 114 Sansome Street, San Francisco, CA 94104. email: dporter@savetheredwoods.org

[3] Resource Ecologist. California State Parks, 121200 Highway 101, PO Box 7. Orick, CA 95555. email: Valerie_Gizinski@partner.nps.gov

[4] Director of Conservation. Save-the-Redwoods League, 114 Sansome Street, San Francisco, CA 94104. email: rhartley@savetheredwoods.org

[5] Senior Aquatic Ecologist, Stillwater Sciences, 850 G Street, Suite K., Arcata, CA 95521. email: Sharon@stillwatersci.com

4, 2002. Support for the 60 million dollar, 25,000 acre acquisition was based on its strategic location between existing preserves and the outstanding quality of its salmon-bearing streams. Included in the sale's terms and conditions is an agreement that the property's primary objective shall be to restore late successional forest conditions and the associated natural functions for the benefit of the areas' fish and wildlife.

Restoration of Mill Creek's forests, streams and wildlife will require decades of carefully planned resource work to ameliorate the effects of timber harvesting and associated road building. For example, most of the park's 410 km (255 mi) of roads and skid trails will need to be decommissioned or re-engineered to minimize impacts to listed fish species. Thousands of acres of young, even-aged, nearly monospecific forest plantations will need to be thinned and treated to jumpstart ecological processes important to the development of complex habitats and biological diversity. Opportunities for public recreation and education will need to be built around ongoing restoration while considering the public's safety.

Anticipating that the development of a General Plan for the Property would take several years, Save-the-Redwoods League and the California Coastal Conservancy spearheaded the development of Interim Management Recommendations (IMR) using public and private funds. Based on existing geographic information system (GIS) data, public input, and advice from over 60 resource professionals, the IMR identifies four priority management areas to focus on in the short term: (1) road management, (2) forest management, (3) aquatic and terrestrial habitat protection, and (4) public access. The IMR identifies emergency management actions that if not taken in the next 10 years could jeopardize Mill Creek's natural resources and constrain future management opportunities intended to meet the Property's primary objective.

Today, unnaturally dense stands of young trees dominate Mill Creek's landscape because commercial extraction of timber occurred over a relatively short period (1954 to 2000). Cleared areas were either re-seeded by neighboring trees or were planted at a high density with native conifers. A high level of annual precipitation, productive soils and the use of seed-tree and clearcut silviculture led to the rapid and dense re-vegetation of cleared areas. Use of pre-emergent herbicides favored the growth of commercial conifers (principally Douglas-fir) to the detriment of native shrubs and herb species. Because Douglas-fir is better able to colonize newly cleared areas, it often dominated sites before other native conifers such as coast redwood could establish, resulting in unnaturally dense, monospecific stands.

Studies of pre-settlement disturbance regimes in the northern redwood region suggest that catastrophic stand-replacing events were rare, and that stand-modifying disturbances such as low intensity ground fire, flooding and localized windthrow were more common (Sawyer and others 2000). Stand development following human induced stand-replacing events such as those caused by the repeated use of seed tree or clearcut silviculture therefore establish artificial growth trajectories that are rarely found in the natural ecosystem. Undesirable effects such as stagnated growth, unbalanced species compositions, altered fire regimes and outbreaks of insect and pathogens can be expected from the artificially dense conditions. Plot data from the study region in thinned and unthinned even-aged regenerated stands indicate that in some locations, high tree densities may preclude the development of conditions found in older forests for decades or centuries (Thornburgh and others 2000, Veirs and Lennox 1982). In coastal Douglas-fir forests high tree densities, even, narrow spacing and late thinning have been shown to delay canopy differentiation, decrease biocomplexity, and lead to unstable stands (Wilson and Oliver 2000).

Experimentally conducted, restoration–based variable density thinning may be used to redirect unnatural growth trajectories to improve short and long term ecological values while contributing to an adaptive forest management framework. While thinning young second growth forest plantations is relatively new for the purposes of conservation, its foundation is based on a growing number of carefully conducted studies. In the redwood region for example, thinning has been shown to accelerate the growth of coast redwood (Oliver 1994, Veirs 1986) and lead to the development of understory shrub and bird densities typical of old-growth forests (Menges 1994, Thornburgh and others 2000). In Washington's coastal forests, thinning experiments designed to induce heterogeneity into forest canopies have demonstrated the importance of biocomplexity to various biotic communities, including soil microorganisms, vascular plants, fungi, birds, small mammals and vertebrate predators. Over ten years of experimentally conducted variable density thinning in these forest plantations has demonstrated a short term (<5 years) increase of these taxa in thinned stands relative to unthinned stands (Carey 2003).

The long-term ecological effects of restoration-based variable density thinning are as yet unknown, particularly for older stands. Initial studies from the redwood region suggest that cryptic elements of biodiversity (arthropods) may be significantly affected by repeated stand management (Willett 2001) and that older relatively unmanaged areas, even at the single tree scale, can support a rich and unique fauna (Camann and others 2001, Mazurek and Zielinski 2003). These studies suggest that over time, decisions regarding restoration thinning will need to consider benefits and impacts to a wide array of native species at different scales. Young (11 to 20 years), dense even-aged regenerated stands, like those needing immediate treatment at Mill Creek, are less likely to have redeveloped such cryptic elements of biodiversity. Such organisms tend to require a degree of habitat complexity and stability that develops over many years.

Methods

The Study Area

Over 410 km (255 mi) of road exist on the Mill Creek Property, and most were built over a relatively short period of time (1958 to 1977) (Madej and others 1986). Road densities across the property average four km/km^2 (6.4 mi/mi^2). Approximately 97 km (60 mi) of road were temporarily decommissioned in 2001 to 2002 by the former owner, and have not received maintenance since. State Park geologists have determined these roads pose an immediate and significant risk to Mill Creek's aquatic resources.

The IMR analysis determined that approximately 69 percent of its area supports trees than are pole size or smaller (<28 cm; 11 inches) while only six percent of its area supports trees that would qualify as mature (>61 cm; 24 inches). Historically, this area supported trees ranging from one to four meters (three to 12 ft) in diameter as the predominant forest type. Today, less than 81 hectares (200 acres) of old-growth forest remains, most of which is located in five isolated stands.

Despite the Property's rapid and nearly complete transformation from old-growth to early successional habitat, its underlying geology, productive capacity, and proximity to existing preserves will facilitate relatively rapid habitat recovery. Habitat for 26 special status wildlife species exists and continues to develop at Mill Creek, including habitat for species such as the northern spotted owl, marbled murrelet and Pacific fisher. Mill Creek's productive streams support a diverse assemblage of special status fish species including coho, Chinook, and chum salmon as well as steelhead and coastal cutthroat trout. At least 15 vegetation series and over 300 vascular plant species are present, including as many as 52 sensitive plant species (SHN 2000).

The Interim Management Recommendations

The intent of the IMR was to synthesize and analyze existing resource information and based on this information, provide recommendations to DPR to guide interim management actions. The synthesis and analysis was also intended to provide a solid base from which DPR could prepare a General Plan for the park. Fortunately, the previous owner had collected and organized a large quantity of road, forest and stream data including sediment source locations and a detailed harvest history which was made available with the sale and transfer of the Property (Stimson Lumber Company 1998).

Scoping meetings and working groups were organized to solicit the input of resource professionals, experts from local universities and members of the public. Attendees identified management issues and defined the desired future conditions of Mill Creek. An analysis of opportunities and constraints and potential risks and benefits to resources was conducted using existing geographic data, and a systematic prioritization matrix linked to the geographic data. The analysis was then synthesized into a set of interim recommendations, suggestions for research and monitoring and recommendations for long-term restoration and management. This information was then checked against existing regulatory permitting requirements and thresholds to develop short-term interim management recommendations and a roadmap for more detailed planning and permitting.

Interim management objectives include promoting development of old-growth forest characteristics and minimizing the risk of catastrophic wildfire, both of which can be attained through silvicultural methods. Priorities for forest management during the interim period were established by identifying young second-growth forests where forest restoration would provide the greatest potential ecological benefits including buffering known old growth, protecting habitat of sensitive species, providing connectivity, and effectively reducing catastrophic wildfire risk. The IMR emphasized that for young even-aged regenerated stands, a narrow window of opportunity exists to realize the benefits of thinning before these stands enter the stem exclusion phase.

Data used for the analyses included an electronic vegetation database developed by Stimson Lumber Company to manage industrial forests and associated natural resources within their ownership; the database was substantially modified by Stimson to track stand age and silvicultural treatments. Database attributes used in this analysis include (1) date of stand birth (time since regeneration began) based on aerial photographs dating back to the 1950s, and (2) vegetation type based on aerial photograph interpretation and/or field mapping. In addition, another electronic database of silvicultural treatments and timing, including records of pre-commercial thinning (but not commercial thinning) was provided by Stimson Lumber Company.

Other data included buffers around marbled murrelet and northern spotted owl nesting sites, the regional surface fuel maps for California developed by the California Interagency Fuel Mapping Group, a 10-m Digital Terrain Model (DTM) to identify ridge crest positions where lightning strikes are most likely to occur, and buffers around public use areas, the mill site, picnic sites, adjacent to the primary road network and along the interior of the property boundary.

Opportunities were identified and priorities for management were established using a matrix. Indicators were assigned various scores based on their relative influence on the desired objective; higher scores reflected a greater risk or management priority *(table 1)*. The spatial distribution and coincidence of indicators across the property were analyzed over a 10-meter grid spacing using GIS. The cumulative score for each grid cell was then used to establish priority areas for interim management activities, such as thinning for ecological benefits and fuels reduction.

Table 1—*Stand level characteristics used to prioritize forest restoration. Landscape scale variables (connectivity) were used in combination these scores to provide a composite priority ranking for each polygon.*

Age class (years)	Precommercial thinning	Area	Score	Relative ranking
0-5	-	-	0	low
6-10	-	<3 ha	2	mod-high
6-10	-	>3 ha	3	high
11-20	unthinned	<3 ha	3	high
11-20	unthinned	>3 ha	4	very high
11-20	thinned	-	0	low
21-40	-	-	1	moderate
>40	-	-	0	low

Initial Projects

The pilot forest restoration project's planning and implementation generated useful information with respect to old forest structure that will help guide the design of future projects. A project-related literature review revealed that old redwood forest canopy densities are highly variable, ranging from 50 to 380 trees/ha (20 to 150 trees/ac) (Dagley and O'Hara 2003). Locally, historic timber plot data from three large ownerships indicate average old-growth tree densities of 79 trees/ha (32 trees/ac) (Hammon and others 1969). The same data indicates that for the Mill Creek Property as a whole, coast redwood was the dominant species in the old-growth condition, representing 70 percent of the standing volume whereas Douglas-fir only represented 27 percent of the overall volume. Today young, even-aged regenerated stands at Mill Creek 12 to 15 years of age can support 1,728 to 2,222 trees/ha (700 to 900 trees/ac) and have tree species compositions with an over-abundance of Douglas-fir. One stand for example, had a nearly inverse composition (76 percent Douglas-fir, 24 percent coast redwood) compared to the old-growth reference condition despite it being located on an lowland alluvial terrace.

Approximately 49 ha (120 ac) of overly dense high priority forest was selected for the pilot experimental thinning trial. Stands were selected based on the IMR analysis, and aerial photo interpretation; field reconnaissance was then used to find simple, even-aged stands with little to no apparent crown differentiation. The three sites selected were divided into a total of 18 treatment blocks, in which one of three treatments was applied *(table 2).* Target densities refer to the density of dominant trees expected to develop with time assuming a maximum mortality of 33 percent[6] over the vulnerable period in the stand's development. Target densities were based on a desire to adjust the dominant tree density to a level resembling old forest tree densities in the region, using one relatively heavy thinning. Such an approach, if successful, may be used in situations where road decommissioning will preclude future management access.

Table 2—*Variable density thinning treatment design.[1] Acreage column sums the treatment acreage for all three sites.*

Treatment name	Post treatment density	Target density	Acreage
High Density VDT	150 trees/acre	100 trees/acre	44.5
Low Density VDT	75 trees/acre	50 trees/acre	51.5
Controls (No Treatment)	Unchanged	NA	24
		Treatment acreage	96
		Total acreage	120

[1] Treatment design by O'Hara & Dagley, UC Berkeley Natural Resources Department.

Trees to be retained were selected using a randomization process and a species priority list. Within small (6.4 x 6.4 m [21 x 21 ft]) marking cells, a random number between zero and three was generated, corresponding to the number of trees to be retained (leave trees) within that cell. After the number of leave trees was determined, the tallest individual(s) of preferred species were marked for retention. Leave tree preference was based on the current under–representation of coast redwood and the over–representation of Douglas-fir. Species priorities were arranged as follows: (1) coast redwood; (2) western hemlock, grand fir, Sitka spruce, western red cedar, Port Orford cedar and red alder; (3) Douglas-fir, tanoak, madrone, and bay. The minimum retained tree height was set at approximately 1.4 m (4.5 ft), in order to retain enough redwood saplings to significantly alter the stand species composition. The pattern of retention was randomized to eliminate the uniform plantation pattern of live trees and to encourage a non-uniform pattern of future branch development. The latter structural goal may in time facilitate the development of a more diverse understory plant assemblage (Carey 2003).

Trees were thinned using chainsaws by the local California Conservation Corps (CCC) and a private sub-contractor. Felled trees were left lying on the forest floor, with the exception of roadside areas in which fuels were removed for a distance of 10 m (30 ft) from the road edge.

[6] This mortality estimate is an educated guess based on experience. Annual mortality figures for stands of this age class are not available. A 33 percent attrition rate is a conservative over-estimate of the expected mortality.

Prior to the commencement of thinning operations, wildlife habitat monitoring plots were installed by biologists from the California Department of Fish and Game, in a subset of treatment units (High, Low, and Control). Wildlife habitat elements as used in the California Wildlife Habitat Relations (CWHR) model were recorded, as was the presence or absence of small and large mammals using live traps and motion sensing cameras.

Following the completion of thinning, permanent tree growth and mortality monitoring plots were installed in each of the 18 treatment blocks. Trees per plot, species composition and diameters were measured to establish the baseline post–thinning conditions. These plots will be measured again on a five to 10 year monitoring schedule to track changes in growth and mortality over time.

Results

In the three and a half years since the acquisition, State Parks and its partners have initiated implementation of the IMR *(table 3)*. In addition to focusing on the highest priority resource problems, initial projects have sought to establish the technical and professional knowledge base needed to develop a long-term integrated ecological restoration plan for the entire 25,000 park. For example, data from a property-wide road sediment inventory is being combined with geologic and hydrologic information related to erosion potential to identify and prioritize those areas that pose the greatest risk to aquatic resources. Concurrently, data from a detailed forest inventory of young stands (recently completed) has been analyzed to provide a refined assessment (relative to the IMR) of thinning priorities. Together, these informational building blocks have been used to develop a five-year coordinated approach to watershed restoration.

Table 3—*A sample of priority interim management recommendations and current status. See the Interim Management Recommendations (Stillwater Sciences 2002) for a more detailed list of recommended actions.*

Interim Management Recommendation	Status: March 2006
Inventory and establish treatment priorities for existing road and log landings	DPR lead field inventory complete. Data entry and site-specific repair prescriptions being developed
Develop an inspection and maintenance schedule	Roads have been continually inspected and maintained every winter since the acquisition. A more detailed maintenance schedule is being developed
Decommission non-essential roads and treat associated hillslope erosion	Funding to decommission 17 – miles of high risk roads has been secured. Ca. 12 miles of decommissioning work conducted in the summer of 2004 and 2005.
Conduct field surveys to prioritize forest restoration projects	A streamlined forest inventory for unthinned stands 11 – 24 years of age has been designed, implemented and analyzed for ca. 5,500 acres. A prioritization matrix reflecting within stand conditions and the road removal schedule has identified 3,500 acres of high and very high priority forest needing restoration.
Restore second growth forests	100 – acres of overly dense plantation-like forest were thinned in the winter of 2003. An additional 400 – acres received treatment in 2004/2005. A five-year, 3,500 acre forest restoration project has been developed based on the recently completed forest inventory and in collaboration with local resource experts. The *Forest Ecosystem Restoration and Protection Project (FERPP)* is coordinated with the approved five-year road removal plan. An accompanying effectiveness monitoring plan has been developed to evaluate tree level and stand level responses to different restoration prescriptions.
Control the spread of plant pathogens	Localized eradication of infected trees has been conducted and seasonal equipment use limitations observed.
Establish special management areas	Isolated old-growth stands receive special consideration (setbacks / observance of critical periods) when conducting restoration projects.
Continue long-term salmonid monitoring	The 10 – year salmonid monitoring record has remained unbroken using public funds from the Department of Fish and Game. Funds secured to analyze population data and refine monitoring. White paper generated by Stillwater Sciences and the Mill Creek Fisheries program in collaboration with Humboldt State University.
Provide educational and service – learning opportunities	Summer lead docent tours have been conducted for the public every summer since the acquisition. More recently, winter salmon viewing and equestrian tours have been provided by DPR.

Given the recent completion of the pilot forest restoration project, only the pre-thinning habitat data and post-thinning tree counts are currently available. Pre-treatment wildlife habitat and wildlife use measurements indicate the treated stands are structurally simple, inhibit primary productivity and do not support rich or abundant wildlife populations. Stands were found to be dominated by a uniform monoculture of closely spaced Douglas-fir trees with canopy cover estimates ranging from 97 to 100 percent. The principal ground cover consisted of decomposing conifer needles (duff). Shrub and herbaceous cover is nearly absent, consisting of sparse collections of evergreen huckleberry and sword fern. Legacy structures are limited to logging slash, which includes pieces typically <51 cm (<20 inches) in diameter and stumps. Out of 176 live traps set, biologists captured a total of eight deer mice; no other small mammal species were noted. After more than 60 days, baited motion sensing wildlife cameras detected a single mule deer.

Preliminary inspection of the permanent tree growth and mortality plot data suggest that the two post thinning target densities were generally met *(table 4)* and that species composition was successfully shifted to include more coast redwood *(table 5)* (O'Hara and Dagley 2006)[7]. Where the pre-thinning condition consisted almost entirely of Douglas-fir, inter-planting of coast redwood may be needed to achieve a representative species balance.

Table 4—*Post treatment tree densities for three experimental thinning sites on the Mill Creek Property, Del Norte County.*

Site	Treatment	Trees per acre
Childs Hill	Control	657
Cougar Ridge		682
Moratorium		1357
Average		899
Childs Hill	High (Target – 150 TPA)*	166
Cougar Ridge		113
Moratorium		140
Average		140
Childs Hill	Low (Target – 75 TPA)*	78
Cougar Ridge		67
Moratorium		84
Average		76

*TPA – trees per acre

[7] Data adapted with permission from O'Hara and Dagley.

Table 5—*Percent composition by tree species from unthinned (controls) and post thinning sample plots.*

Childs Hill	***Controls***	***High density***	***Low density***
Redwood	3.3 percent	36.7 percent	18.1 percent
Douglas-fir	89.6 percent	59.6 percent	74.8 percent
Other	7.1 percent	3.6 percent	7.1 percent
Cougar Ridge			
Redwood	41.1 percent	93.8 percent	52.2 percent
Douglas-fir	55.0 percent	6.2 percent	38.1 percent
Other	3.9 percent	0.0 percent	9.7 percent
Moratorium			
Redwood	24.0 percent	67.9 percent	71.9 percent
Douglas-fir	33.2 percent	22.1 percent	20.4 percent
Other	42.9 percent	10.0 percent	7.8 percent

Discussion

In his conclusion to The Redwood Forest, Reed Noss presents recommendations for protection of the coast redwood forest: a globally significant ecosystem (Noss 2000). These include protection of landscapes with rare species and communities, better connectivity among late-successional redwood forests, increased protection of critical watersheds, and the restoration of areas of degraded redwood forests. The Mill Creek project embodies these recommendations and has the potential to serve as a model for restoring the integrity and resilience of California's low elevation coastal forest ecosystems, which have been simplified by years of intensive timber management.

The IMR process and resulting report has provided a valuable decision-making tool used to guide short-term, emergency management actions. It demonstrated how data collected for one purpose, in this instance commercial timber extraction, can be utilized to inform new management objectives, in this instance ecological restoration. In conjunction with DPR's strong commitment and a focused Advisory Committee, Mill Creek's restoration is not only underway, but has attracted the support of both public and private conservation interests. Implementation hurdles and cost estimates from the pilot projects have been used to build credible proposals for ongoing restoration, several of which are expected to receive funding in 2004.

Restoring complexity to industrially managed timberlands is an emerging field with a relatively short history. The Mill Creek Property presents a unique opportunity to develop and test forest restoration techniques at a large scale with a clear focus on forest ecology. In the coast redwood ecosystem, where the extent of primeval redwood forest is estimated to be four to five percent of historic levels, forest restoration promises to fulfill the acknowledged need to grow more old growth (Noss 2000) and assure the long-term viability of the redwood forest ecosystem.

References

Camann, M.A.; Lamoncha, K.L.; Jones, C.B. 2001. **Old-growth redwood forest canopy arthropod prey base for arboreal wandering salamander.** Unpublished report prepared for Save-the-Redwoods League.

Carey, A.B. 2003. **Biocomplexity and restoration of biodiversity in temperate coniferous forests: inducing spatial heterogeneity with variable-density thinning.** Forestry 76(2): 127–136.

Dagley, C.M.; O'Hara, K.L. 2003. **Potential for old forest restoration and development of restoration tools in coast redwood: a literature review and synthesis.** Prepared for Save-the-Redwoods League.

Hammon, A.; Jensen H.; Wallen A. 1969. **Timber inventory report**. Proposed Rellim Redwood Company allocation of the northern redwood purchase unit lands. Del Norte County, California. A report to the U.S. Dept. of Interior. 36 p. Map archived at HSU Library, Humboldt Room. Arcata, CA.

Madej, M.A.; O'Sullivan, C.; Varnum, N. 1986. **An evaluation of land use, hydrology, and sediment yield in the Mill Creek watershed, northern California.** Redwood National Park Research and Development Technical Report Number. Arcata, CA. 17.

Mazurek, M.J.; Zielinski, W.J. 2003. **The importance of legacy old-growth trees in the maintenance of biodiversity in commercial redwood forests.** Final Report. Arcata, CA: Redwood Sciences Laboratory, Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture.

Menges, K.M. 1994. **Bird abundance in old-growth, thinned and unthinned redwood forest.** Unpublished report. Forestry Department, Humboldt State University. Arcata, CA.

Noss, R.F. 2000. **The redwood forest: history, ecology and conservation of the coast redwoods.** Covelo, CA: Island Press.

O'Hara, K.L.; Dagley, C.M. 2006. **A literature review to examine the potential of silviculture to enhance the formation of old forest characteristics in coast redwood stands.** Poster In: Standiford, R.B.; Giusti, G.A.; Valachovic, Y.; Zielinski, W.J., technical coordinators. Proceedings of the Redwood Region Forest Science Symposium; 2004 March 15-17; Rohnert Park, CA Gen. Tech. Rep. PSW-GTR-194. Albany, CA: Pacific Southwest Research Station. USDA Forest Service; [this volume].

Oliver, W.W.; Lindquist, J.L.; Stothmann, R.O. 1994. **Young growth redwood stands respond well to various thinning intensities.** Western Journal of Applied Forestry 9: 106–112.

Sawyer, J.O.; Sillett, S.C.; Libby, W.J.; Dawson, T.E.; Popenoe, J.H.; Largent, D.L.; Van Pelt, R.V.; Veirs, S.D., Jr.; Noss, R.F.; Thornburgh, D.A.; Del Tredici, P. **Redwood trees, communities and ecosystems: a closer look.** In: Noss, R.F., ed. The redwood forest: history, ecology and conservation of the coast redwoods. Washington, DC: Island Press; 81–118.

SHN Consulting Engineers and Geologists, Inc. 2000. **Habitat assessment, Stimpson Lands, Del Norte County, California.** SHN Consulting Engineers and Geologists, Inc. Eureka, CA.

Stillwater Sciences. 2002. **Mill Creek property interim management recommendations.** Prepared by Stillwater Sciences, Arcata, California for Save-the-Redwoods League, San Francisco and California Coastal Conservancy, Oakland, California.

Stimson Lumber Company. 1998. **DRAFT Multi-species habitat conservation plan for timberlands managed by Stimson Lumber Company, Del Norte County, California.** Prepared by Beak Consultants, Inc., Sacramento, California.

Thornburgh, D.A.; Noss, R.F.; Angelides, D.P.; Olson, C.M.; Euphrat, F.; Welsh, H.H., Jr. 2000. **Managing redwoods.** In: Noss, R.F., ed. **The redwood forest: history, ecology and conservation of the coast redwoods.** Washington, DC: Island Press; 229–262.

Veirs, S.D., Jr. 1986. **Redwood second growth forest stand rehabilitation study, Redwood National Park: Evaluation of 1978–1979 thinning experiments.** Unpublished technical report. Redwood National Park. Arcata, CA.

Veirs, S.D., Jr.; Lennox, W.S. 1982. **Rehabilitation and long-term park management of cutover redwood forest; problems of natural succession.** In: Coats, R.N., ed. Watershed Rehabilitation in Redwood National Park and Other Pacific Coastal Areas. Proceedings of the symposium held August 24–28, 1981. Center for Natural Resources Studies of the John Muir Institute, National Park Service; 50–55

Willett, T.R. 2001. **Spiders and other arthropods as indicators of old-growth versus logged redwood stands.** Restoration Ecology 9: 410–420.

Wilson, J.S.; Oliver, C.D. 2000. **Stability and density management in Douglas-fir plantations.** Canadian Journal of Forest Research 30: 910–920.

Precommercial Stocking Control of Coast Redwood at Caspar Creek, Jackson Demonstration State Forest[1,2]

James Lindquist[3]

Abstract

Regeneration of coast redwood by stump sprouting often results in a stand condition that is very dense. A precommercial thinning of redwood sprouts allows managers to select trees and spacing that can best utilize the productivity of the site. The study of five thinning treatments with an unthinned control was initiated on a 19-year old third growth stand. Treatments were 100, 150, 200, 250, and 300 trees per acre plus an unthinned control. All 18 plots were measured in 1981 immediately after precommercial thinning, again in 1986 (5-year growth), and again in 1998 (12-year growth). Results indicated that despite the range of thinning, the 38-year-old stand showed no statistical differences in volume growth or yield between the thinning treatments. Heavily thinned treatments concentrated more growth on fewer trees to match the stand volume growth in the lightly thinned treatments. A trend appears to be developing that indicates a drop in stand productivity for the heaviest thinning and the control. More time is needed to determine if the trend will continue.

Key words: precommercial, redwood, stocking, thin

Introduction

This report documents the growth response of 18 precommercially thinned and control plots in the coast redwood *(Sequoia sempervirens)* forest type on the Jackson Demonstration State Forest (JDSF). These plots are located in a unit that had been clearcut as part of the 1961 Caspar Creek Cutting Trials (CCCT). This study originated in 1981 as a cooperative effort by the California Department of Forestry and Fire Protection (CDF) and the USDA Forest Service - Redwood Sciences Lab. CDF has since continued the study with a 1986 measurement and report that detailed the first five years of growth following the precommercial thinning (Lindquist 1988a). This paper is a summary of the update of the study, incorporating the 1998 measurement.

Improving the productive capacity of forest stands is the primary reason why forest managers invest in timber stand improvement work. Site preparation, planting, thinning, and other cultural operations are designed to produce a positive return when considering operational and capital costs. This study is designed to determine the growth response of coast redwoods to a variety of stocking levels following precommercial thinning.

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] The California Department of Forestry and Fire Protection funded this study under contract number 8CA85004.

[3] Research Forester, U.S. Forest Service, Retired. 733 Patrick Court, Arcata, CA 95521, (707) 822-3729.

The optimal stand density for young coast redwoods is currently unknown and a complex question. Thinning too heavily may result in inefficient use of the site's productive capacity. In addition, the sprouting ability of coast redwood can result in excessive stocking and spacing problems that are not common to other conifer species. A single redwood stump may often produce more than 20 stems as the result of sprouting. The intense competition among these sprouts may eventually produce one or two codominant trees over the course of many years. Thinning is designed to accelerate this natural process.

Another characteristic of coast redwood is to produce dense, wide crowns if given abundant space, where a low density stand will expend much of its growth capacity on limbs and development of stems with a large amount of taper. Stem selection early in stand development is an important and reasonable option for the forest manager.

Methods

The study is a complete random block design with three blocks. Each block contains six treatments including an unthinned control plot. Blocks 1 and 2 are located in the unburned portion of the clearcut and block 3 is located in the burned portion. The block design allows for the identification of effects due to treatments while controlling for block effects and providing replication. Eighteen plots were established in the study area as shown in *figure 1*. Each plot is 0.4-acres in size. Trees were tagged and measured in the central 0.2-acre area of each plot, while leaving the trees in the 0.2-acre perimeter area as a treatment buffer. The designated treatment was applied to the entire 0.4-acre plot. In addition, there is an unthinned control in each of the three blocks. Trees were selected to retain an equal number of stems in each quadrant of the central plot and in each of the four buffer areas. Redwood was designated as the highest priority for retention, with Douglas-fir *(Pseudotsuga menziesii)*, retained where suitable redwood were not available.

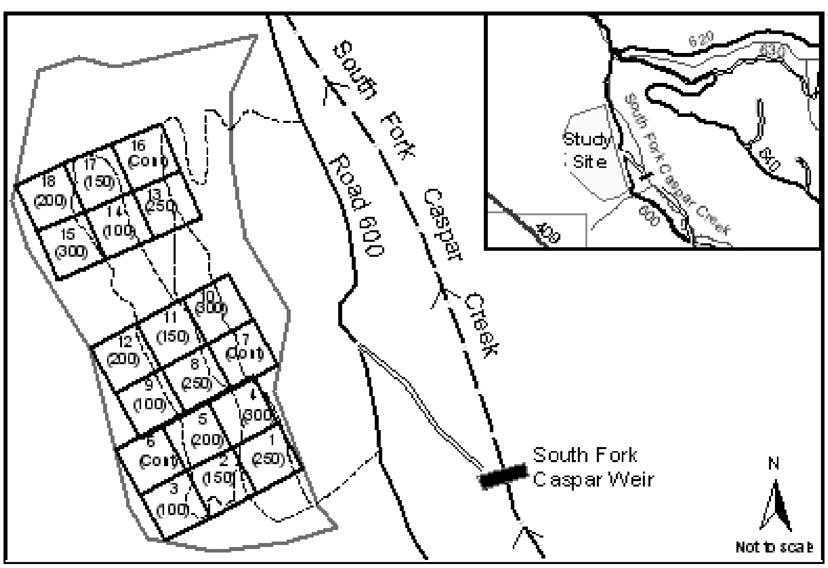


Figure 1—Study design layout for the pre-commercial thinning study.

Figure 2 summarizes the sequence of events in the life of the stand and for the study. Block 3 was burned during site preparation in 1962, which resulted in a dense growth of blue blossom *(Ceonothus thyrsiflorus)* that has since declined in site occupancy. Redwood sprouts, in contrast to natural seedlings, represent 91 percent of all of the redwood regeneration prior to thinning in 1981.

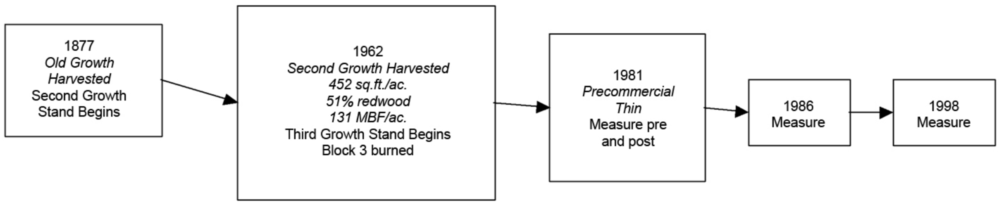


Figure 2—Sequence of events for the stand and the study.

Redwood sprout clumps were thinned to at least a 24-inch stem spacing within a single stump due to the need to distribute the trees as uniformly as possible. Trees as small as 1.5 inches in diameter were selected for retention. The primary objective in thinning redwood clumps was to achieve average spacing targets. Where spacing was adequate, thrifty trees were retained and the remaining stems were thinned from below.

Sample cores for the determination of age and site index (Lindquist and Palley 1961) were taken at each point in time. Local volume equations were developed to account for local conditions and allow for predictions that did not require a total height measurement for each tree. These equations were structured to predict total volume per plot. Individual tree volume was computed for each redwood and Douglas-fir tree with a height measurement, using the Krumland and Wensel (1979) equations for cubic (four inch top) and Scribner board foot (six inch top). A full description is available in Lindquist (2004).

Treatments in this study are the reduction of stand density, as measured by the number of trees per acre. Response to treatments may be exhibited in the average standing volume, periodic growth, mean annual growth as a function of basal area, average diameters, ingrowth, and mortality. Simply identifying differences between treatments is not the only intent of this study. Identification of a suitable stocking level to either maximize yield or accelerate the production of larger trees may be of practical use to forest landowners and managers. The statistical analysis relies upon the use of Analysis of Variance (ANOVA) to determine differences in response as the result of various levels of treatment. Blocks are included as a factor in the ANOVA to account for block effects. Other factors are considered and, where appropriate and not influenced by treatment effects, are accounted for using analysis of covariance (Gomez and Gomez 1984). Specific treatment differences are identified by the Scheffe test of multiple contrasts (SMC). SMC is only appropriate when the ANOVA indicates a significant difference in treatments overall (Neter and others 1985).

Results

Results are presented for diameter, basal area, and cubic and board foot volume response. Mortality was primarily limited to the smallest trees. Ingrowth was still occurring in 1998, particularly for the 10.5-inch dbh threshold. Less Douglas-fir ingrowth occurred as the overstory density increased.

Diameter

Average stand diameter is expressed as quadratic mean diameter (QMD), the diameter of the tree of average basal area. The average stem diameter for trees greater than 1.5 inches immediately after thinning in 1981, ranged from 5.6 inches (control) to 10.7 inches (T150); a range of 5.1 inches *(fig .3).* This exhibited a highly significant difference between treatments, due partially to removal of most of the smallest trees in the heavily thinned plots. This difference remained significant in 1986 and 1998, due to the greater number of slow-growing small stems in the control and higher residual-density treatments. Changes in average stand diameter were also affected by increased mortality of suppressed Douglas-fir. The range of average diameters in 1998 increased to 9.3 inches, partially due to the rapid radial growth in the T100 and T150 treatments.

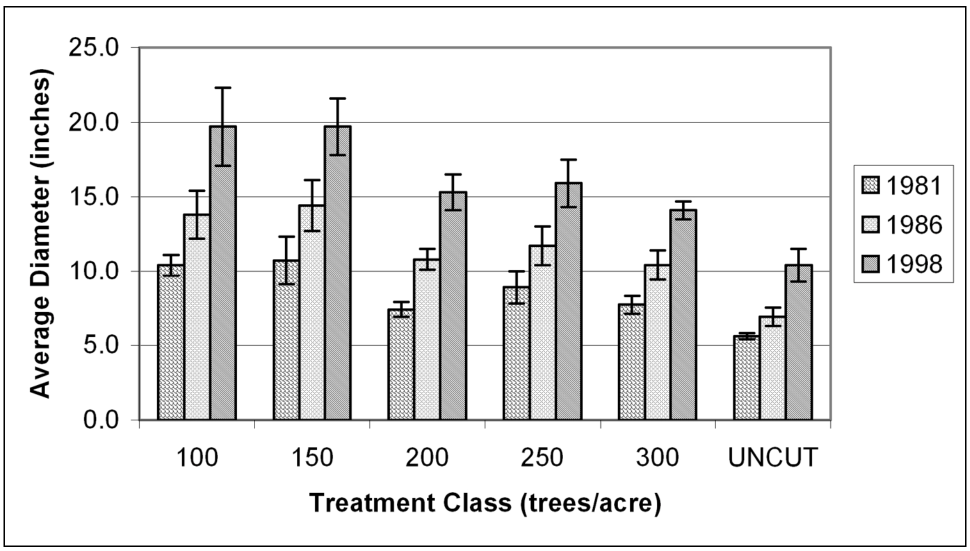


Figure 3—Average stand diameter for trees greater than 1.5 inch dbh. Error bars are one standard deviation.

Basal Area

Stand basal area is a useful variable to consider in the evaluation of growth because it is highly correlated to stand volume. In 1981, immediately after thinning, the differences between treatments were highly significant for stands greater than 1.5 inches. The basal area of trees greater than 10.5 inches *(fig. 4)* was not significantly different between treatments in 1981, in spite of the fact that thinning changed the number of stems in each treatment. These values ranged from a low of 16.9 to 65.3 square feet per acre, with an average of 44.2 square feet per acre. The 1986 inventory also showed no significant differences between treatments despite the fact that the average basal area in thinned stands increased by a factor of 4.3. By 1986, an average of 67.8 trees per acre passed the 10.5-inch threshold. Average stand basal area in 1998 again showed no significant difference between treatments.

Despite the highly significant differences in the number of trees retained in the stands after treatment, the basal area was not significantly different after 17 years. The trees have adjusted to the space provided by thinning. However, there is an apparent response trend developing when both block and pre-treatment large tree inventory are considered *(fig. 5)*. *Table 1* shows that pre-harvest basal area in trees greater than 10.5 inches is an important covariate. Pre-harvest basal area is correlated to 1998 basal area (0.66), cubic volume (0.66), and board foot volume (0.78).

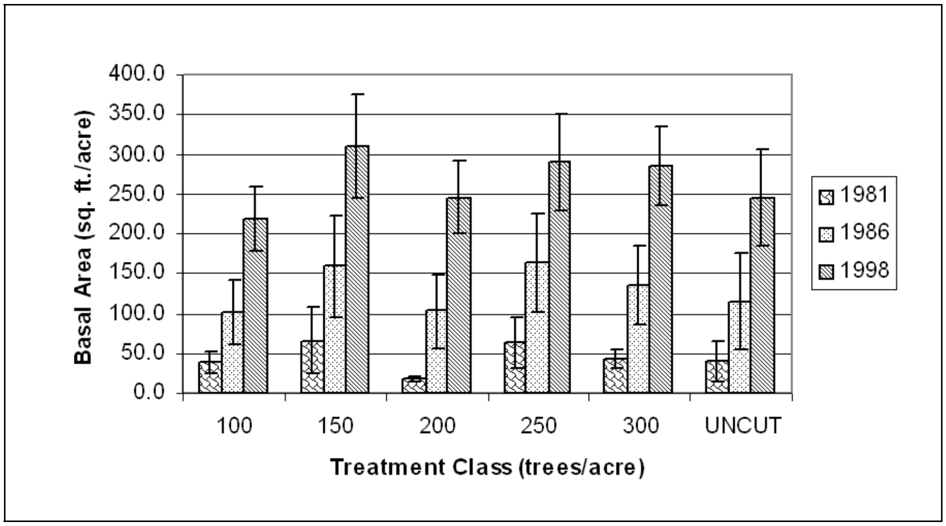


Figure 4—Average basal area for trees greater than 10.5″ dbh. Error bars are one standard deviation.

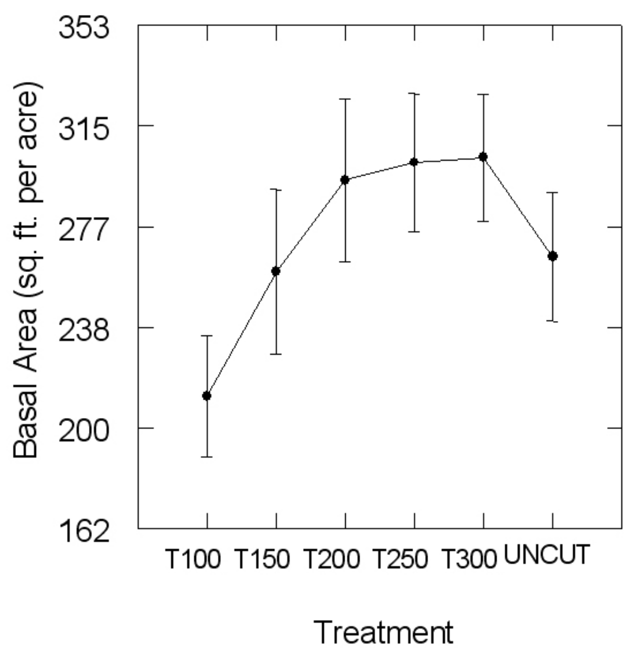


Figure 5—Basal area treatment means for trees greater than 10.5 inches for 1998. ANOVA includes block factor and pre-harvest basal area (>10.5″) as concomitant variable.

Table 1—*Analysis of variance for basal area (ft^2 per acre, trees >10.5 inches) inventory in 1998.*

N: 18 Multiple R: 0.860

Source	*Sum-of-Squares*	*d*	*Mean-Square*	*F-ratio*	*P*
Block	2068.578	1	2068.578	1.221	0.295
Treatment	18915.927	5	3783.185	2.233	0.131
Pre-BA10	18871.893	1	18871.893	11.141	0.008
Error	16939.214	10	1693.921		

The most recent periodic basal area growth in all treatments except T100 was greater than the control, but not statistically significant. Periodic basal area growth for the 17-year period shows no significant differences but all thinned treatments have higher basal area growth than the uncut plots.

Cubic Foot Volume

Cubic volume was computed for all trees greater than 1.5 inches at breast height. Except for the unthinned plots, there were very few stems smaller than 4.5 inches diameter. No ingrowth was observed in terms of cubic volume (trees greater than 1.5 inches). Average cubic-foot inventory was not statistically different between treatments within the 1981, 1986, or 1998 *(table 2)* inventories. Plot variation within treatments was large (*fig. 6* error bars). *Figure 6* illustrates that a trend is developing, similar to basal area *(fig. 5)*, where although not statistically significant yet, the two least-stocked treatments appear to exhibit lower cubic foot productivity. The cubic volume response to thinning in terms of the five, 12, and 17-year periodic growth rate showed no difference between treatments.

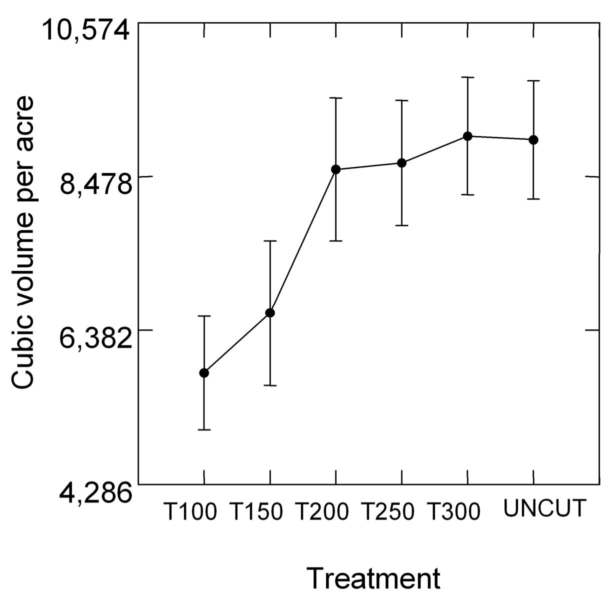


Figure 6—Cubic volume treatment means for 1998. ANOVA includes block factor and pre-harvest basal area (>10.5 inch) as concomitant variable.

Table 2—*Analysis of variance for cubic volume per acre inventory in 1998.*

N: 18 Multiple R: 0.759

Source	*Sum-of-Squares*	*df*	*Mean-Square*	*F-ratio*	*P*
Treatment	2.36960E+07	5	4739194.722	2.676	0.087
Block	274960.137	1	274960.137	0.155	0.702
Pre-BA10	2.01839E+07	1	2.01839E+07	11.398	0.007
Error	1.77090E+07	10	1770896.300		

Board Foot Volume

Board-foot volumes were computed for trees greater than 10.5 inches dbh *(fig. 7)*. Over time, board-foot inventories were affected somewhat by in-growth of smaller trees into the 10.5-inch class. This effect was greatest in the lower density treatments. The board-foot values for each of the three inventory dates indicated no significant differences between treatments *(table 3)*. Adjusting the average treatment means for block and starting inventory effects *(fig. 8)*, the relationship, although not statistically significant, appears to be that T250 has produced the maximum board foot volume after seventeen years.

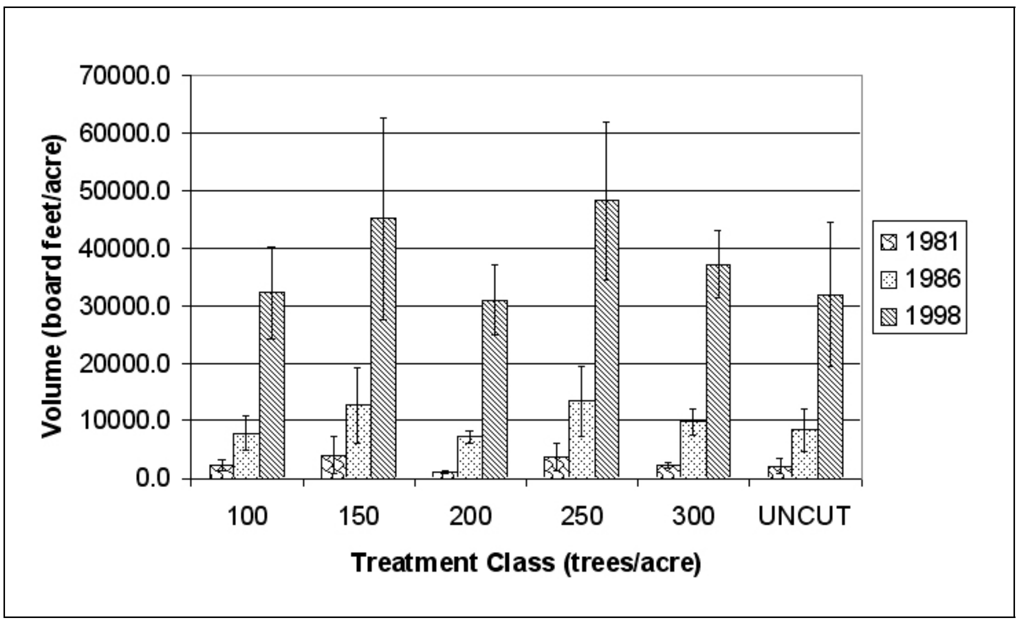


Figure 7—Average board foot per acre volumes (Scribner). Error bars are one standard deviation.

Table 3—*Analysis of variance for board foot volume inventory in 1998.*

N: 18 Multiple R: 0.903

Source	*Sum-of-Squares*	*df*	*Mean-Square*	*F-ratio*	*P*
Treatment	2.70464E+08	5	5.40928E+07	1.087	0.429
Block	1.71297E+08	2	8.56487E+07	1.722	0.233
Pre-BA10	2.94656E+08	1	2.94656E+08	5.923	0.038
Error	4.47759E+08	9	4.97510E+07		

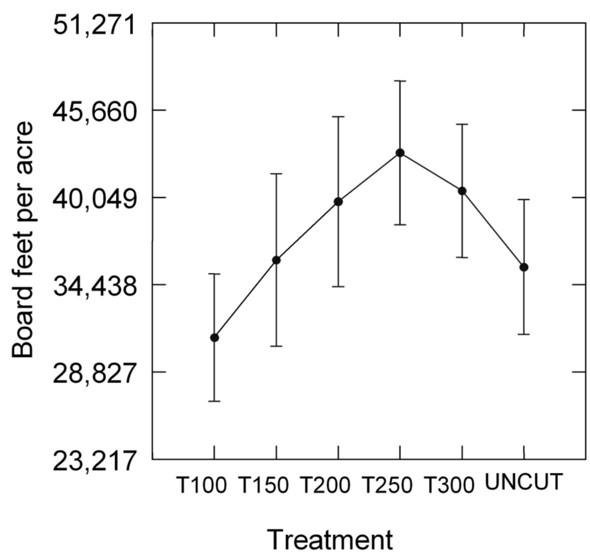


Figure 8—Board foot volume treatment means for 1998. ANOVA includes block factor and pre-harvest basal area (>10.5 inch) as concomitant variable.

The rapid diameter growth increase in the more heavily thinned plots has allowed them to keep pace with total volume growth in the more heavily stocked treatments. The process of thinning has left a stand comprised of vigorous redwood sprouts larger than 10.5 inches dbh. These trees, primarily in the T100 and T150, have responded by growing an average of 7.25 inches between 1981 and 1998. The result was that the thinning treatments to date have not shown statistically significant volume growth differences between treatments. The relative seventeen-year growth trend was the same pattern as that for standing volume shown in *figure 8*, with T250 exhibiting the maximum annual growth.

Discussion

This study provides some valuable insight into the effective use of precommercial thinning in young robust redwood stands. Most of the results related to inventory and growth measures do not exhibit a statistically significant difference between treatments. Variability within treatments, while common in the redwood forest type, results in finding more subtle differences between treatments problematic.

The treatments appear equal due to the ability of redwood to take advantage of space created by thinning. *Figures 6* and *8* demonstrate that volume as a function of treatment resulted in a highly variable result that is not statistically significant at age 38. There are apparent trends in the graphs that suggest a thinning level of 250 or more trees per acre for cubic foot volume and 250 trees per acre for board foot volumes to maximize site utilization.

Maximum diameter growth occurs in the most heavily thinned plots that have a high percentage of redwood. Volume growth in the large diameter trees of the T100 and T150 treatments is keeping these plots growing at a rate that is equal to the more heavily stocked plots. For a short rotation strategy of 60 years, thinning to these levels while retaining mostly redwood sprouts in the stand structure seems to be a reasonable strategy for increasing the size of crop trees. However, *Figures 6* and *8* suggest that as the stand develops there may be a drop in productivity for T100 and T150.

Production of redwood heartwood is a prime economic consideration. Allowing the crop trees to increase in diameter at an optimum rate may be most profitable from the standpoint of volume and lumber grade. The lightly stocked treatments can achieve maximum diameter growth, but may also produce stems with large, wide and spreading branches. In addition, the relationship between branches and heartwood production is unclear (Gartner and others 2002). The plots were thinned at 19 years of age, which may be older than desired. Earlier thinning would result in a stand of small crowns that may take longer to close. Allowing redwood to develop as an open grown stem may result in the production of a highly tapered stem with an over abundance of growth dedicated to branch wood.

In the current third growth stand, there has been an abrupt change in the species composition of the stand. The number of trees in species other than redwood has dropped sharply. The first inventory of the current stand was 19 years after the clearcut and Douglas-fir stems accounted for about 25 percent of the stand. Other conifers are virtually eliminated from the stand composition. This is in contrast to partially logged stands of the Caspar Creek Cutting Trials where about 75 percent of the regeneration was grand fir (Lindquist 1988b). The full sunlight in the clearcut provided the level of light that Douglas-fir requires for growth and regeneration. However, the large number of redwood stump sprouts and heavy brush growth in the burned portion of the study (block 3) resulted in heavy competition for the Douglas-fir and many did not get the height growth necessary to survive in the overstory. However, this is a high Douglas-fir site and Douglas-fir will be an increasing percentage of the stand volume as the stand matures. If rotations are longer than 60 years, Douglas-fir will have adequate time to make an important contribution to the stand yield.

Conclusion

The ANOVA results indicate that despite the range of thinning, the 38-year old stand shows no statistical differences in volume yield or growth between the thinning treatments. However, the decline in volume yield for the unthinned plots seems to indicate the need to reduce the density of regeneration early in the rotation. While this study provides some insight into production at various levels of stocking, the optimum stocking level is still an open question. More time will be required to see if growth rates can be maintained. In time, if smaller codominant and intermediate trees can continue to exhibit good radial growth, the more heavily stocked treatment may yield total greater volume than the lighter stocked treatments.

References

Gartner, B.L.; North, E.M.; Johnson, G.R.; Singleton, R. 2002. **Effects of live crown on vertical patterns of wood density and growth in Douglas-fir.** Canadian Journal of Forest Research 32: 439-447.

Gomez, K.A.; Gomez, A.A. 1984. **Statistical procedures for agricultural research.** 2d ed. New York: John Wiley and Sons; 680 p.

Krumland, B.; Wensel, L. 1979. **Volume and taper relationships for redwood, Douglas-fir, and other conifers in the north coast of California**. Res. Note 9. Berkeley, CA: Coop. Redwood Yield Research Project; 66 p.

Lindquist, J.L. 1988a. **Third growth precommercial thinning at Caspar Creek.** Unpublished report. Calif. Dept. Forestry, Fort Bragg, CA.

Lindquist, J.L. 1988b. **The Caspar Creek cutting trials**. Forestry Note 99. Sacramento, CA: Calif. Dept. of Forestry; 25 p.

Lindquist, J.L. 2004. **Precommercial stocking control of coast redwood: a seventeen-year status report (1981-1998).** California Department of Forestry and Fire Protection. California Forestry Report No. 2. 26p.

Lindquist, J.L.; Palley, M.N. 1961. **Site curves for young-growth coastal redwood.** Calif. Forestry and Forest Products 29. Berkeley, CA: Univ. of Calif.; 4 p.

Lindquist, J.L.; Palley, M.N. 1963. **Empirical yield tables for young-growth redwood.** Calif. Ag. Exp. Sta. Bull. 796; 42 p.

Lindquist, J.L.; Palley, M.N. 1967. **Prediction of stand growth of young redwood.** Calif. Ag. Exp. Sta. Bull. 831; 62 p.

Malain, R.J.; Burns, D.M. 1962. **Young growth redwood and Douglas-fir cutting trials, a preliminary report.** Unpublished report. Calif. Div. Forestry, Fort Bragg, CA.

Neter, J.; Wasserman, W.; Kutner, M. 1985. **Applied linear statistical models.** 2nd ed. Homewood, IL: Irwin; 1127 p.

Smith, D.M. 1962. **The practice of silviculture.** New York: John Wiley and Sons; 578 p.

The Whiskey Springs Redwood Commercial Thinning Study: A 29-Year Status Report (1970 to 1999)[1,2]

James Lindquist[3]

Abstract
The response of well-stocked second-growth coastal redwood stands to three levels of commercial thinning is reported after 29 years of growth. Commercial thinning treatments left 25 percent, 50 percent, 75 percent and 100 percent (uncut control) of the original basal area (400 ft^2 per acre) in a 40-year old stand on the Jackson Demonstration State Forest. Stand values are given for basal area, number of trees, average diameter, and cubic foot volume and board foot volume per acre for the pre-treatment stand and five post-treatment inventory dates from 1970 to 1999.

There were significant differences between the treatments in basal area, average diameter, and volume throughout the course of the study. Analysis of the periodic growth rate revealed strong statistical differences between the treatments in diameter growth but no significant differences in the basal area or volume growth (cubic and board foot). Total board foot volume was not statistically different between thinning treatments due to high variation but there is a trend indicating that the 25 percent retention treatment produced a lower yield than the other two treatments and the uncut control.

Though this was not a regeneration harvest, regeneration was measured for inference to partial harvest management. The response of the understory regeneration was strongly affected by the density of the overstory canopy. A precommercial thinning study of the redwood sprouts showed a response only in the 25 percent overstory retention treatment.

Key words: commercial, precommerical, redwood, regeneration, thin

Introduction
The Whiskey Springs commercial thinning study was established as a cooperative project between the California Department of Forestry and Fire Protection (CDF) and the USDA Forest Service Redwood Sciences Lab (USFS) in 1970 on the Jackson Demonstration State Forest (JDSF) near Fort Bragg, California. The original study design included three sites in the redwood region. Oliver and others (1994) reported the first 15 years of growth (1970 to 1985) for all three sites (Korbel, Crescent City, and Whiskey Springs). This paper covers the growth and yield of the overstory stand in the 12 Whiskey Springs thinning plots for the period 1970 through 1999.

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] This study was funded by the California Department of Forestry and Fire Protection.
[3] Research Forester, U.S. Forest Service, Retired. 733 Patrick Court, Arcata, CA 95521, (707) 822-3729.

The 40-year old well-stocked second growth redwood stand was commercially thinned to four levels of overstory retention to assess stand structure changes for redwood *(Sequoia sempervirens)* over a range of commercial thinning intensities. This growth and yield information will aid in planning silvicultural prescriptions in redwood to achieve volume production and tree size goals.

A second part of this study reports on the establishment and growth of sprout regeneration established as a result of thinning the overstory. The analysis of sprout response provides information for multi-cohort management of redwood stands. A study of new sprout growth after the thinning of these plots described height growth at 15-years of age and developed an equation to predict future growth (Allen and Barrett 1985).

Methods

The stand selected for this study is located approximately nine miles from the Pacific coast, in the drainage of the Little North Fork of Big River near the divide with the Noyo River. There is an east to northeast aspect and an elevation range from 600 to 800 feet. The original old-growth stand was clearcut in the late 1920s and a nearly pure stand of second growth redwood sprouts was established. There is evidence suggesting that the area was planted with redwood seedlings after the initial harvest in the 1920s. There are small intermittent stream courses through plots 1, 2, 3, 4, and 12 that create different moisture regimes and site index than plots located higher on the slopes.

Twelve 0.4-acre plots (132 by 132 feet) were established in the stand. The plots were selected such that intra-plot basal area, site index, and species composition were as uniform as possible. Breast-height age of the redwood ranged from 39 to 41 years in 1970, and the site index ranged from 172 to 212 feet, using 100 years as the base age (Lindquist and Palley 1961). Initial basal area of trees greater than 4.5 inches dbh prior to thinning averaged 401.3 ± 7.6 ft^2 per acre. The species composition of trees by basal area was 91.2 percent redwood, 6.7 percent other conifers, and one percent hardwood. The layout of the plots and assigned treatments are shown in *figure 1*.

Thinning of the plots occurred in 1970 and was based on leaving a residual stand that was 25 percent, 50 percent, or 75 percent (100, 200, 300 ft^2 per acre) of the average basal area of the uncut control plots (400 ft^2 per acre). The interior 0.2-acre section (93.3 by 93.3 ft) of each plot was measured to eliminate edge effect. Leave trees were tagged and evaluated in subsequent inventories. Buffer zones around each central core plot of 0.2 acres were 19.3 ft wide, and were treated in the same way as the tagged portion of the plot. Three plot replicates of the four treatments were assigned in a completely randomized design. Marking of trees was done to favor healthy dominant and codominant redwoods while leaving trees well distributed on the plot. All hardwoods *(Lithocarpus densiflorous)* and any conifers less than 4.5 inches dbh were removed. Over the period 1970 to 1985, five-year measurement intervals were maintained. There have been two measurements since 1985, in 1991 and 1999.

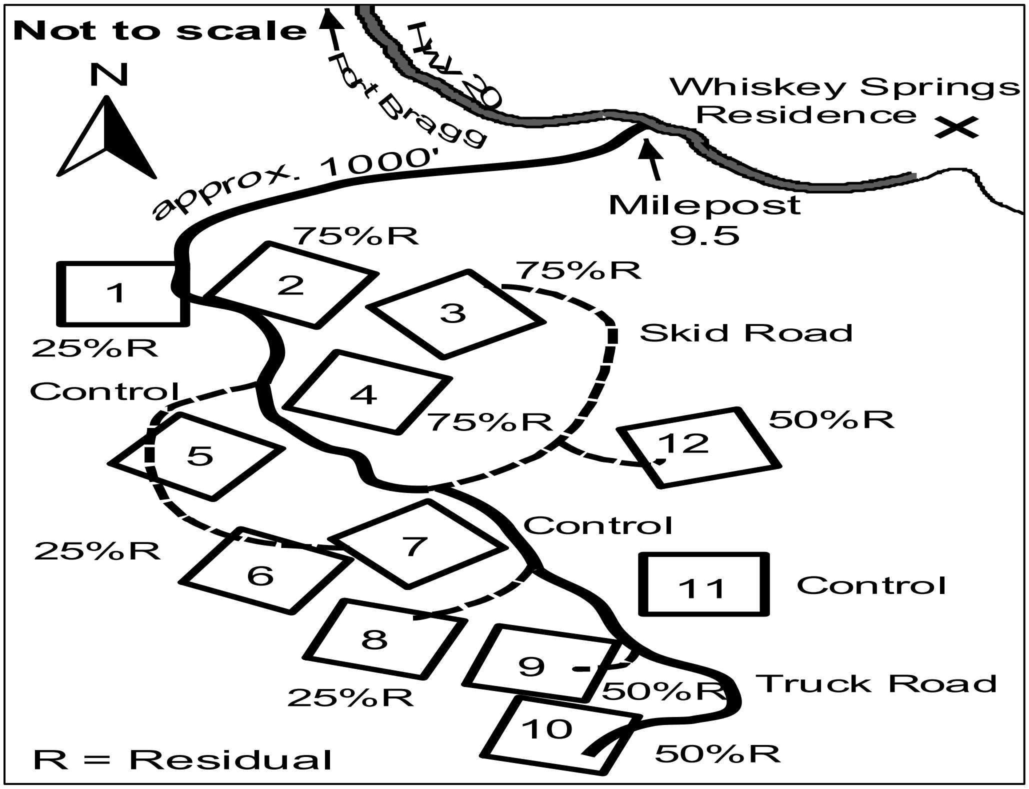


Figure 1—Whiskey Springs location and plot layout.

Evaluation of the stand volumes for all the inventories relied on linear local volume equations for redwood and Douglas-fir *(Pseudotsuga menziesii)*, computed for each plot and inventory year. The small number of Douglas-fir required that all fir be combined. These cubic and board foot tree local volumes were computed from standard equations developed by the Redwood Yield Cooperative (Krumland and Wensel l979). Board foot volumes were computed by the Scribner log rule using a one-foot stump height and a top diameter of 6 inches.

Results and Discussion

Initial Stand Conditions

The average pre-treatment basal area of all plots was 401.3 ± 7.6 ft^2 per acre (range 392 to 410 ft^2 per acre). The number of trees and basal area per acre for trees greater than 4.5 inches dbh include both conifers and hardwoods, but the volumes refer only to conifers. Analysis of variance tests (ANOVA) of the pre-harvest stands indicated that a block design was not required.

Prior to thinning, the thinned plots had similar diameter distributions. Trees less than 10.5 inches dbh are 45 percent to 49 percent of the total numbers of stems. Thinning removed nearly all the stems less than 10.5 inches from the 25 percent and 50 percent retention treatments and only a small percent in the 75 percent retention treatment. The uncut plots averaged about 150 more stems per acre than the treated plots with 72 percent of the trees less than 10.5 inches. The average stand diameter of the control was about 1.5 inches less than the treated plots before thinning, and the board foot volume of the control was about 14,330 less than the average of the pre-treatment volume of the other plots. This may be due to the fact that the three control plots were located higher on the slope and had the lowest average site of any of the treatments.

Within the 25 percent retention treatment, thinning removed 75 percent of the basal area, and 72 percent of the cubic foot volume. The 75 percent retention treatment removed only 20 percent of the cubic volume and 10 percent of the board foot stand.

Repeated measurement of the total heights of dominant redwood in these plots shows that the estimates of site index have remained stable over the 29 years of measurements. Site index estimates of treatments with 25 percent and 75 percent retention are virtually the same in all years. The control had the lowest average site index. An ANOVA of these site estimates showed highly significant differences between treatments, with the uncut plots different than the thinned treatments. This is accounted for in later analyses so that treatment effects may be tested. Testing the site data against the ages of the stands showed no statistical differences.

Basal Area

Basal area growth over a 29-year period for trees greater than 4.5 inches was similar for each treatment *(fig. 2)*. The 25 percent retention treatment added about 150 ft^2 per acre of growth, the 50 percent and 75 percent treatments added an average of 201 ft^2 per acre, and the control increased by 174 ft^2 per acre. Maximum periodic annual increment (PAI) for all harvested stands was in the period of 46 to 51 years of age, which was five to 10 years after cutting the thinning treatment *(fig. 3)*. There was a highly significant difference in basal area growth between treatments in the first growth period of 41 to 46 years ($p = 0.001$). All other growth periods showed no significant differences in periodic basal area growth between the treatments. This was explained by the increased variation within the treatments and the similarity of the PAI between all but the 25 percent treatment. Over the entire 29-year growth period, there was no significant difference in basal area growth between the four treatments. Over a period of 29 years, over 200 suppressed trees have died in the control, which reduced the basal area growth. *Figure 4* shows the relationship of initial basal area and PAI for the 29-year period. This trend agrees with Oliver and others' (1994) findings for the first 15 years of this study.

Diameter

The most obvious effect produced by the thinning was the immediate increase of the quadratic mean diameter, in proportion to the amount of overstory reduction. Average diameters for trees greater than 4.5 inches were not significantly different prior to thinning. In the two heaviest thinnings, the large increase in residual diameter was the result of cutting nearly all of the stems less than 11 inches. After the thinning in 1970 the average stand diameters were statistically different in each treatment ($p = 0.000$).

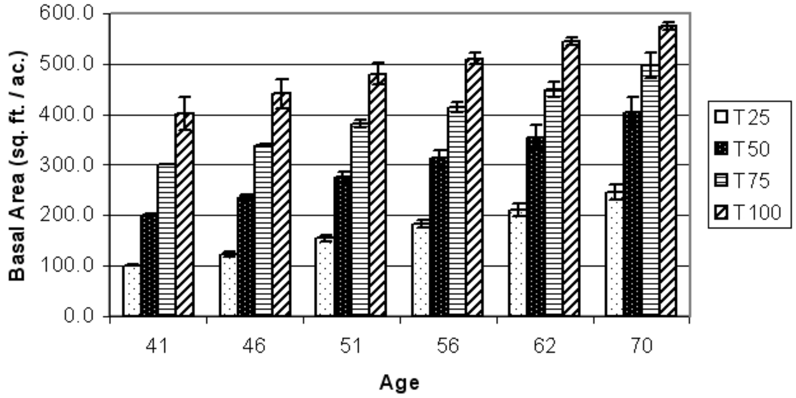


Figure 2—Average stand basal area of trees >4.5" dbh by age and treatment. Error bars are one standard deviation.

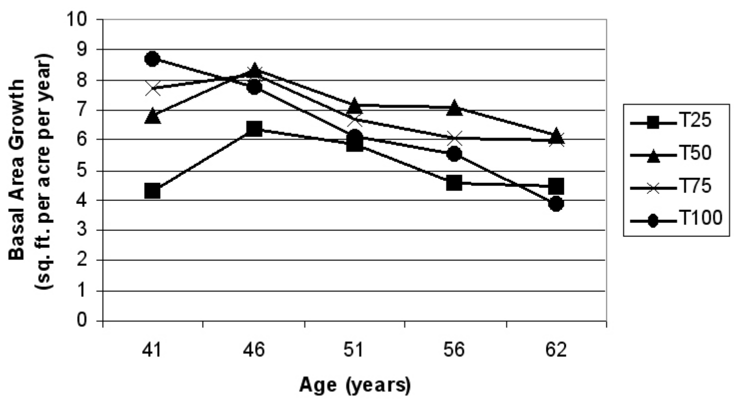


Figure 3—Periodic annual growth of basal area for trees greater than 4.5 inches dbh.

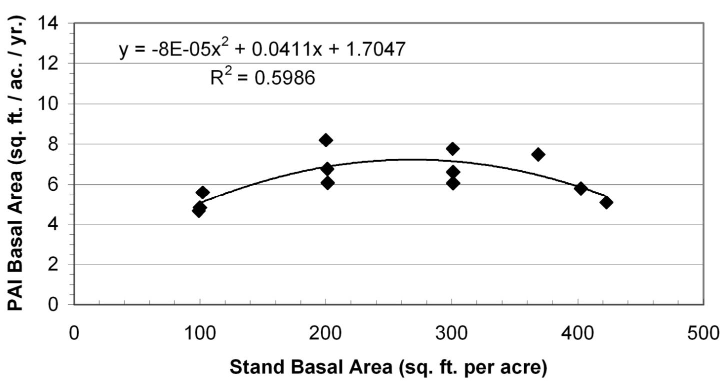


Figure 4—Basal area PAI based on 29 years of post treatment growth, as a function of initial basal area. For trees greater than 4.5 inches dbh.

.

Diameter growth in the residual stand for each treatment has remained consistent over the five growth periods in the 29 years of measurement. The periodic growth percent peaked in the 1975 to 1980 period for the 25 percent and 50 percent retention treatments and then slowly decreased. In the 75 percent treatment plots, the periodic growth dropped consistently. Graphic expression of these growth trends by treatment and age is shown in *figure 5*. Growth of the residual trees was most obvious for the 25 percent retention treatment.

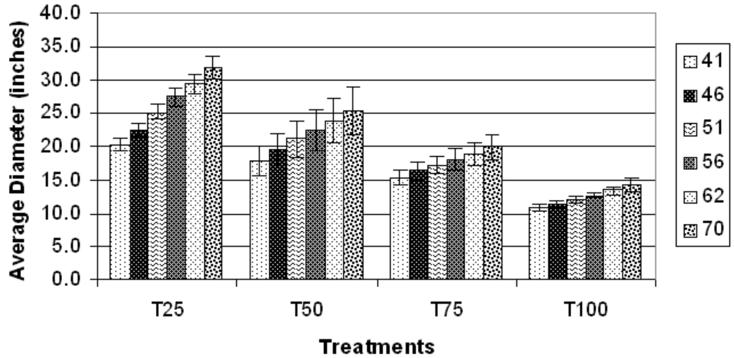


Figure 5—Average diameter of trees greater than 4.5". Error bars are one standard deviation.

Cubic Volume

Cubic volume for trees 4.5 inches diameter and greater over time is shown in *figure 6*. There was a very consistent increase across the five growth periods. Some distortion exists in the graph since the periods between 1970 and 1985 were five-year increments and last two periods are six and eight years long. Volume for the 75 percent basal area retention lags behind the control until 1991 (age 62). The thinning created cubic volumes that were significantly different across the six inventory dates ($p = 0.002$). The Scheffe multiple comparison tests for 1999 showed that the 25 percent retention treatment was statistically different than the lighter cut 75 percent retention treatment and the control.

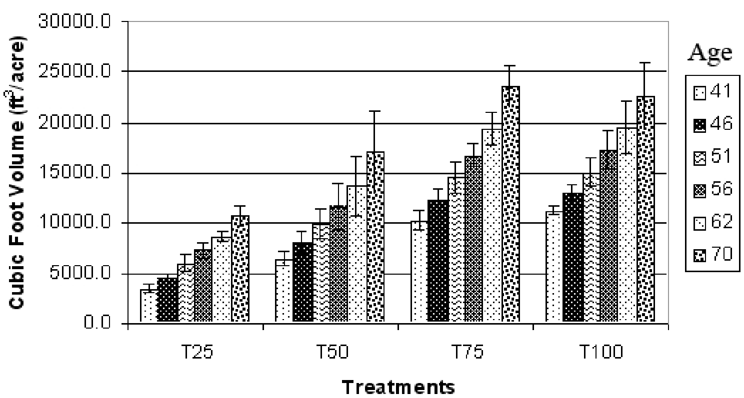


Figure 6—Cubic volume per acre over time by treatment. Error bars are one standard deviation.

Although cubic volume was different, the periodic cubic foot growth in each treatment was uniform with a significant difference in only the period 1970 to 1975 when the Scheffe comparison test indicated a significant difference between the 25 percent retention treatment and the control. The trend of cubic foot volume growth over stand age showed a relatively flat linear trend with an initial increase after thinning and periodic variation *(fig. 7)*.

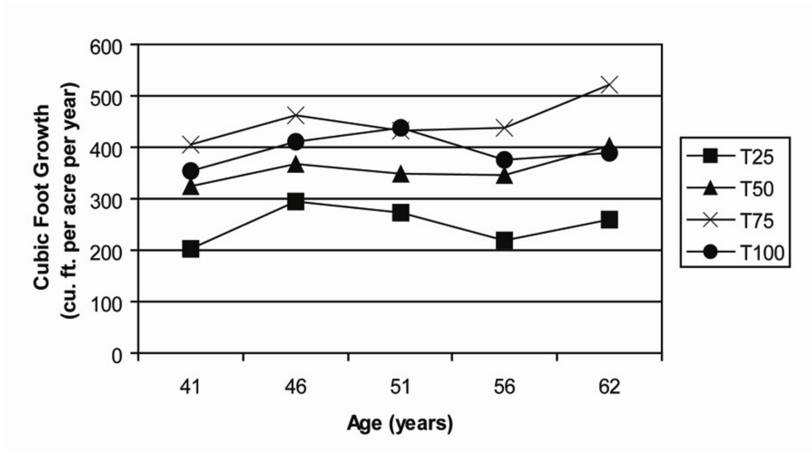


Figure 7—Cubic foot periodic volume growth.

A plot *(fig. 8)* of stand density index (SDI; Reineke 1933) over time indicates a linear increase. The maximum biomass carrying capacity, which is estimated to be 1,000 SDI for redwood (Reineke 1933), has not been reached even by the control. This explains why the cubic stand volume, which is highly correlated with stand biomass, is still increasing.

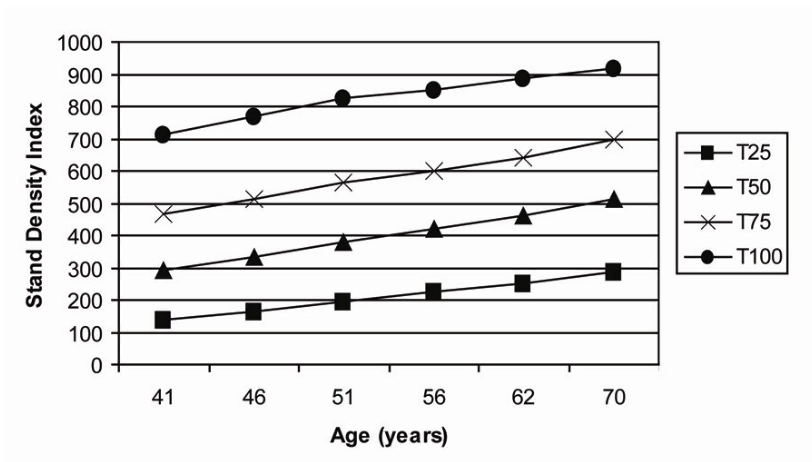


Figure 8—Cubic foot periodic volume growth.

When periodic cubic foot increment was plotted *(fig. 9)* over the initial stand basal area, there was a quadratic form with the maximum at 300 square feet per acre. Cubic-foot increment indicated that for each thinning level, the growth generally peaked five to 10 years after thinning, and then dropped slightly over the remaining periods. The 75 percent retention treatment consistently had the highest growth (458 ft^3/acre/year) over the 29 years, a result of the good site and the presence of large diameter trees. The poorest rates were in the 25 percent retention treatment (250 ft^3/acre/year) where there were only 45 trees per acre. The relatively low cubic volume growth per unit of basal area in the control *(fig. 9)* was partially the result of the loss of many small trees that were part of the cubic foot inventory.

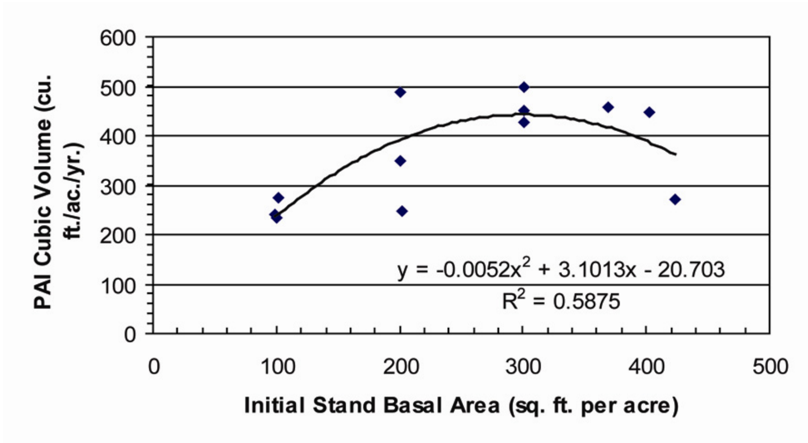


Figure 9—Cubic volume 29-year growth after commercial thinning as a function of initial basal area.

Board Foot Growth and Yield

Many aspects of the results observed for cubic volume were similar for board foot volume. Testing the board-foot volume by ANOVA for each inventory year shows that after thinning, there were significant differences at the P = .05 level. The Scheffe multiple comparisons tests showed that in all five tests the difference was between the 25 percent and 75 percent treatments (p = 0.044 in 1999).

Site index is a significant covariate (P = .039) *(fig. 10)*. Random assignment of treatments to plots resulted in the control plots averaging about 71 percent (14,330 board feet per acre less) of the pre-treatment volume in the treated areas. The control plots are located higher on the slope, which is reflected in their lower site values. Despite these differences in the initial volumes there were no statistically significant differences in the cubic and board foot volumes of the treatment sites. Site index is correlated to the pre-treatment board foot volume ($r^2 = 0.772$).

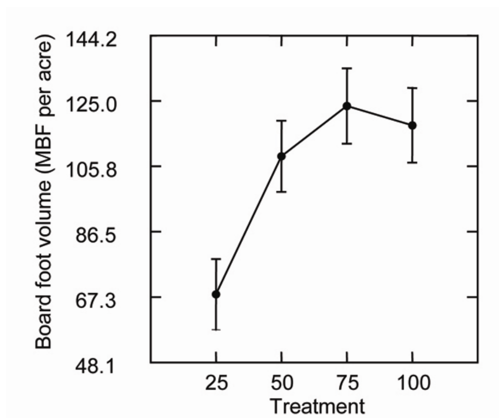


Figure 10—Board foot volume treatment means for 1999.

Growth (board feet) was not significantly different between treatments for any of the periods. The greatest board foot growth in the 29-year period was in the 75 percent retention treatment where the stand volume increased 200 percent, an annual increase of 3,002 board feet per acre per year. The control increased 207 percent or 2,525 board feet per acre per year over the 29 years. Despite the reduction of the stand by 50 percent of the original basal area and leaving only 118 trees per acre, the

50 percent retention treatment grew by 238 percent or 2,544 board feet per acre per year. The 25 percent retention treatment grew 276 percent or 1,803 board feet per acre per year.

The plot of board foot increment *(fig. 11)* as a function of initial post-harvest volume, reveals a linear trend. There is no apparent flattening of the board foot PAI similar to that found in the cubic volume stand *(fig. 9)*. Culmination of the mean annual increment (CMAI) is the age when average volume growth peaks. Analysis to determine if the CMAI had been reached was conducted by adding the standing inventory to the volume removed by thinning, and then dividing by stand age. The values were then averaged for each treatment. The age range covered was 41 to 70 years. There is no evidence of culmination in any of the treatments. The four treatments showed a gradual increase across the 29 years.

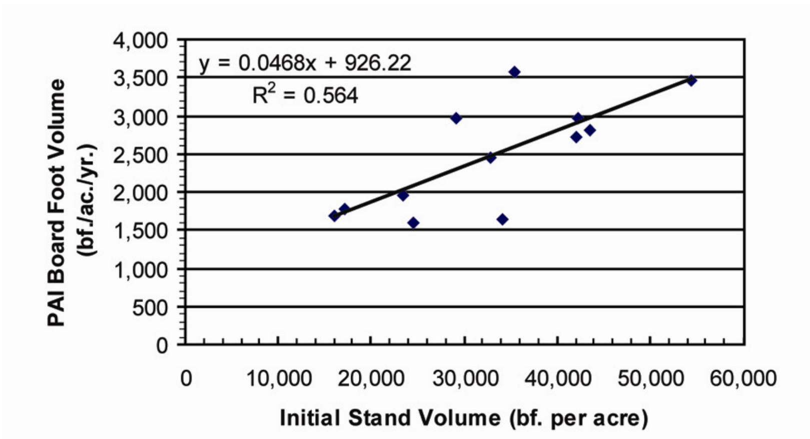


Figure 11—Mean annual increment of board foot volume including commercial harvest yield.

Yield is the total production of a stand and in this case is the sum of the 1999 stand volume and what was harvested in 1970. An ANOVA of yield, including site as a significant covariate (p = 0.024), did not show the treatments to be significantly different (p = 0.283). *Figure 12* shows the adjusted treatment means for yield.

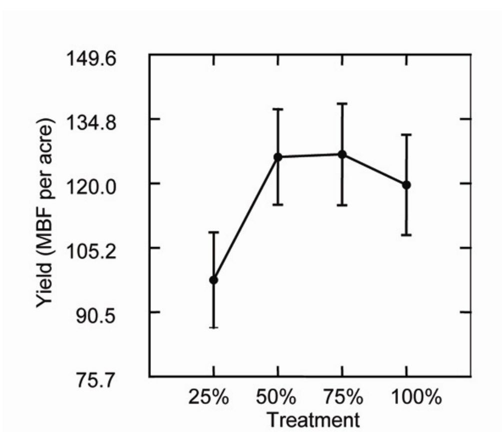


Figure 12—Yield treatment means adjusted by site as a covariate.

Understory Regeneration

The commercial thinning harvest in 1970 created an opportunity to study the effects of the varying amounts of overstory on redwood sprout development in the understory over 29 years of growth. Although the commercial thinning was not designed to promote regeneration, the removal of the trees at various intensities created a stand situation similar to other partial harvest methods such as selection, transition or shelterwood.

The regeneration study was initiated in 1972, two years after the commercial thinning. Six of the nine thinned plots were examined in three overstory levels. The number of sprout clumps, sprouts, and the height of the tallest sprout were measured. These data were used initially to describe and test a method of sampling for redwood sprouts (Lindquist 1979).

In 1986 an understory redwood sprout inventory was made of all nine commercially thinned plots and a precommercial thinning study was established in the 25 percent and 50 percent retention treatments. Plots were established and subdivided into four 0.1-acre plots prior to precommercial thinning the 16-year-old understory. Treatments leaving 200, 300, 400 trees per acre and an unthinned quadrat were assigned at random in the six plots. The retained trees were subsequently measured in 1986, 1991 and 1999.

A large number of sprouts had developed in all treatments, by 1972, ranging from a low of 2,052 sprouts per acre (75 percent retention treatment) to 8,123 (25 percent retention treatment). The 1986 inventory showed a reduction to 268 in sprouts per acre (75 percent retention treatment) and 1,719 sprouts per acre (25 percent retention treatment).

In 1986 both average sprout diameter and height were much larger where 25 percent of the stand was retained than where 50 percent was retained (2.1 inches dbh vs. 0.9 inches and 17.9 feet vs. 6.9 feet). The sprouts in the 75 percent retention treatment were so severely impacted that subsequent measurements were not warranted.

The precommercial thinning treatments appear to have had no effect within the 50 percent overstory retention treatment but appear to have had an effect in the 25 percent overstory retention treatment. In 1986, the 25 percent overstory retention treatment showed no significant difference between precommercial thinning treatments ($p = 0.164$) and there was a difference in 1999 ($p = 0.048$). The 300 stems per acre precommercial target have shown the maximum diameter growth.

The total height of all trees measured indicated highly significant differences ($p = 0.000$) between the 25 percent overstory and 50 percent overstory treatments in all three inventory years. At age 29 these heights were 32.6 feet and 15.2 feet respectively.

Conclusions

Periodic response to the commercial thinning of a well-stocked 41-year old second growth redwood stand was determined by implementing six measurements over a 29-year period. The basal area was reduced by thinning to retain 25 percent, 50 percent, and 75 percent of the average stocking of the three control plots, which was 401 ft^2 per acre. Tests of the plots prior to thinning showed no significant differences between the plot basal area and a simple random assignment of treatments was adopted. Differences in growth attributed to site quality between treatments were accounted for in the board foot volume analysis.

This study showed that young-growth redwood stands are very productive and respond well to a range of stocking levels. Thinning produced a substantial increase in the average diameter of the residual stands. By 1999, the average diameters ranged from 14.2 inches in the control to 31.8 inches in the 25 percent retention treatment, a difference of 17.6 inches. The highest thinning intensity produced the highest radial growth. Intensive thinning from below is capable of producing larger diameter trees relatively quickly in stands at 40 years-of-age. Despite good diameter growth in the 25percent retention treatment, a 75 percent reduction of basal area did not produce total growth at a rate consistent with the site potential. The 50 percent retention treatment, with twice the initial basal area, had a net board-foot growth that was 140 percent of the 25 percent retention treatment.

The relationship between understory growth and overstory density indicated that growth of redwood regeneration was inversely proportional to overstory canopy. The average height of the dominant redwood understory sprouts was 48 feet at 29 years-of-age. This was nearly 20 feet shorter than the average height of 24-year old dominant redwood in a clear-cut block with full sunlight in Caspar Creek (Lindquist 2004a). The time needed to produce trees of commercial size under stands even as lightly stocked as the 25 percent retention treatment with 45 trees per acre will be much longer than under full sunlight. A sub-study of precommercial thinning treatments of the redwood sprouts showed an effect only in the most heavily thinned overstory condition. The results of this study, including data tables, are discussed in greater detail in Lindquist (2004b).

References

Allen, G.M.; Barrett, M.M. 1985. **A model of third growth coastal redwood sprout establishment and growth under various levels of overstory removal**. Unpubl. report; Humboldt State University, Arcata, CA.; 46 p.

Krumland, B.; Wensel, L. 1979. **Volume and taper relationships for redwood, Douglas-fir and other conifers in the north coast of California.** Res. Note 9. Berkeley, CA. Cooperative Redwood Yield Research Project; 66 p.

Lindquist, J.L. 1979. **Sprout regeneration of young-growth redwood: sampling methods compared.** Res. Note PSW-337. Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 8 p.

Lindquist, J.L. 2004a. **Precommercial stocking control of coast redwood: a seventeen-year status report (1981-1998).** California Department of Forestry and Fire Protection. California Forestry Report No. 2; 26 p.

Lindquist, J.L. 2004b. **Growth and yield report for the Whiskey Springs redwood commercial thinning study: a twenty-nine year status report.** California Department of Forestry and Fire Protection. California Forestry Report No. 3; 26 p.

Lindquist, J.L.; Palley, M.N. 1961. **Site curves for young-growth coastal redwood.** California Forestry and Forest Products 26: 1-4.

Oliver, William W.; Lindquist, J.L.; Strothmann, R.O. 1994. **Young-growth redwood stands respond well to various thinning intensities.** Western Journal of Applied Forestry 9(4): 106–112.

Reineke, L.H. 1933. **Perfecting a stand-density index for even-aged forests.** Journal of Agricultural Research 46(7): 627-638.

Silvicultural Challenges for Coast Redwood Management[1]

Kevin L. O'Hara[2]

Key words: leaf area index, multiaged, restoration, Sequoia sempervirens, *silvicultural systems, uneven-aged*

Coast redwood silviculture is probably more varied at present than at any time in the history of redwood forest management. For some forest products companies, management utilizes a series of intensive silvicultural operations where density, composition, competing vegetation, and rotation length are carefully controlled. At the other extreme, some ownerships are beginning to use silvicultural operations to enhance old forest characteristics or favor other non-timber values. Several trends are influencing current silviculture but are also continuing the historical trend of widely divergent approaches to forest management. These include expanded use of tree breeding to improve growth, an increasing recognition of the role of natural disturbances in guiding silvicultural operations, and basing silviculture on a more fundamental understanding of production ecology and stand dynamics.

Current silviculture is presented in the context of using operations to direct stand development on a variety of pathways or trajectories. Silvicultural operations are combined into systems that meet different management goals. These include the role of early density control of a sprouting species with complex spatial patterns of stems and complex early growth patterns. The relationship of leaf area index (LAI)—a surrogate for growing space occupancy—to stand density and growth is used to describe several silviculture systems. Assuming a maximum LAI of 14.0, a system designed to achieve old forest characteristics might use a precommercial thinning treatment to adjust initial spacing and then allow the stand to develop without intervention *(fig. 1)*. A multiaged system might use a 20-year cutting cycle but would attempt to maintain LAI within a range that is less than the maximum. The intensive system uses a 50-year rotation and also maintains LAI at less than 12 over the rotation. Two thinnings are used to adjusting density during each even-aged rotation.

Leaf area index has potential as a unifying variable to link management for objectives associated with stand management with other resource management objectives such as hydrologic response, wildlife habitat, or attainment of prescribed levels of forest cover. For stand management, LAI can be arranged in different structures that have a profound influence on stand productivity. A variety of silvicultural operations can also be described using LAI and LAI represents a viable way of describing the trajectories of stands following treatment.

---

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] 137 Mulford Hall, MC 3114, University of California, Berkeley, CA 94720-3114. email: ohara@nature.berkeley.edu

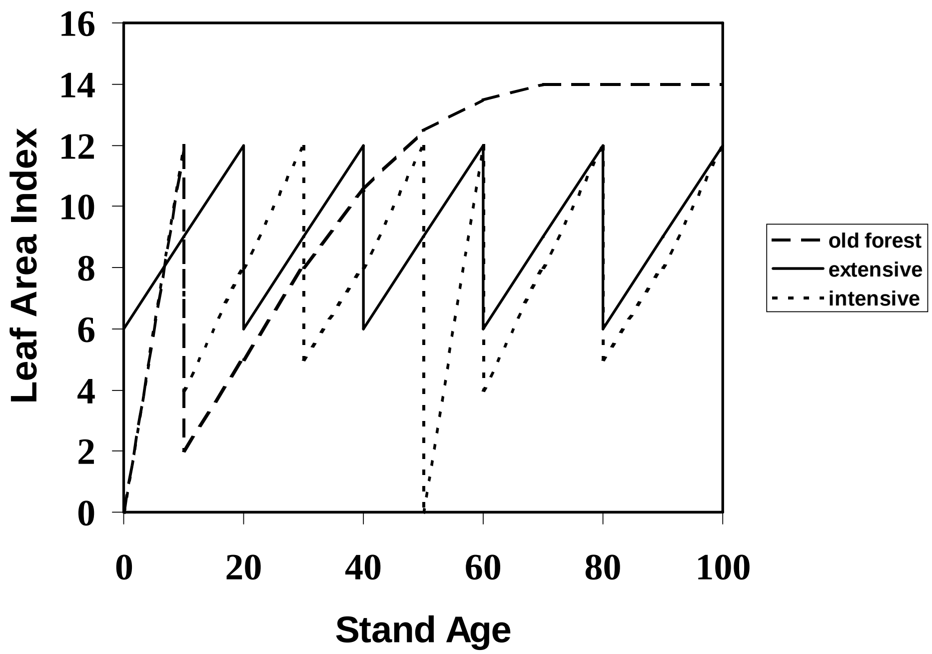


Figure 1—Hypothetical development of LAI for three different management systems. The "old forest" system is designed to restore a cutover site to and old forest. The "extensive" system is a multiaged system with five cutting cycles that fluctuate in LAI from 56 to 12. The "intensive" system is an even-aged regime where LAI develops rapidly after clearcutting, a precommercial thinning treatment at age 10 and a commercial thinning at age 30.

Restoration of Old-Growth Redwood Structural Characteristics With Frequent Variable Silvicultural Entries [1]

Dale A. Thornburgh[2]

Approximately 2,000 acres of second growth redwood forests in the Redwood Region of northwest. California has been divided into ten units. Each unit is entered every 10 years to timber harvest 50 percent of the growth since the last entry. Timber is removed using a variety of silvicultural systems; single tree selection, variable density thinnings and 1/2 acre to 2.5 acre size group cuts. Native conifer seedlings of all species are planted in the openings.

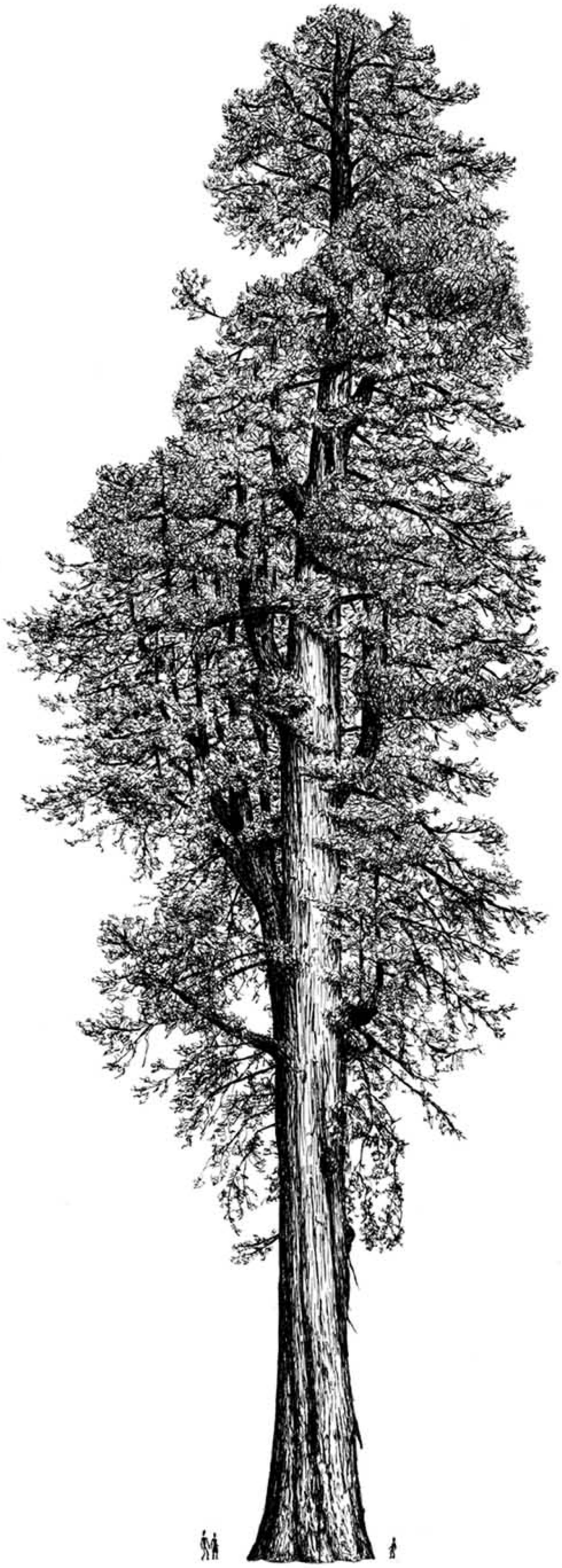
The goal of these cuttings is to develop and maintain a complex old growth redwood forest while obtaining ecomomic returns for the owner. The conifers in these stands will be mainly redwood with a mixture of Douglas fir, Sitka spruce, grand fir and western hemlock. A range of age classes from one year to over 600 years will be achieved over time. These stands will be managed to create and maintain habitat for all species of concern; marbled murrelet, spotted owls, and so forth. To minimize blowdown, redwood sprouts will be selectively thinned out, favoring seedling originated trees to grow into the large 600+ year old growth trees. The frequent entries will provide annual economic return for the owner and maintain habitat for all species

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Forestry Dept., Humboldt State University, Arcata, CA. email: dat1@humboldt.edu

SESSION 8

Erosion and Physical Processes I



Even-Aged Management and Landslide Inventory, Jackson Demonstration State Forest, Mendocino County, California[1]

Julie A. Bawcom[2]

Abstract

Tree removal associated with clearcutting in a coastal redwood forest does not alone initiate numerous shallow landslides that deliver large quantities of sediment to watercourses. This landslide inventory focused on the relationship between vegetation removal in a predominantly second-growth redwood forest and shallow landslides. Deep-seated dormant landslide features were mapped to record if reactivation had occurred within clearcuts. This field-based inventory included mapping all fifty clearcut units and characterizing landslides by type, cause, age, sediment delivery and stream classification. Thirty-two active landslides were found; and all except two were associated with older roads. The percent of the total landslide volume to the volume of landslide sediment reaching a watercourse for each watershed varied from 34 to 75 percent delivery. Sediment delivery depended on the proximity of the road to higher order watercourses. Many deep-seated dormant rockslides were mapped within clearcut units with no reactivation except within old road fills. Mapping nearby uncut or partially cut control units with similar slope characteristics as the clearcuts have yielded similar road failures.[3] Results of this study support the current focus on road rehabilitation and decommissioning for watershed restoration.

Key words: forestry, Jackson State Forest, landslides, redwoods

Introduction

Clearcutting in North Coast redwood forests is suggested as being a significant factor contributing to an increase in landsliding. Available scientific data are not consistent in documenting an association between the two. To better understand the relationship between vegetation removal and slope stability in a second-growth redwood dominated conifer forest, a landslide inventory was prepared for all units that have been clearcut on Jackson Demonstration State Forest (JDSF) in coastal Mendocino County *(fig. 1)*. JDSF is within the redwood forest region in western Mendocino County between Fort Bragg and Willits, California *(fig. 1)*. It is the largest of eight California Demonstration State Forests, having a total area of 48,652 acres, including over 100 miles of fish bearing streams and over 300 miles of actively maintained forest roads.

The four watersheds where clearcutting occurred are in the west half of JDSF. These are the South Fork Noyo River (17,348 acres), a tributary to the Noyo River;

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Julie A. Bawcom, California Geological Survey, 17501 N. Highway 101, Willits, California 95490, (707) 456-1814. email: julie.bawcom@fire.ca.gov

[3] Unpublished mapping by author.

Berry Gulch (7,993 acres), tributary to the Little North Fork of Big River; Hare Creek (6,179 acres), a small coastal stream, and the North Fork of Caspar Creek (1,168 acres),[4] a portion of a small coastal stream *(fig. 1)*. All four of these watersheds are dominated by second-growth redwood with varying percentages of Douglas-fir and hardwoods, primarily tanoak (Henry 1998).

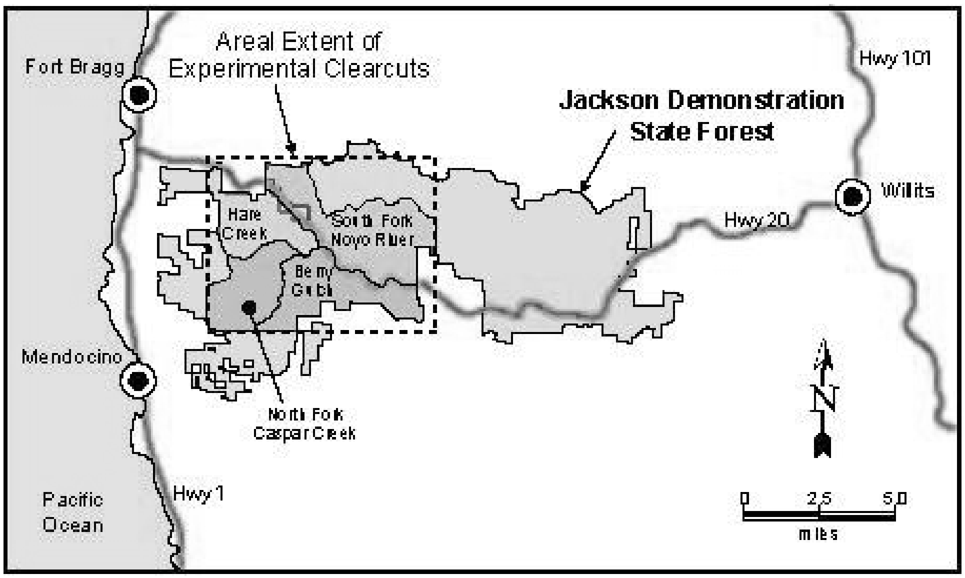


Figure 1—Map of Jackson Demonstration State Forest and the study watersheds. (Unpublished mapping by the author.)

The bedrock geology of the four watersheds is fairly uniform Coastal Belt of the Franciscan Complex (Kilbourne 1982, Kilbourne and Mata-Sol 1983, Manson and others 2001, Short and Spittler 2001, Spittler and McKittrick 1995). Marine sandstones, siltstones, mudstones and conglomerates are dominant, with minor belts of volcanic rocks locally exposed. Weathering and mechanical fracturing is highly variable, and can change dramatically over short distances. The sedimentary rocks are generally massive to poorly bedded sandstone with local outcrops of thin-bedded friable siltstone and shale. A blocky fracture overprints all of the bedrock units. Local areas of deeply weathered and sheared materials with low shear strengths occur in the area studied. The slopes of the clearcut areas are highly variable.

Historic land use pre-dating the establishment of JDSF includes logging of old-growth forest between 1860 and 1947. Splash dams and animal teams were utilized along portions of Big River and Caspar Creek (1860 to 1890). Logging methods changed to steam donkey and railroad in the late 1800s and continued until the beginning of World War II. Many remnant historic logging features (railroad grades, trestles, steam donkey cables and blocks and portions of splash dams) are evident throughout the State Forest. Tractors were first used in the woods prior to World War II, gradually becoming the predominant method of yarding timber by the 1940s. Road construction and use of logging trucks began in the 1930s and replaced the

[4] The North Fork of Caspar Creek watershed is defined as the portion of the basin above the weir constructed in 1962 for the Caspar Creek Watershed Study (Ziemer 1998).

railroad by the mid-1940s. The majority of the main roads on JDSF were constructed between 1950 and 1980.

An association between clearcut harvesting of coniferous forests and landsliding is clearly demonstrated in many parts of the world (Bishop and Stevens 1964, Gray 1970, Montgomery and others 2000, O'Loughlin 1974, Swanston and Swanson 1976). Analyses of this association typically identify the loss of root strength as a controlling mechanism (Abe and Ziemer 1991, Wu and Swanston 1980, Ziemer 1981). Most of these studies involve non-sprouting species, such as pine, varieties of fir, and Douglas fir.

Methods

Between 1980 and 1995, fifty forested blocks were clearcut within nineteen timber sales on JDSF. Approximately 1800 acres were clearcut in four separate watersheds. Most of the clearcut blocks in this study consisted of 80- to 100-year old second-growth stands that naturally regenerated following clearcut logging of the old-growth forest between 1860 and 1947. This majority of second entry clearcuts on JDSF represent a unique data set. Since 1994, JDSF has limited the amount of even-aged management conducted and has transitioned from clearcutting to a system that includes retention of structure trees for habitat purposes.

For this study all of the landslides within modern clearcut units were mapped on the ground using field-mapping methods. Aerial photo interpretation was completed for gathering background data using several sets of photos (1947, 1964, 1981, 1984, 1988, 1996, 2000). Logging history, road construction, date and type of logging method and site preparation were recorded. Within one of the four watersheds, the North Fork of Caspar Creek is part of a paired watershed study of the effects of logging and road building on stream flow, sedimentation, anadromous fish and fish habitat (Lewis and others 2001, Ziemer 1998). Detailed information on the sub-watersheds of the North Fork (14 gauging stations, rain-gauges, subsurface drainage and soil piping sites, and the solar radiometer site) is documented in Ziemer (1998). In addition to geomorphic mapping (Spittler and McKittrick 1995) an inventory of sediment sources, including landslides, are compiled yearly in the North Fork Caspar Creek database (Elizabeth Keppeler, USFS-PSW, personal communication, 2004). Cafferata and Spittler (1998) identified that storm sequences meeting the criteria for triggering landsides have occurred in all phases of the Caspar Creek study, with the greatest number in 1998. Precipitation amounts of at least two inches in one day combined with five inches in three days or eight inches in ten days are thought to trigger landslide events. Three record water years of precipitation occurred with 79.03 inches in 1983, 61.38 inches in 1995, and 80.50 inches in 1998. Because of the close proximity of the other watersheds to Caspar Creek *(fig. 1)*, all of the clearcuts in this study were exposed to rainfall conditions capable of triggering landslides. The landslides mapped for this study were compared to a shallow landslide potential map developed for JDSF by Vestra Inc. in 1997 using the SHALSTAB model. Just over half of the landslide failures occurred within a potential instability rating of moderate or high.

Results

South Fork Noyo River

Seventeen clearcut units totaling 557 acres were mapped for landslides along the South Fork Noyo River *(fig. 2)*. Six of the clearcut units had been broadcast burned after logging. Most of the clearcut units have a Northwest orientation except for three units along the Bear Gulch tributary. Clearcut logging of the original old growth occurred in this basin between 1900 and 1930. Modern clearcutting occurred between 1985 and 1990. Only three westernmost clearcut units had been selectively harvested using tractors in 1968 and 1969. A total of 67 miles of roads are present in the watershed. Slopes are variable, ranging from 10 percent to over 70 percent. About one-half of the units are on dormant landslides (deep-seated rockslides) mapped for this study. None of these deep-seated landslides reactivated after clearcutting. Fifteen landslides (not including the dormant landslides) were mapped in and near the modern clearcut units, with six of these exhibiting evidence of displacement since the most recent logging entry. All of the 15 recently active landslides in clearcut units are shallow debris slides and fill slumps (rotational fill-slides) related to midslope roads constructed in the 1980's. No in-unit landslides occurred in South Fork Noyo clearcut units following the recent harvest. Three slides represent a total of 117 cubic yards of sediment measured in the field delivered into first order watercourses with a delivery ratio for South Fork Noyo River of about 70 percent *(table 1, fig. 2)*.

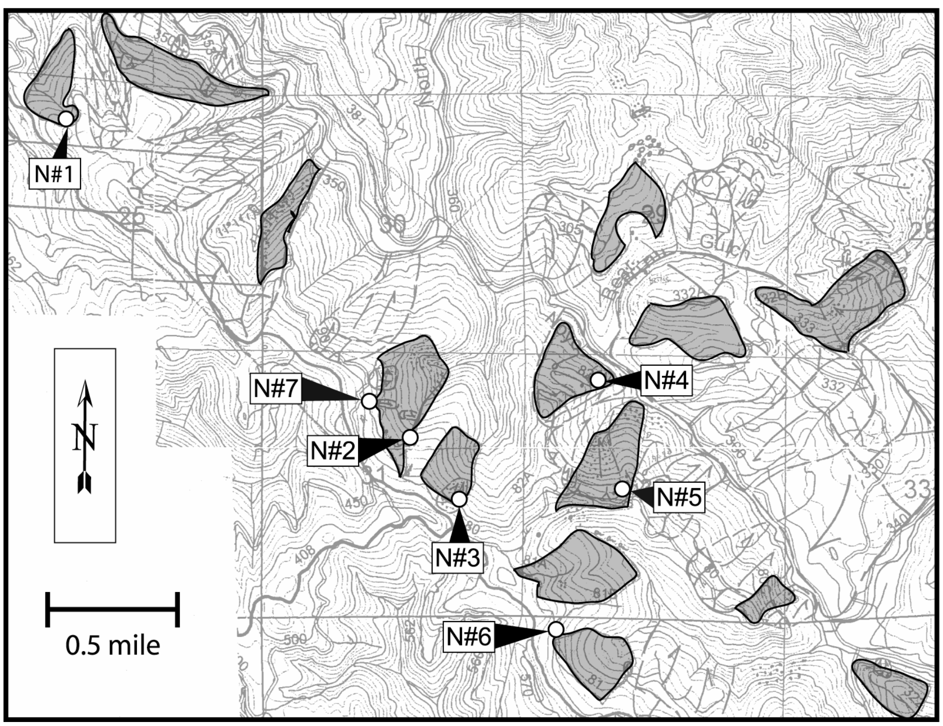


Figure 2—South Fork Noyo River clearcuts and mapped landslides.

Table 1—*South Fork Noyo River landslides summary (refer to map in* figure 2*).*

Slide no.	Volume (yd^3)	Age	Road # age	Delivery (yd^3)	Stream type	Delivery ratio in percent
N1	90	1988	# 259 1979	90	Class III	100
N2		1988	#91 1987	None	None	N.A.
N3	15	1997	#91 1987	15	Class III first order	100
N4	213	1988-1989	#80 1987	None	N.A.	N.A.
N5	415	1997	#80 1987	None	N.A.	N.A.
N6	670	1987	# 81 1987	None	N.A.	N.A.
N7	60	1997	# 91 1987	12	Class III first order	20

Berry Gulch

Nine clearcut units totaling 228 acres in Berry Gulch are located along an unnamed northern tributary of Berry Gulch on south-southwest facing slopes *(fig. 3)*. Most of the units are situated in the upper reaches of the watershed. Five of the nine units were broadcast burned after logging. In addition to the clearcut units, a total of forty miles of roads are present in the Berry Gulch watershed. The old-growth forest was clearcut between 1900 to 1920 using steam donkeys to yard the trees to a railroad spur within the stream channel. The second-growth stands were harvested between 1964-1968 using tractors and 1960s roads constructed along mid and lower slopes. Recent clearcuts occurred between 1987 and 1994 using the 1960s road system. Hill slopes range between 20 and 80 percent. About half of the clearcuts are within dormant deep-seated landslides. A total of eight landslides were mapped (not including the dormant landslides) in the area of the clearcut units occurring between the 1960s logging and the present. Only three of the slides occurred in the 1980s and 1990s, following modern clearcutting. All eight landslides are shallow debris slides and shallow rockslides related to roads or landings. Three landslides delivered sediment to watercourses *(table 2, fig. 3)*. No in-unit landslides occurred in the Berry Creek clearcuts following the recent harvesting. A total of 675 cubic yards of sediment measured in the field was delivered to first order watercourses, with a total delivery ratio for Berry Gulch at about 75 percent.

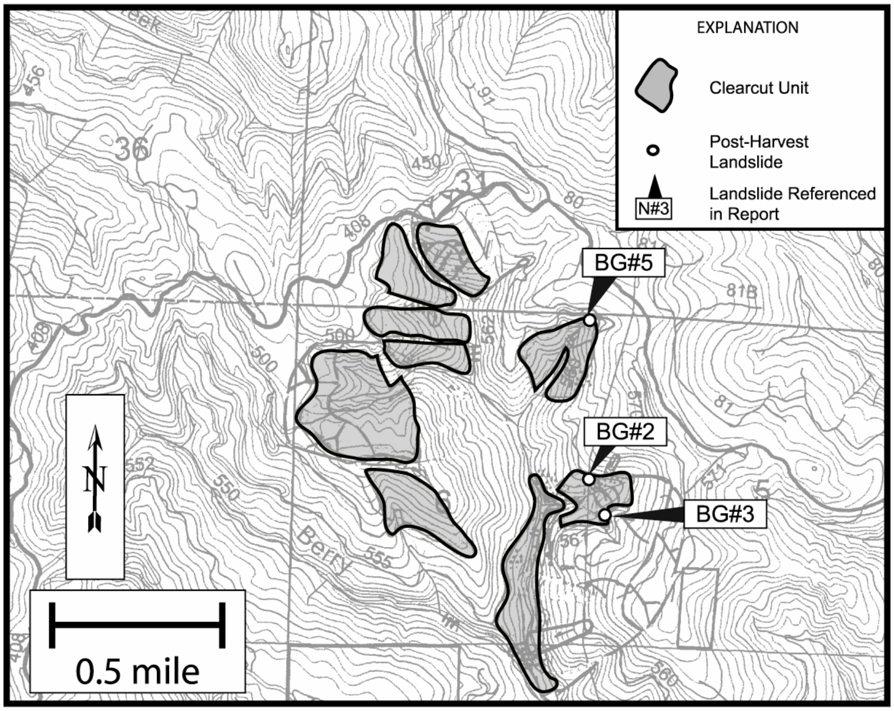


Figure 3—Berry Gulch clearcuts and mapped landslides.

Table 2—*Berry Gulch landslide summary(refer to map,* figure 3).

Slide no.	Volume (yd^3)	Failure age	Road # age	Delivery (yd^3)	Stream type	Delivery ratio in percent
#BG2	129	1975?	# 560 1965	90	Intermit. class III	70
#BG3	100	1995?	# 561 1965	100	Intermit. class III	100
#BG5	670	1980's	# 560 1965	485	class III	72

Hare Creek

Hare Creek is a northwest trending coastal stream with two main tributaries: Bunker Gulch to the north and South Fork Hare Creek to the south. Sixteen clearcut units totaling 353 acres were harvested between 1982 and 1990 *(fig. 4)*. Seven of the sixteen units were broadcast burned after logging. Forty-six miles of roads are located within the watershed built in the 1950s to1980s. The original old growth was clearcut between 1880 and 1900. Topography is highly variable, with slopes under 50 percent along the eastern portion of Hare Creek, and steeper 70 percent slopes along portions of the South Fork and western section of Hare Creek. Only one dormant landslide was mapped in the area of modern clearcutting. A total of twelve landslides occurred, many around the time of road construction before modern clearcutting. All these historically active features are shallow debris slides and fill slumps (rotational fill/slides) related to landings, roads and skid trails constructed in the 1950s to 1980s on lower mid-slopes and along steep stream channel banks. No in-unit landslides occurred in Hare Creek clearcut units following the recent harvesting. Six of the landslides delivered sediment to a watercourse. A total of 4,530 cubic yards of sediment measured in the field was delivered to a first, second or third order watercourse, with a total delivery ratio for Hare Creek of about 34 percent *(table 3, fig. 4)*.

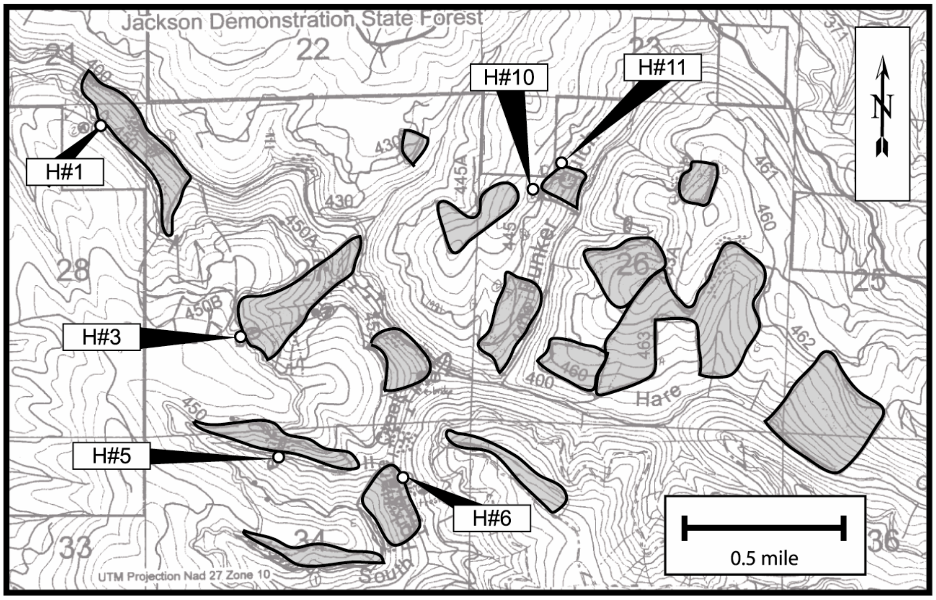


Figure 4—Hare Creek clearcuts and mapped landslides.

Table 3—*Hare Creek landslides summary (refer to map,* figure 4*).*

Slide no.	Volume (yd^3)	Failure age	Road # age	Delivery (yd^3)	Stream type	Delivery ratio in percent
#H1	4800	1995-96	#450 1950's+	300	Hare Creek Class I	6
#H3	930	1997	#450 1950'+	400	Class III	43
#H5	4000	1992-93	#453 1973-74	3500	Class II watercourse	87
#6	3000	1970's	Railroad 1920s	50	South Fork Hare Creek	2
#10	267	1984 +1997	#445 1983-84	200 +50 1997	Bunker Gulch,	94
#11	500	1984-85	#445 1983-84	20-30	Class III	6

North Fork Caspar Creek*:* The North Fork of Caspar Creek is a north-northeast trending tributary of a small coastal stream. Caspar Creek was initially logged between 1860 and 1904 using splash dams to transport logs (Napolitano and others 1989). Ten modern clearcut units totaling 681 acres (about half the watershed) were harvested between 1985 and 1992 in the North Fork of Caspar Creek *(fig.5)*.

These clearcuts are part of a forty-year ongoing study of the watershed effects of harvesting and road building. Four of the ten clearcut-units were broadcast burned for site preparation. The recent logging used 7.1 miles of existing roads and 5.2 miles of new roads located near ridges (Cafferata and Spittler, 1998). The main road providing access to the units (Road 500) was constructed in the 1950s. After the modern clearcut logging in the North Fork Caspar Creek, both the logged and unlogged areas were examined, and a total of six slope failures were found in the clearcut units: four in-unit and two related to roads. Four of the slope failures contributed sediment to a first order watercourse. Two of the largest in-unit delivering landslides (C98, C207) were partly triggered by concentrated road drainage into the clearcut slope. The two slope failures were gully formation of a collapsed soil pipe and within the watercourse and lake protection zone (WLPZ) buffer where channel slumping occurred (C129, C160). As of the spring of 1998, the size and number of landslides in the North Fork Caspar Creek basin were similar in logged and unlogged units and the volume of sediment discharged by landslides from areas cut or uncut was also about the same (Cafferata and Spittler 1998). The two landslides and two slope failures in the North Fork Caspar Fork had a total volume of 9,464 cubic yards delivering about 4000 cubic yards and a total delivery ratio of 42 percent *(tables 4* and *5, fig.6)*.

Table 4—*Caspar Creek landslides summary (refer to map,* figure 5*).*

Slide no.	Volume (yd^3)	Failure age	Road # age	Delivery (yd^3)	Stream type	Delivery ratio in percent
#C98	5574	1995	1985	2200	Class III	39
#C129	110	Jan 1996	N.A.	110	Class III	100
#C160	100	1998	N.A.	50	Class I	50
#C207	3680	Dec 2002	1950	1600	Class III	44

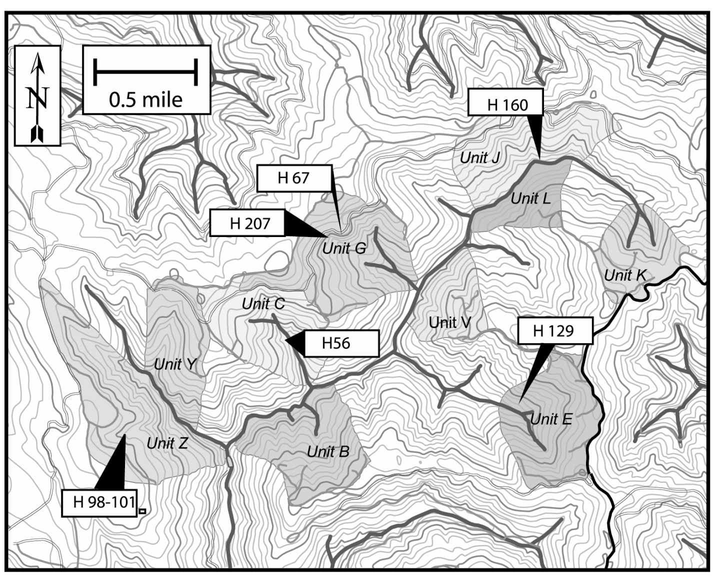


Figure 5—North Fork Caspar Creek clearcuts and mapped landslides.

Discussion and Conclusions

Jackson Demonstration State Forest is unique due to its availability for all types of research in the coastal redwood region, the previously un-entered second growth redwood stands and the uniformity of the underlying geologic bedrock. This research project was able to study the removal of trees in 80- to 100-year old second-growth stands that naturally regenerated following clearcut logging of the old-growth forest between 1860 and 1947. All other studies related to harvesting and slope stability has been completed in areas with predominantly non-sprouting tree species. The inventory on JDSF represents a unique data set.

This study attempts to document the relationship between clearcutting and shallow landslides that deliver sediment to watercourses in a redwood dominated conifer forest. All fifty units, totaling about 1800 acres, within four watersheds were clearcut, and many were broadcast burned for site preparation following logging. All were subjected to storms capable of triggering landslides (Cafferata and Spittler 1998) with thirteen storm sequences above thresholds for initiating landsliding. Following the most recent clearcut logging (1980-1995), a total of thirty-two landslides failed within clearcut units and no deep-seated dormant landslides showed evidence of reactivation except road fill failures.

Of the thirty-two landslide failures, all but two have an association with old roads and landings. Mapping by the author of nearby uncut or partial cut control units revealed similar road related failures. The four in-unit landslides occurred in the Caspar Creek watershed. The first one in the uncut watercourse and lake protection zone (WLPZ), another as gully formation from a collapsed soil pipe and the last two in-unit debris slides associated with poor road drainage. Only the North Fork Caspar Creek watershed with 50 percent of the watershed clearcut as part of a paired watershed study had in-unit landsliding associated with upslope road drainage. The in-unit landsliding may be attributed to the large unit area of vegetation removal along steep slopes and use of other treatment methods such as pre-commercial thinning.

The total amount of sediment delivered from landslides after clearcutting in all four watersheds was about 8,800 cubic yards *(table 5),* with 98 percent of that total associated with old roads and landings. In comparison, one natural debris slide that failed in 1975 and 1998 in another portion of the State Forest (North Fork of the South Fork Noyo River) in uncut fifty-year old second growth delivered about 10,000 cubic yards.

Table 5—*A summary of the number of sediment delivering landslides in each watershed, amount delivered and the delivery ratio (in percent) for each watershed.*

Watershed	Percent acres clearcut	# of slides	Road-related	In-unit slides	# of slides delivering sediment	Delivery (yd^3)	Delivery ratio percent
SF Noyo	4	6	6	0	3	115	70
Berry Gulch	5	8	8	0	3	675	75
Hare Creek	9	12	12	0	6	4530	34
NF Caspar	50	6	4	2	4	3960	42
Total		**32**	**28**	**4**	**16**	**8,740**	

The results of this inventory suggests that vegetation removal associated with clearcutting alone has not been a significant contributor to slope instability or delivery of sediment, and older road, skidtrail and landing fills are the predominant source of shallow landsliding and stream aggradation. Currently there are no other published studies similar to this inventory within coastal redwood forests. The results of this study are supported by other unpublished studies conducted in several nearby watersheds on private industrial timberlands.[5] The results of this inventory also find no increase in the rate of landsliding or initiation of movements of older dormant landslides within clearcuts on JSDF. Results show a clear relationship between roads, particularly older roads as the main source of sediment delivery to streams from shallow landslides and erosion.

Acknowledgments

I would like to thank Marc Jameson, Pete Cafferata, Tom Spittler, Burt Hardin, Mark Smelser, Sebastian Roberts, Elizabeth Keppeler, Dave Longstreth, Brian Bishop, Jim Bawcom and my canine field assistant "Champ."

References

Abe, K.; Ziemer, R.R. 1991. **Effect of tree roots on shallow-seated landslides: technical session on geomorphic hazards in managed forests.** Gen. Tech. Rep. PSW-GTR-130. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 11-20.

Bishop, D.M.; Stevens, M.E. 1964. **Landslides on logged areas in southeast Alaska.** Res. Paper NOR-1. Northern Forest Experiment Station, Forest Service, U.S. Department of Agriculture; 18 p.

Cafferata, P.H.; Spittler, T.E. 1998. **Logging impacts of the 1970's vs. the 1990's in the Caspar Creek watershed.** In: Ziemer, R.R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story; 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 103-115. *Available at* http://www.fs.fed.us/psw/publications/documents/gtr-168/12-cafferata.html

California Department of Forestry and Fire Protection. 2002. **Jackson Demonstration State Forest; State Forest Management Plan**. *Available at* http://www.fire.ca.gov/MiscDocuments/jdsf_mgt_ plan.asp

Gray, D.H. 1970. **Effects of forest clearcutting on the stability of natural slopes**. Bulletin of the Association of Engineering Geologists VII: 45-67.

Henry, N. 1998. **Overview of the Caspar Creek watershed study.** In: Ziemer, R.R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story; 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 1-9. *Available at* http://www.fs.fed.us/psw/publications/documents/gtr-168/01-henry.html

[5] Timothy C. Best, personal communication, 2004.

Kilbourne, R.T. 1982. **Geology and geomorphic features related to landsliding Glenblair NW 7.5' quadrangle, Mendocino County, California.** Sacramento, California: California Department of Conservation, Division of Mines and Geology. OFR 82-25 S.F.; 1:24,000.

Kilbourne, R.T.; Mata-Sol, A.R. 1983. **Geology and geomorphic features related to landsliding Glenblair SW 7.5' quadrangle, Mendocino County, California.** Sacramento, California: California Department of Conservation, Division of Mines and Geology. OFR 83-20 S.F; 1:24,000.

Manson, M.W.; Bawcom, J.A.; Parker, T. 2001. **Noyo River watershed geologic and geomorphic features.** California Geological Survey. Map Set 1- B;1:24,000.

Montgomery, D.R.; Schmidt, K.M.; Greenberg, H.M.; Dietrich, W.E. 2000. **Forest clearing and regional landsliding.** Geology [28(4)]: 311-314.

Napolitano, M.B. 1998. **Persistence of historical logging impacts on channel form in Mainstem North Fork Caspar Creek.** In: Ziemer, R.R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story; 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 97-101. *See* http://www.fs.fed.us/psw/publications/documents/gtr-168/11-napolitano.html

Napolitano, M.; Francis, J.; Caffarata, P. 1989. **A history of logging in the Caspar Creek basin.** Jackson Demonstration State Forest Newsletter, No. 33, 4/1989; 4-7. *See* http://www.rsl.psw.fs.fed.us/projects/water/JDSF89.pdf

O'Loughlin, C.L. 1974. **The effect of timber removal on the stability of forest soils**. Journal of Hydrology (N.Z.) 13: 121-134.

Short, W.R.; Spittler, T.E. 2001. **Preliminary map of geologic and geomorphic features related to landsliding**, Jackson Demonstration State Forest, Mendocino County, California. California Geological Survey: Draft Map Set 2- B, 1:24,000.

Spittler, T.E.; McKittrick, M.A. 1995. **Geologic and geomorphic features related to landsliding, north and south forks of Caspar Creek, Mendocino County, California.** California Division of Mines and Geology, ORF 95-08, 1:12,000. *Available at* http://www.rsl.psw.fs.fed.us/projects/water/Caspar_Geology.pdf

Swanston, D.N.; Swanson, F.J. 1976. **Timber harvesting, mass erosion, and steepland forest geomorphology in the Pacific Northwest**. In: Coates, D.R., ed. Geomorphology and Engineering. Stroudsburg, PA: Dowden, Hutchinson & Ross; 199-221.

Wu, T.H.; Swantston, D.N. 1980. **Risk of landslides in shallow soils and its relation to clearcutting in southeastern Alaska**. Forest Science 26(3): 495-510.

Ziemer, R.R. 1981. **Roots and the stability of forested slopes.** In: Davies, T.R.H.; Pearce, A.J., eds. Erosion and sediment transport in Pacific Rim Steeplands. Christchurch New Zealand: International Association of Hydrological Sciences Pub. No. 132, 343-361. *Available at* http://www.rsl.psw.fs.fed.us/projects/water/ZiemerIAHS.PDF

Ziemer, R.R., technical coordinator. 1998. **Proceedings of the conference on coastal watersheds: the Caspar Creek story;** 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 129-134. *Available at* http://www.fs.fed.us/psw/publications/documents/gtr-168/14-ziemer.html

Overview of the Ground and Its Movement in Part of Northwestern California[1]

Stephen D. Ellen,[2] Juan de la Fuente,[3] James N. Falls,[4] and Robert J. McLaughlin[5]

Abstract

The Eureka area of northwestern California is characterized by a variety of terrain forms that reflect a variety of geologic materials, most of which are components of the highly disrupted and heterogeneous Franciscan Complex. Recent regional geologic mapping by McLaughlin and others (2000) has delineated the distribution of contrasting materials within the principal recognized units of the Franciscan, and so has revealed for the first time the intricate distribution of materials in much of the area.

This geologic mapping was used by a mass-wasting review panel to select areas for detailed geomorphic mapping of landslide processes. Three study areas were chosen to represent the geologic and topographic diversity of timberlands in the broader area. In addition to abundant debris sliding, this mapping revealed that as much as 70 percent or more of this landscape shows evidence of large landslides of diverse scales and styles, ranging from fresh features produced by recent movement to subdued features suggestive of ancient movement followed by long quiescence. Several lines of evidence suggest that large landslide deposits in this area warrant monitoring for movement, as sediment contribution from subtle movement of these large masses in some areas could be comparable to that from debris sliding.

Key words: California, debris slides, geology, landslides, sediment, terrain mapping

Introduction

This paper describes results from two projects that in recent years have shed light on the nature of the ground and its movement in the Eureka area of northwestern California. The first, geologic mapping by McLaughlin and others (2000), describes the geologic materials and their relation to the varied topography of the region. The second, geomorphic mapping conducted for a mass-wasting scientific review panel, describes the kinds and abundance of landslides that have occurred in study areas representative of these materials and topographies. Together the projects clarify relations relevant to timber harvest in the area.

Below we describe these two studies in sequence: first, the geologic mapping of bedrock materials, which permitted the panel to choose geologically representative

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Retired from U.S. Geological Survey, 116 Huckleberry Trail, Redwood City, CA 94062. email: sdellen@comcast.net

[3] USDA Forest Service, Klamath National Forest, 1312 Fairlane Road, Yreka, CA 96097. email: jdelafuente@fs.fed.us

[4] California Geological Survey, 2120 Campton Road, suite D, Eureka, CA 95503. email: james.falls@conservation.ca.gov

[5] U.S. Geological Survey, 345 Middlefield Road MS-973, Menlo Park, CA 94025. email: rjmcl@usgs.gov

study areas; then the geomorphic mapping that revealed landslide processes within the study areas.

Geologic Mapping of Earth Materials

Methods

Geologic mapping by McLaughlin and others (2000) began by delineating the major geologic units at regional scale (1:100,000). Two principal bedrock sequences underlie the area *(fig. 1)*: the generally older (Jurassic to Miocene) assemblage of disrupted and dismembered geologic materials that constitute the Franciscan Complex; and the Wildcat Group of Ogle (1953), which consists of poorly lithified, young (Miocene to Pleistocene) sedimentary rocks. The Franciscan rocks reflect a history of deformation as severe as any in the world. The Wildcat strata, which were deposited atop the late Eocene and older rocks of the Franciscan, in many places have been folded and faulted, but because their deposition post-dates the severe deformation of the underlying Franciscan rocks, they remain largely intact as flat-lying or tilted sequences of layered rock.

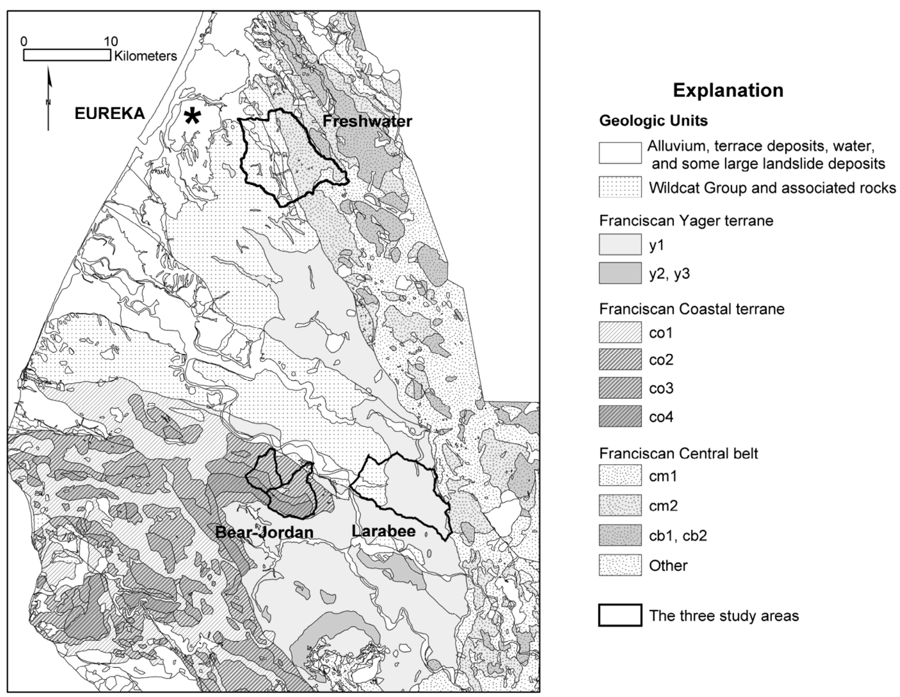


Figure 1—The Eureka area of northwestern California, showing geologic units (modified from McLaughlin and others 2000) and the three areas of detailed study. Shades of gray in the Franciscan distinguish subunits mapped by topographic form; darker tones within each group indicate the "harder" topographies.

The Franciscan Complex in this area includes three major units—the Central belt, Yager terrane, and Coastal terrane. Each of these units consists dominantly of sandstone and clayey rock (mudstone, siltstone, and shale) that have been deformed to varying degrees. Shearing and disruption tend to be concentrated in the clayey rocks. Style and overall degree of deformation differ among these units, but bedrock in each unit ranges from more-or-less intact sandstone-dominated sequences to clayey, intensively sheared shale.

To map the distribution of these disparate materials within the major Franciscan units, McLaughlin and others (2000) used a form of terrain mapping, in which contrasts in topographic form, as viewed in high-altitude aerial photographs, were systematically mapped as a reflection of the mechanical character of the underlying bedrock (see Ellen and Wentworth 1995, McLaughlin and others 2000). The Franciscan displays strong contrasts in topographic form *(fig. 2)*, and the use of these contrasts for distinguishing bedrock materials is widely recognized (for example, Kelsey 1980).

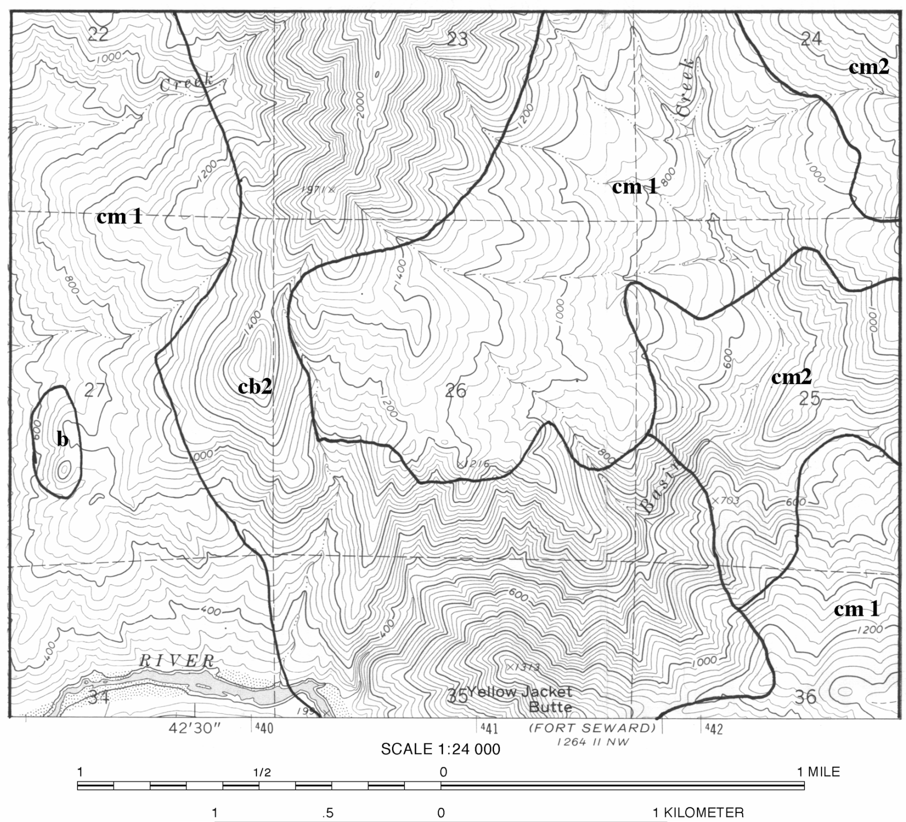


Figure 2—Contrasting topographies in the Franciscan Central belt. Subunits cm1, cm2, cb1 (not shown), and cb2 constitute a spectrum from gently sloping and poorly incised topography (cm1), characterized by a large proportion of weak, highly sheared clayey material, to steep and regularly incised topography (cb2) underlain largely by strong, clay-free materials. Area illustrated is near Fort Seward, California. From McLaughlin and others (2000). Contour interval 12.2 m.

Results

The terrain mapping distinguished 1) steep, angular, and deeply dissected "hard" topography, which tends to develop in rockmasses that lack clayey materials, or in which clayey materials are not mobilized in slope processes, from 2) gently sloping, rounded, shallowly dissected, and more irregular "soft" topography, which tends to develop in clayey sheared materials (Ellen and others 1988). Intermediate topographies, reflecting intermediate conditions, were also distinguished. Subunits are designated by numbers that increase with "hardness" of the topographic form, as shown in the explanation of *figure 1*. Thus, subunits of the Coastal terrane are labeled co1, co2, co3, and co4. In the Yager terrane, which lacks exceedingly "soft" topography, only three subunits (y1, y2, and y3) were distinguished. For the Central belt, the accepted terms "mélange" (for highly sheared rocks) and "broken formation" (for more intact materials) were used in the naming system, so that cm1 and cm2 (for melange), and cb1 and cb2 (for broken formation), form a sequence that corresponds to co1, co2, co3, and co4 of the Coastal terrane (see *fig. 2*).

Discussion

The logic behind the terrain-mapping method is that contrasts in topographic textures evident in small-scale imagery (1:80,000-scale and smaller) arise largely from the different slope processes that characterize clayey and granular materials. Clayey and granular material properties constitute the principal distinctions in engineering behavior and slope stability (Kenney 1984, Mitchell 1976). Granular materials, such as crushed rock or clay-poor fractured rockmasses, are strong and tend to support steep slopes; clayey materials tend to be weaker and can support only gentler slopes (Kirkby 1987). These material properties also result in different landslide behaviors once the ground fails and in different propensity for soil creep, both of which may further affect topographic form. Thus, the systematically mapped spectrum of topographic form *(fig. 2)*, expressed by the numbered subunits, depicts a spectrum of landslide processes and also of engineering materials. The mapping may consequently be useful in reconnaissance planning of conventional engineering activities, such as grading, as well as in considerations of landsliding.

This geologic mapping permitted the mass-wasting review panel to select study areas representative of the geologic and topographic diversity of timberlands in the Eureka area *(fig. 1)*. The watershed of Freshwater Creek, east of Eureka, is underlain principally by the Wildcat Group and by subunits of the Central belt of the Franciscan Complex. The contiguous watersheds of Bear and Jordan Creeks, located well south of Eureka, are underlain dominantly by a spectrum of Franciscan Coastal terrane. The lower part of the Larabee Creek watershed is underlain in part by the most abundant subunit of the Franciscan Yager terrane and in part by the Wildcat Group. The Wildcat Group in Larabee dips sufficiently to provide the possibility of landsliding along weak layers and bedding planes, whereas in Freshwater layering generally is close to horizontal. Thus, these three study areas sample the principal geologic units and configurations of the area (*fig. 1*).

Geomorphic Mapping of Landsliding

Methods

To determine the kinds of landslides and their distributions, the review panel conducted new geomorphic mapping in the Bear-Jordan and Larabee study areas (de la Fuente and Hicks 2001, Ellen 2001) and updated recent mapping in the Freshwater study area (Falls 2001). The mapping was accomplished principally by interpretation of aerial photographs. It made use of multiple sets of photographs in order to gain the fullest possible description, in contrast to previous mapping of the region (for example, Spittler 1982, 1983) which was constrained by a single set of aerial photographs. Field checking of landslides and other geologic conditions was limited to about a week in Bear-Jordan and in Larabee; Freshwater had received periodic field visits over the course of five years.

Results

Kinds of Landsliding

The study areas provided a fresh look at the kinds of landslides and their abundance. Two basic types are recognized. Debris slides are shallow (typically 0.5 to three m thick) landslides that carry vegetation and soils rapidly from steep hillslopes. Debris slides may be called debris avalanches if they are especially fast-moving, and they may become fluid and travel far downslope or down channels as debris flows, mud flows, or debris torrents. Debris slides leave behind barren scars that can be easy to recognize in the field or in aerial photographs until vegetation covers the bare soil or rock, typically after a few years.

The other basic type of landslide that has occurred in the area is generally larger, slower, and deeper (Swanson and Swanston 1977). These are less commonly recognized because their movement typically is more gradual, they largely retain forest cover or other vegetation, and they can remain dormant for many years. These landslides tend to move progressively and intermittently, often in seasonal increments, over periods of years, decades, or longer. They may range in size from a few acres up to the scale of a mountainside (several square kilometers). Movement may be dominantly rotational in profile (called rotational sliding or slumping), in which back-rotation of the sliding ground results in hillside benches or depressions; dominantly translational, where movement parallels the ground surface; or more fluid in character, where moving materials form elongate tongues called “earth flows” that tend to move slowly and persistently. In this paper, we call ground that has moved by any of these larger, slower, and deeper processes “large landslide deposits.” These are commonly bordered by landslide scarps, the steep slopes left behind by displaced ground.

Abundance

Debris slides and large landslide features are more abundant in the study areas than shown in earlier published mapping. *Table 1* shows the comparison for the Bear-Jordan study area. The contrast is especially pronounced for large landslide deposits. The new mapping shows large landslide deposits and scarps occupying 18 percent of the ground in Freshwater, 66 percent in Bear-Jordan, and 76 percent in Larabee.

Table 1—*Abundance of mapped debris slides and large landslide deposits in Bear-Jordan* study area.

	Debris slides		Landslide deposits and scarps	
	Count	**Percent of area**	**Count**	**Percent of area**
Previous mapping (Spittler 1982, 1983)	121	4	25	2
New mapping (de la Fuente and Hicks 2001)	776	6	377	66

Landslide styles in the study areas

Landslides in the three study areas show different styles and distributions

Freshwater (fig. 3)—Large landslide deposits here chiefly occur as discrete masses surrounded by unslid ground, especially in the western half of the study area underlain by the Wildcat Group. Here large landslide deposits typically have moved by rotational or translational sliding. In the Franciscan Central belt of the eastern part of the study area, large landslide deposits typically result from earth flows and tend to be larger and more elongate than landslide deposits in the Wildcat Group.

Debris slides and related phenomena occur scattered about the study area. In the Wildcat Group, most originate high in the landscape in low-order drainage basins, and few occur along major streams. In the Franciscan, most occur low in the topography along major streams.

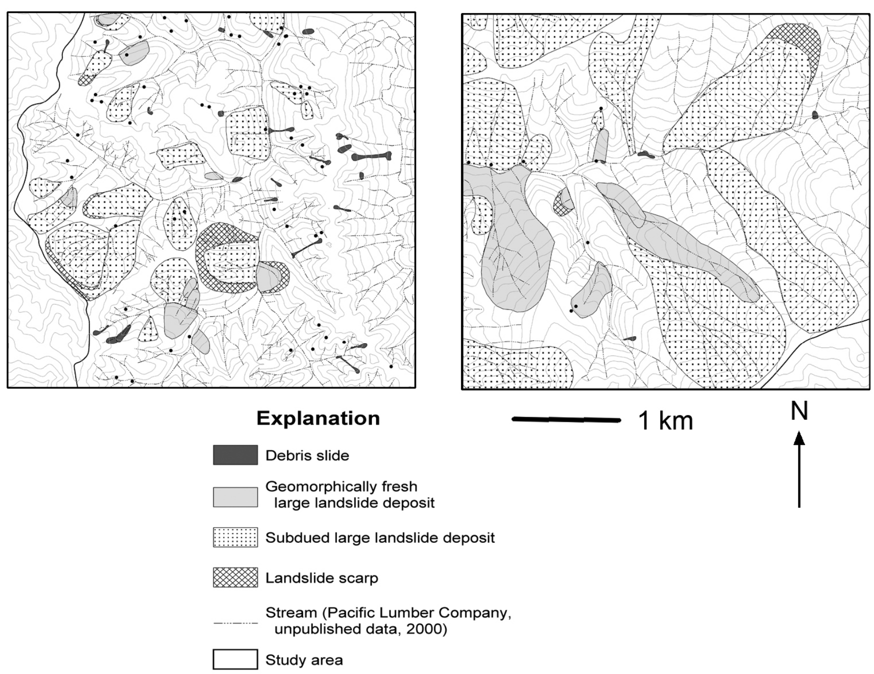


Figure 3—Landslide features in Freshwater study area (Falls 2001). A, Part of western side of study area, underlain by Wildcat Group; B, Part of eastern side of study area, underlain by Franciscan Central belt (mostly unit cm2). Contour interval 25 m.

Bear-Jordan (fig. 4)—Large landslide deposits occupy most of the ground in these watersheds except ridge crests and sidehill ridges. In the steep headwaters area shown in *figure 4,* which constitutes "hard" topography of the Coastal terrane, large landslide deposits occupy the steep hillsides and sidehill draws of this landscape. "Softer" units of the Coastal terrane, which underlie middle and lower parts of these watersheds, include at least one large earth flow in addition to numerous large landslide deposits of less distinctive style, some of which are interrelated as parts of landslide complexes.

Debris slides are abundant and occur dominantly along the main stream channels. Many are large, and some of these large debris slides appear to represent detached pieces of preexisting large landslide deposits.

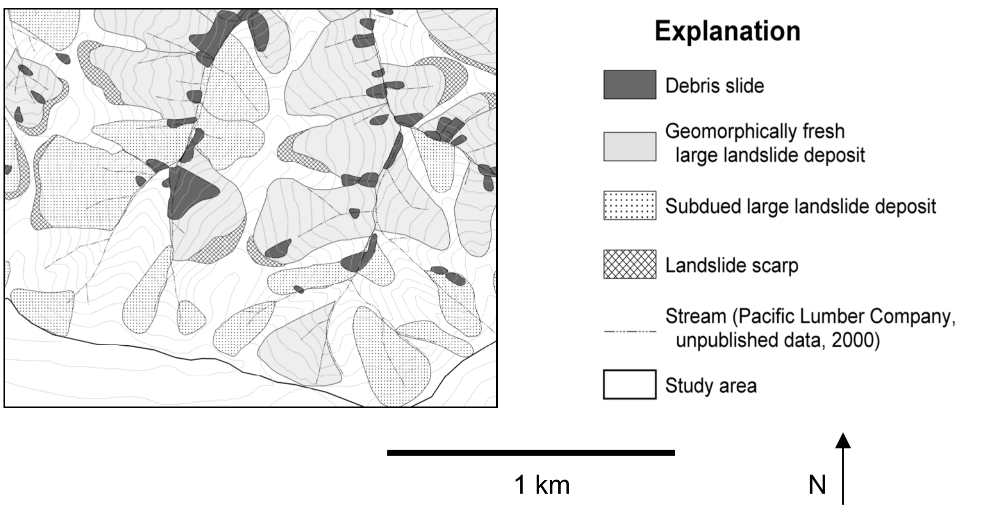


Figure 4—Landslide features in part of Bear-Jordan study area largely underlain by Franciscan Coastal terrane (unit co4). Contour interval 25 m. From de la Fuente and Hicks (2001).

Larabee (fig. 5)—The western part of this study area *(fig. 5A)* is underlain by strata of the Wildcat Group that appear to dip toward Larabee Creek from both north and south. Much of this ground consists of large landslide deposits, on the order of 300 to 1200 m across and as much as 1200 m long, which appear to have been facilitated by the dip slopes. Sometime prior to 1941 (the date of the earliest aerial photographs examined), one of these large landslides transformed into a debris avalanche that extended across Larabee Creek and apparently dammed it temporarily to a depth of about 20 to 30 m. Some steep hillslopes underlain by the Wildcat Group are free of large landslide deposits but show abundant debris slides high in the landscape, as in Freshwater.

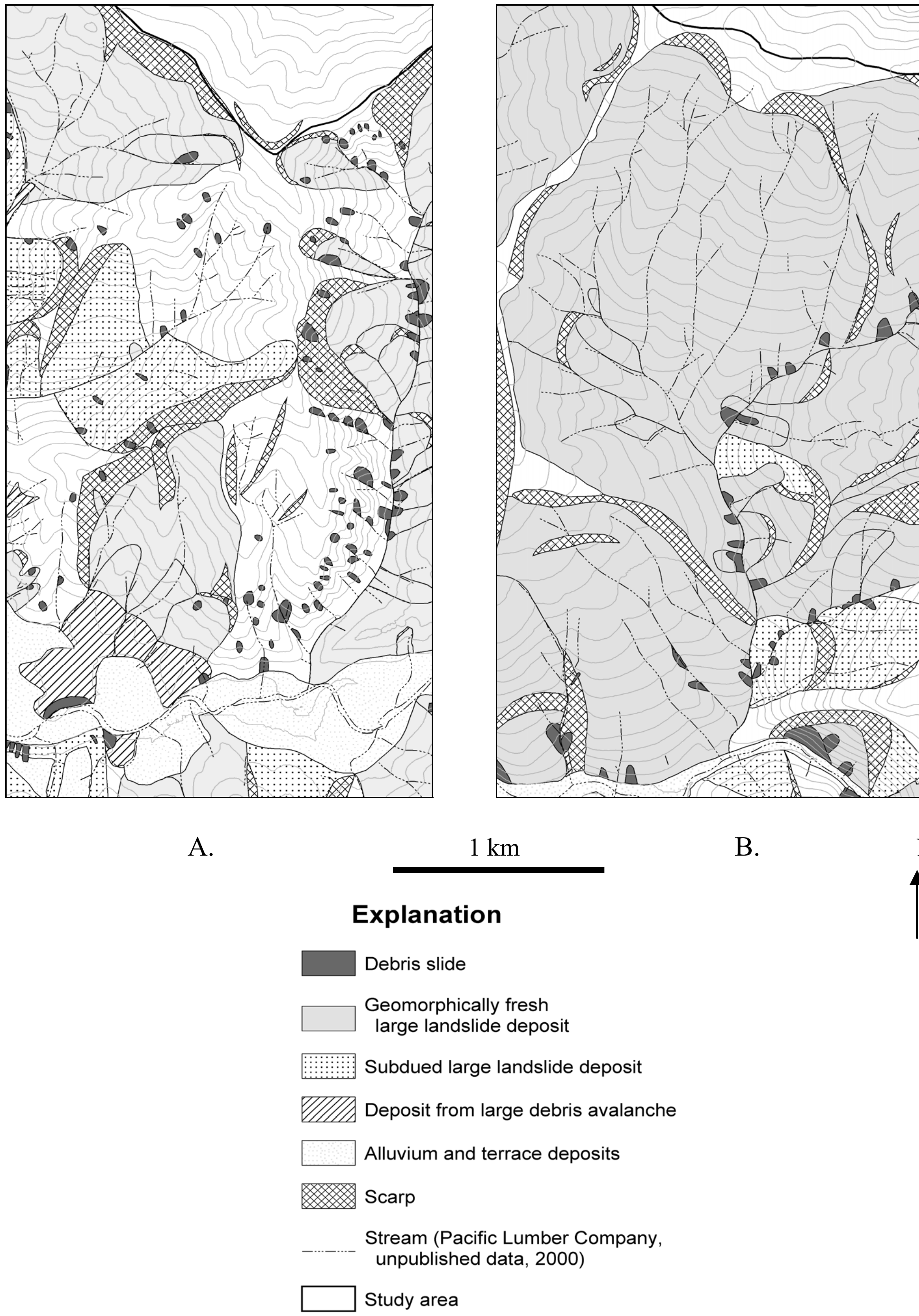


Figure 5—Landslide features in Larabee study area (Ellen 2001). A, Part of western side of study area, underlain by Wildcat Group; B, Part of eastern side of study area, underlain by Franciscan Yager terrane (unit y1). Contour interval 25 m. Larabee Creek crosses southern part of maps.

The eastern part of the study area *(fig. 5B)* is underlain by unit y*1*, the most common subunit of Yager terrane. Unit y1 displays an intermediate topography that includes 1) gently to moderately sloping ground (<35 percent slope) that appears underlain largely by clayey rock, and 2) steep ground that appears underlain largely by more resistant rock dominated by sandstone and lesser conglomerate. The gently to moderately sloping ground is irregular in form and consists of landslide complexes that cover areas as large as several square kilometers and commonly extend from major streams almost to the ridgecrests. In such areas, debris slides are not abundant and occur mainly along creeks. The steep parts of the Yager include more discrete large landslide deposits, large debris slides along Larabee Creek, and unslid ground.

Activity

Debris slides are well documented in the study areas by mapping from multiple generations of aerial photographs. In each study area, the mapping indicates that debris slides have occurred throughout the period of record, most abundantly during photo intervals that include major rainstorms.

Activity of large landslide deposits is not so easily detected because field evidence for movement may be subtle and easily overlooked. During field reconnaissance, we found some evidence for historical activity, such as fresh scarps and displaced roads. Reliable detection of current activity, however, generally requires instrumental monitoring. The absence of all but minimal monitoring in the immediate area of this study leaves major questions for any attempt to quantify sediment sources: Are enough large landslide deposits sufficiently active to constitute significant sources of sediment? Or alternatively, can the large landslide deposits be considered relict, that is, "…clearly developed under different geomorphic or climatic conditions, perhaps thousands of years ago" (Cruden and Varnes 1996)?

Several lines of evidence bear on these questions. 1) Significant activity during the current geomorphic and climatic regime is suggested by the fact that the vast majority of large landslide deposits in the study areas toe along stream channels. These streams are active geomorphic agents responding to the high degree of tectonic activity that characterizes the area (Clarke and Carver 1992). 2) Large landslide deposits display a broad range of geomorphic freshness, as rated during the authors' systematic appraisal of landslide features in the study areas *(figs. 3* through *5)*. Differences in freshness suggest differences in recency, rather than a scenario in which movement is confined to the distant past. 3) Monitoring of hillslope movement in other parts of northern California and adjacent areas reveals significant movement under current climatic conditions and in the absence of earthquake shaking. 4) As reported by Reid and others (2006), debris sliding in the Bear-Jordan and Larabee study areas is significantly more abundant in geomorphically fresh large landslide deposits than in other parts of these study areas. This observation suggests that these fresher large landslide deposits have moved recently enough that shallow materials remain destabilized. These several lines of evidence, taken together, suggest that most large landslide deposits are not relict but rather hold the potential for movement and delivery of sediment to the streams at their toes.

Discussion

Whereas debris slides in the study areas were easily recognized and mapped, and their sediment production tallied routinely during other local studies, the potential for movement of large landslide deposits in this area has been hindered by lack of monitoring. Even modest efforts, such as inventories of displacements along roads following wet winters or sequences of wet winters, would go a long way toward clarifying this issue. Note that even slight movement of these enormous deposits could generate large volumes of sediment. Calculations indicate that 10 cm of yearly movement on six to 15 percent of the large landslide deposits in places like the Larabee study area could provide annual sediment inputs comparable to the impressive contributions from debris sliding documented by Pacific Watershed Associates (1998, 1999) in the Bear Creek and Jordan Creek watersheds.

Another tool mentioned for investigation of large landslide deposits is site-specific stability analysis. Such quantitative analysis, which is common in urban areas, typically is based on borings and engineering testing aimed at determining the geometry, strength, and ground-water conditions at a site. Such methods may be difficult to justify in timberlands, and they may be especially difficult in timberlands underlain by Franciscan rocks because the heterogeneity and disruption of these materials would diminish confidence in subsurface determinations of landslide geometry and material properties. Interrelation of landslides in complexes, which is common in much of the area, would add to the difficulty of stability analysis and diminish its effectiveness.

Conclusions

The ground in much of the region is a mixture of variably deformed materials, and regional geologic mapping by McLaughlin and others (2000) has revealed for the first time the intricate distribution of engineering materials in much of this terrain. These materials are further modified by more recent movement of large landslides, which cover as much as 70 percent or more of the landscape and range from fresh features produced by recent movement to subdued features suggestive of ancient movement followed by long quiescence. In three study areas chosen to be representative of geologic materials in the Eureka area, both debris slides and large landslide deposits are common, and both kinds of landslides show variation in scale, style, and abundance among the different materials represented in the study areas. Debris slides have been shown to be an important source of sediment; movement of large landslide deposits has not been adequately monitored, and even slight movement of a small proportion of these large masses could contribute significant sediment to streams.

Acknowledgments

Much of this work was undertaken for the Mass Wasting Scientific Review Panel, which was convened by the National Marine Fisheries Service, the U.S. Fish and Wildlife Service, the California Department of Fish and Game, and the Pacific Lumber Company as part of the Headwaters agreement. Work was facilitated by Paul Vincent, Alan Foreman, Todd Flackus, and Tim Spear of the GIS Department at the Pacific Lumber Company. Eric Johnson of VESTRA Resources, Inc., also assisted in GIS map preparation and analysis, as did D.L. Brien, J.B. Barnes, J.D. Cecil, and K.A. Cyr of the U.S. Geological Survey. Aerial photographs were provided by the Pacific Lumber Company, the California Geological Survey, and the U.S. Geological Survey Library. The Pacific Lumber Company also provided base maps and access to the ground.

References

Clarke, S.H., Jr.; Carver, G.A. 1992. **Late Holocene tectonics and paleoseismicity, southern Cascadia subduction zone.** Science 255: 188-192.

Cruden, D.M.; Varnes, D.J. 1996. **Landslide types and processes.** In: Turner, A.K.; Schuster, R.L., eds. Landslides: investigation and mitigation. Transportation Research Board Special Report 247. Washington, DC: National Academy Press; 36-75.

de la Fuente, J.; Hicks, B.G. 2001. **Landslide features in the Bear Creek and Jordan Creek study area.** Unpublished mapping for Mass Wasting Scientific Review Panel.

Ellen, S.D. 2001. **Landslide features in the Larabee Creek study area.** Unpublished mapping for Mass Wasting Scientific Review Panel.

Ellen, S.D.; Cannon, S.H.; Reneau, S.L. 1988. **Distribution of debris flows in Marin County.** In: Ellen, S.D.; Wieczorek, G.F., eds**.** Landslides, floods, and marine effects of the storm of January 3-5, 1982, in the San Francisco Bay region, California. U.S. Geological Survey Professional Paper 1434; 113-131.

Ellen, S.D.; Wentworth, C.M. 1995. **Hillside materials and slopes of the San Francisco Bay region.** U.S. Geological Survey Professional Paper 1357; 215 p.

Falls, J.N. 2001. **Landslide features in the Freshwater Creek study area.** Unpublished mapping for Mass Wasting Scientific Review Panel.

Kelsey, H.M. 1980. **A sediment budget and an analysis of geomorphic process in the Van Duzen River basin, north coastal California, 1941-1975: Summary.** Geological Society of America Bulletin 91: 190-195.

Kenney, C. 1984. **Properties and behaviors of soils relevant to slope instability.** In: Brunsden, D.; Prior, D.B., eds. Slope instability. Chichester, UK: John Wiley and Sons; 27-65.

Kirkby, M.J. 1987. **General models of long-term slope evolution through mass movement.** In: Anderson, M.G.; Richards, K.S., eds. Slope stability. Chichester, UK: John Wiley and Sons; 359-379.

McLaughlin, R.J.; Ellen, S.D.; Blake, M.C., Jr.; Jayko, A.S.; Irwin, W.P.; Aalto, K.R.; Carver, G.A.; Clarke, S.H., Jr. 2000. **Geology of the Cape Mendocino, Eureka, Garberville, and southwestern part of the Hayfork 30 × 60 minute quadrangles and adjacent offshore area, northern California.** U.S. Geological Survey Miscellaneous Field Studies Map MF-2336, version 1.0; 26 p., 6 plates. With digital database by Barnes, J.B.; Cecil, J.D.; Cyr, K.A.; 16 p.

Mitchell, J.K. 1976. **Fundamentals of soil behavior.** New York: John Wiley and Sons; 422 p.

Ogle, B.A. 1953. **Geology of the Eel River Valley area, Humboldt County, California.** California Division of Mines Bulletin 164; 128 p., 6 plates.

Pacific Watershed Associates. 1998. **Sediment source investigation and sediment reduction plan for the Bear Creek watershed, Humboldt County, California.** Report prepared for The Pacific Lumber Company, dated April 1998; 42 p. plus maps and appendices.

Pacific Watershed Associates. 1999. **Sediment source investigation and sediment reduction plan for the Jordan Creek watershed, Humboldt County, California.** Report prepared for The Pacific Lumber Company, dated January 1999; 63 p. plus maps and appendices.

Reid, M.E.; Ellen, S.D.; Brien, D.L.; de la Fuente, J.; Falls, J.N.; Hicks, B.G.; Johnson, E.C. 2006. **Predicting debris-slide locations in northwestern California.** In: Standiford, R.B.; Giusti, G.A.; Valachovic, Y.; Zielinski, W.J., technical coordinators. Proceedings of the Redwood Region Forest Science Symposium; 2004 March 15-17; Rohnert Park, CA Gen. Tech. Rep. PSW-GTR-194. Albany, CA: Pacific Southwest Research Station. USDA Forest Service; [this volume].

Spittler, T.E. 1982. **Geology and geomorphic features related to landsliding, Scotia 7.5' quadrangle, Humboldt County, California.** California Department of Conservation, Division of Mines and Geology, Open-File Report OFR 82-20 S.F.

Spittler, T.E. 1983. **Geology and geomorphic features related to landsliding, Redcrest 7.5' quadrangle, Humboldt County, California.** California Department of Conservation, Division of Mines and Geology, Open-File Report OFR 83-17 S.F.

Swanson, F.J.; Swanston, D.N. 1977. **Complex mass-movement terrains in the western Cascade Range, Oregon.** In: Coates, D.R., ed. Landslides. Geological Society of America Reviews in Engineering Geology 3: 113-124.

Predicting Debris-Slide Locations in Northwestern California[1]

Mark E. Reid,[2] Stephen D. Ellen,[3] Dianne L. Brien,[4] Juan de la Fuente,[5] James N. Falls,[6] Billie G. Hicks,[7] and Eric C. Johnson[8]

Abstract

We tested four topographic models for predicting locations of debris-slide sources: 1) slope; 2) proximity to stream; 3) SHALSTAB with "standard" parameters; and 4) debris-slide-prone landforms, which delineates areas similar to "inner gorge" and "headwall swale" using experience-based rules. These approaches were compared in three diverse study areas of northwestern California having multiple inventories of historical non-road-related debris slides in a variety of topographic settings. We implemented the models in a GIS using USGS 10-m digital elevation models (DEMs).

The topographic models show moderate predictive success. Slope performs comparatively well in all study areas. SHALSTAB is rarely superior. The proximity-to-stream model is competitive in one area but falls short in the others. The landforms model performs somewhat better than the others for nearly all the debris-slide data sets in all three areas, and appears especially effective for large debris slides that deliver sediment to streams.

Large landslide deposits also influence the propensity for debris sliding in some areas. The areal density of historical debris-slide sources in steep ground within large, geomorphically fresh landslide deposits can be more than twice that in steep ground outside landslide deposits. Thus, prediction of debris-slide sources can be improved using maps of geomorphically fresh large landslide deposits.

Key words: debris slide, landslide, model, northwestern California, slope, topography

Introduction

Debris slides, shallow, fast-moving landslides from steep slopes, are widely recognized as a significant source of sediment to streams. As part of a study of mass wasting in forested terrain of northwestern California, we tested four topographic models aimed at predicting locations of debris-slide sources. Topographic models can be effective in predicting locations of debris slides because this kind of landslide tends to develop within characteristic parts of the landscape. We examined model

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] U.S. Geological Survey, 345 Middlefield Road, MS-910, Menlo Park, CA 94025. email: mreid@usgs.gov
[3] Retired from U.S. Geological Survey, 116 Huckleberry Trail, Redwood City, CA 94062. email: sdellen@comcast.net
[4] U.S. Geological Survey, 345 Middlefield Road, MS-910, Menlo Park, CA 94025. email: dbrien@usgs.gov
[5] U.S. Forest Service, Klamath National Forest, 1312 Fairlane Road, Yreka, CA 96097. email: jdelafuente@fs.fed.us
[6] California Geological Survey, 2120 Campton Road, Suite D, Eureka, CA 95503. email: jfalls@consrv.ca.gov
[7] Consulting Engineering Geologist, 190 Vista Street, Ashland, OR 97520. email: georegon@aol.com
[8] VESTRA Resources, Inc., 962 Maraglia Street, Redding, CA 96002. email: eric@vestra.com

performance in three diverse study areas, each of which had several maps of historical debris slides prepared by different investigators. We also investigated the relation of debris sliding to deposits of large, slower moving kinds of landslides, which are abundant in the region.

Methods of Model Comparison

The Models

We compared four topographic models: 1) slope (as measured at each 10-m cell), 2) proximity to stream, 3) SHALSTAB with "standard" parameters (Dietrich and others 2001), and 4) debris-slide-prone landforms. Each model was implemented in a geographic information system (GIS) using USGS 10-m digital elevation models (DEMs). The models are generated solely from topography and stream channels, and so can be tested against all available debris-slide data sets.

Each model consists of several categories that form a descending hierarchy of hazard. Categories in the slope model are 1) slopes steeper than 80 percent, 2) slopes 65 to 80 percent, 3) slopes 50 to 65 percent, and 4) slopes 35 to 50 percent. Categories in the proximity-to-stream model are zones (buffers) based on horizontal distance from any class I, II, or III stream plus valley bottom (low flat area adjacent to a stream, similar to a channel migration zone). Here, categories are 0 to 30 m, 30 to 60 m and 60 to 90 m from the stream plus valley bottom. SHALSTAB categories are the values of log (q/T) commonly used by Dietrich and others (2001), where q is defined as effective precipitation and T is soil hydraulic transmissivity.

The landforms model uses rules, based on accumulated geologists' experience with debris sliding in northwestern California, to identify two landforms widely considered prone to debris slides: steep slopes contiguous to streams (comparable, but not identical, to "inner gorge"), and steep convergent areas that lead directly to streams (comparable to "headwall swale"). We created a GIS approach to implement these rules. The "steep to stream" categories of the model consist of steep slopes that extend up from streams until interrupted by a DEM cell with a specified gentler slope. This category may include ground traditionally identified as inner gorge, but can also include contiguous steep ground extending to a ridgeline. "Steep swales" consist of steep, transversely concave areas that are connected to a stream by a continuous transverse concavity. Thus, both these landforms explicitly incorporate direct pathways to the stream system. The model categories form the hierarchy: 1) steep (>65 percent) to stream, 2) steep (>65 percent) swales, 3) steep (>50 percent) to stream, 4) steep (>50 percent) swales, 5) other steep (>65 percent) ground not in the previous categories, and 6) other steep (>50 percent) ground not in the previous categories.

Study Areas

To test the models, we selected three diverse study areas that are representative of the geologic materials, topography, and landslide processes in the area *(fig. 1)*. The areas and their selection are described further by Ellen and others (2006). All three areas have been subjected to rapid tectonic deformation, large rainstorms, strong earthquake shaking, and a hundred-year history of timber-harvest activities. Each study area contains historical debris sliding documented in at least three inventories of debris slides mapped by different investigators, as well as an inventory of larger landslide deposits. Some debris-slide sources are mapped as point locations, others as polygons delineating the source area. Tracks or deposits from debris flows or debris slides were not included in our analyses. In all, we used 16 different data sets of past debris slides spanning multiple triggering events. From these data sets, we used only locations of debris-slide sources that the mapper considered unrelated to roads or other significant grading.

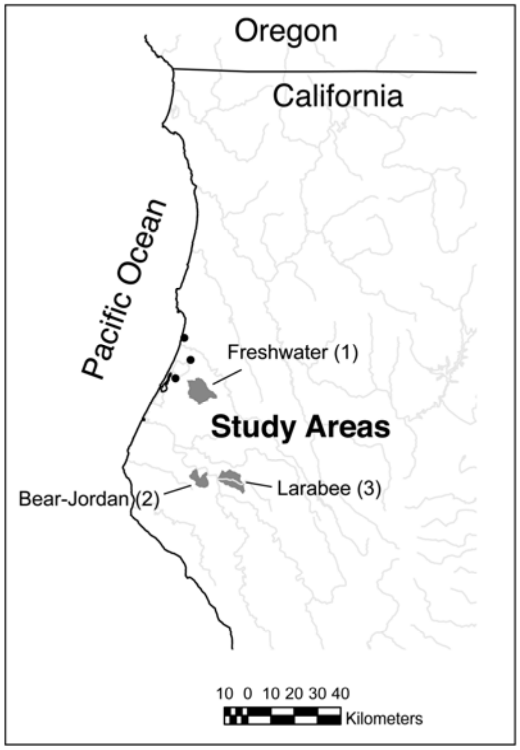


Figure 1—Map showing locations of the three study areas in northwestern California. Dots mark the communities of Eureka, Arcata, and McKinleyville, from south to north.

Debris slides in different study areas tend to originate in different parts of the landscape. In the Freshwater study area, most debris slides initiate from steep upper hillslopes within scalloped or swale topography. In the Bear-Jordan study area, most initiate from steep foot-slopes along the larger streams. Within the Larabee study area, debris slides originate in both of these settings.

Measure of Success

Predictive success was measured using the interplay of two criteria, 1) the percentage of debris-slide cells captured in a hazard class, and 2) the percentage of the study area occupied by that hazard class. This interplay shows the ability of a model to target debris slides, that is, to selectively capture areas of abundant debris sliding without encompassing areas of sparse debris sliding. *Figure 2* explains the resulting plots, which are derived from map data similar to that illustrated in *figure 3*.

This measure allows rational comparison among topographic models regardless of whether debris slides have been mapped as points or polygons. Where debris slides have been mapped as polygons, this measure provides a more conservative appraisal of success than that used in SHALSTAB, wherein an entire slide area is attributed to the highest hazard class of any cell within the slide (Dietrich and others

2001, p. 205). The measure used here makes no assumptions about the location of initial failure within mapped debris-slide polygons.

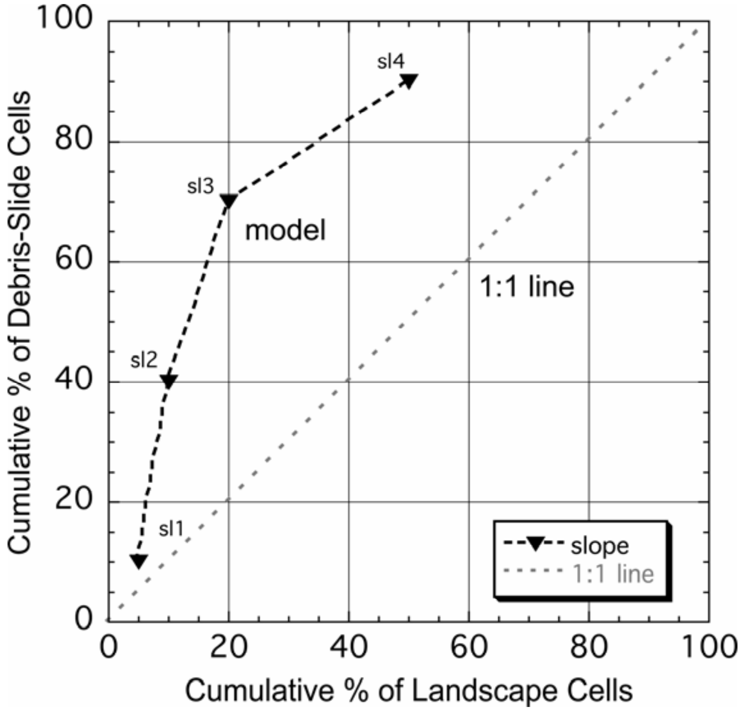


Figure 2—Schematic plot showing the measure of predictive success used in this study, for one topographic model applied to one data set of mapped debris slides. The slope model illustrated here, like all models evaluated in the study, includes several categories (see *fig. 3*). Point SL1 marks the predictive success of the most severe category, slopes greater than 80 percent. This category captures 10 percent of the debris-slide cells while occupying 5 percent of cells in the study area (landscape cells). Point SL2 marks the cumulative success of the two most severe categories, the >80 percent slope class and the 65-80 percent slope class. Thus, point SL2 represents all slopes greater than 65 percent, and these capture 40 percent of the debris-slide cells while occupying 10 percent of the study area. Points SL3 and SL4 include progressively gentler slope categories. The 1:1 line represents random success; at any point on this line, a model captures the same percentage of debris-slide cells as the ground it occupies in the study area. Model performance is best when points lie far from the 1:1 line toward the upper left-hand corner. In this part of the plot, a model captures a large percentage of the debris-slide cells while encompassing only a small percentage of the study area.

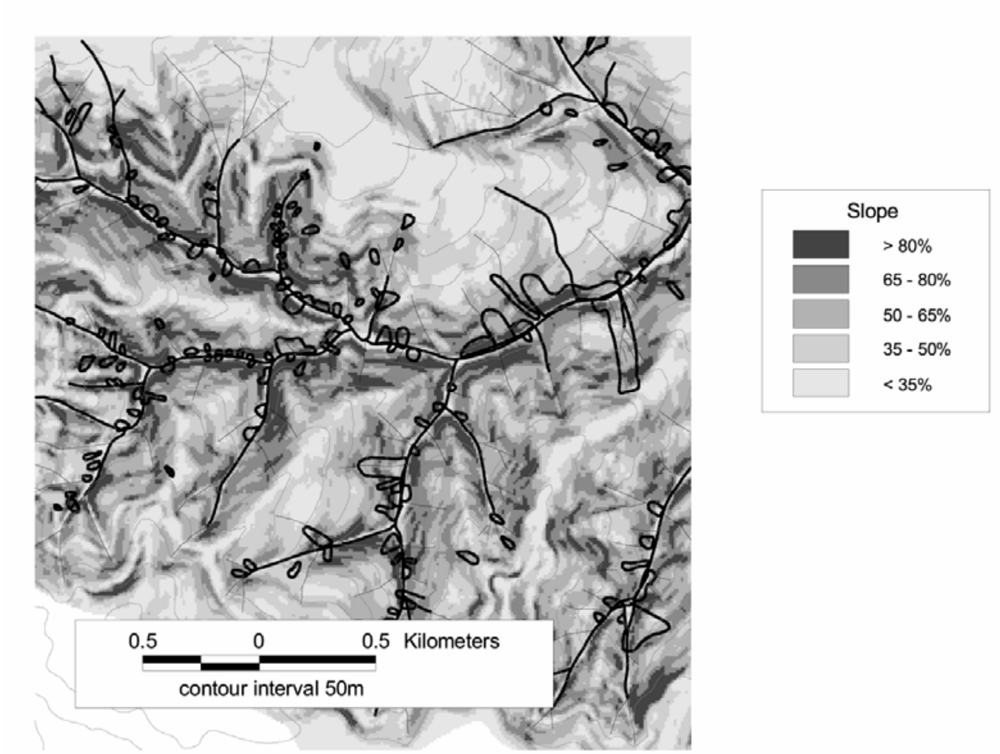


Figure 3—Map showing the slope model in part of the Bear-Jordan study area. Gray tones (or colors) show 10-m DEM cells that fall within the designated slope classes. Black outlines show debris slides mapped as polygons. Note that most debris-slide sources include cells of different slope classes.

Results

Model Comparisons

In most tests, the topographic models display a consistent pattern *(fig. 4)*. The presumed strongest predictors for each model (for example, the steepest hillside slopes) are plotted first and so are represented by the points near the lower left-hand corner of each plot. These points tend to capture the greatest proportion of mapped debris-slide sources in the smallest proportion of land, and so they plot steeply above the origin. As weaker-predictor categories are added (for example, gentler slope intervals), the lines tend to curve over toward the 1:1 line as fewer debris slides are captured per unit area.

The lines generally arch well above the 1:1 line, indicating better than random prediction. In most cases, the models manage to capture about 45 to 75 percent of debris-slide cells before encompassing 20 percent of the ground. Thus, at the resolution of topography and mapping tested here, the models can be considered moderately successful.

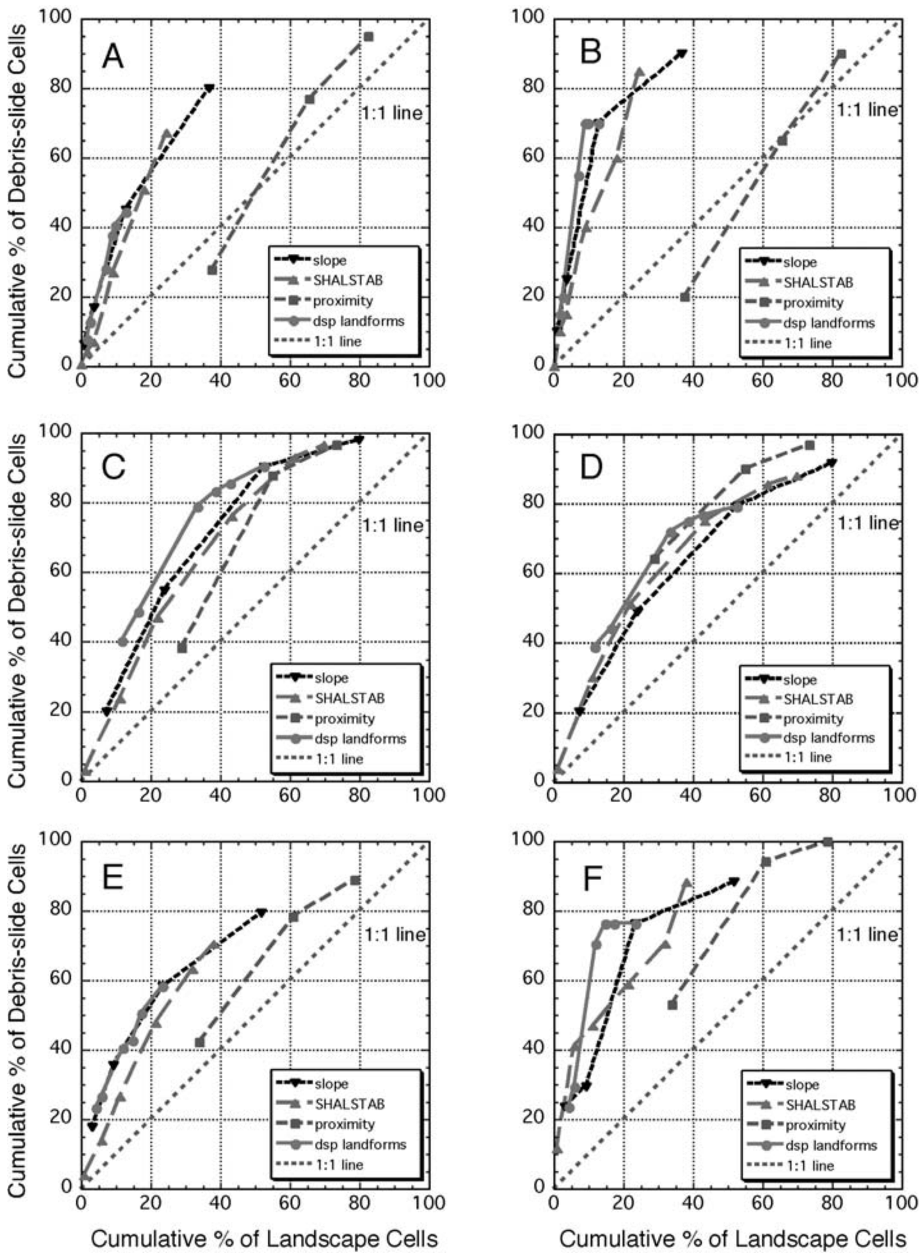


Figure 4—Some of the plots that compare success of models in predicting locations of non-road-related debris slides. Each graph shows results for a different debris-slide data set. A. Using debris slides delivering sediment to streams in Freshwater study area, mapped as points (PWA, 1999b). B. Using large (>2300 m^3) debris slides delivering sediment to streams in Freshwater study area, mapped as points (PWA, 1999b). C. Using debris slides delivering sediment to streams in Bear-Jordan study area, mapped as points (PWA, 1998, 1999a). D. Using debris slides in Bear-Jordan study area, mapped as polygons.[9] Using debris slides in Larabee study area, mapped as polygons.[10] F. Using large (>1500 m^3) debris slides delivering sediment to streams in Larabee study area, mapped as points.[11]

[9] Unpublished mapping from de la Fuente and Hicks, 2001.
[10] Unpublished mapping from Ellen, 2001.
[11] Unpublished mapping from Golder Associates Ltd., 2001.

For many of the data sets, tightly grouped plots indicate that some models are about equally successful *(figs. 4A, 4D, 4E)*. The full spectrum of tests across the different terrains, however, reveals differences in model performance. Slope, the simplest model, performs consistently and comparatively well in all study areas. SHALSTAB is fairly consistent but rarely superior, even in Freshwater where many debris slides initiate from topographic swales *(figs. 4A, 4B)*. The proximity-to-stream model is competitive only in the Bear-Jordan study area, where many debris slides initiate adjacent to streams *(fig. 4D)*. In Freshwater, some categories of this model perform more poorly than random.

The debris-slide-prone landforms model exhibits slightly to somewhat better predictive success than the other models for nearly all the debris-slide data sets in all three study areas. This model has an added advantage, in that the areas it delineates as most hazardous include explicit pathways to the stream system. Some of the comparisons suggest that this model may be especially effective in data sets where points represent large debris slides that deliver sediment directly to streams (compare *figs. 4A* to *4B*, and *4E* to *4F*). This advantage is significant, because a large proportion of the debris-slide sediment delivered to streams in this region commonly originates from a small number of especially large debris slides (Kelsey and others 1995; PWA 1999b, p. 31-33).

Effect of Large Landslide Deposits

In two of the three study areas, large deposits from slower moving kinds of landslides influence the propensity for debris sliding *(fig. 5)*. In Bear-Jordan and Larabee, the areal density of mapped debris-slide sources in steep ground (>65 percent) within geomorphically fresh, large landslide deposits is more than twice that in ground outside landslide deposits *(fig. 6)*. Geomorphic freshness of landslide deposits was rated systematically using qualitative measures of apparent freshness of landslide scarps, toes, and internal topography. The Freshwater study area did not show this relation.

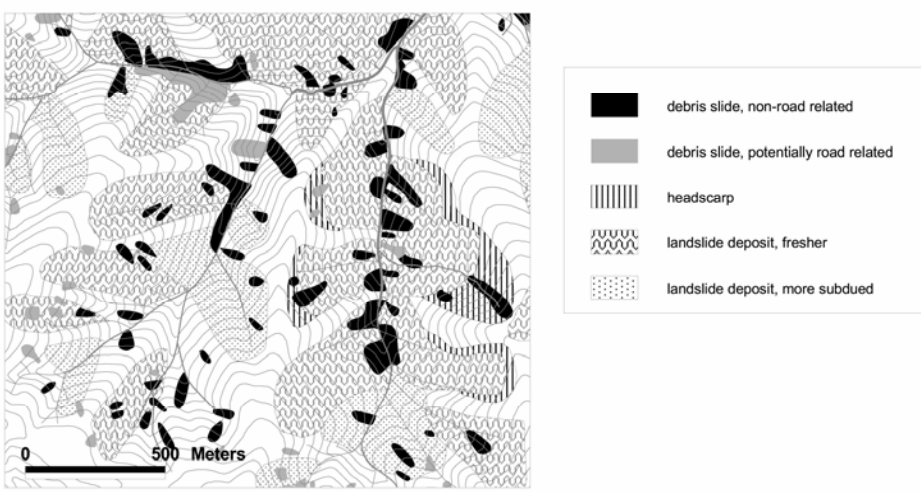


Figure 5—Map of part of the Bear-Jordan study area, showing relation between debris slides and large landslide deposits.[12]

[12] Unpublished mapping from de la Fuente and Hicks, 2001.

To determine whether the differences shown in *figure 6* are significant statistically, we examined the differences using a one-sided chi-square statistical test (Conover 1980). A one-sided test for discrete (not continuous) data is useful here because rejection of the null hypothesis indicates that the areal density in one category is greater than in the other category. For each area, the statistical tests indicate that debris-slide areal density is higher in fresher landslide deposits than in more subdued landslide deposits or in ground outside landslide deposits; in both study areas the tests reject the null hypothesis at a 95 percent confidence level.

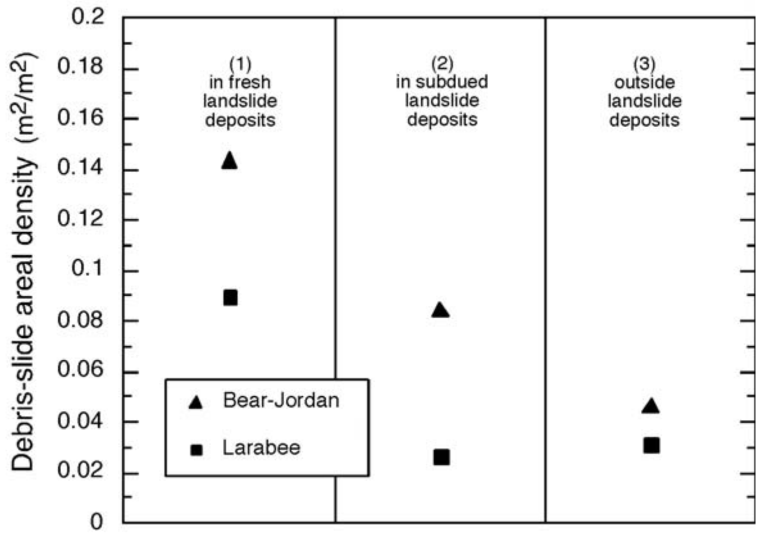


Figure 6—Plot showing relation of debris slides, from one data set in each study area, to mapped large landslide deposits in steep (>65 percent) ground of the Bear-Jordan and Larabee study areas. Areal density, the proportion of ground mapped as debris-slide sources, is distinguished for debris slides located in fresher landslide deposits, in more subdued landslide deposits, and outside of landslide deposits.

Discussion

The topographic models compared here meld aspects of local slope, topographic convergence, and pathways to streams in different combinations to predict future debris-slide locations; all these topographic factors can be obtained from a DEM. Model success varies in different terrain, however local slope is a strong predictor in all the study areas. Slope alone often performs better than slope plus topographic convergence (as represented by SHALSTAB); thus convergence appears to be a detriment to SHALSTAB's performance in some cases. The general success of the debris-slide-prone landforms model, for both point and polygon data in all three study areas, indicates that it targets debris-slide sources slightly better than just slope or slope plus convergence (SHALSTAB). The model identifies landforms that appear to be prime locations for debris slides, probably because of factors related to landform evolution beyond just slope and convergence. The apparent success of this model in identifying areas prone to large delivering debris slides likely results from the pathways to streams incorporated in the model. This aspect of the model's success could be significant in assessing sediment delivery and merits further testing.

The debris-slide-prone landforms model requires two factors in addition to slope to distinguish the intended landforms: 1) the stream system, including flat or gently sloping ground that forms part of the valley bottom; and 2) the degree of concavity that defines steep swales and their path to the stream system. Both these items can be identified from a DEM. In some terrains, these factors may need to be adjusted to properly distinguish the intended landforms within the regional topography.

One might anticipate that topographic model performance could be enhanced by “calibrating” factors to specific terrain. For example, slope or proximity-to-stream intervals could be adjusted to improve predictive success. However, the overall performance of specific topographic models cannot be improved significantly by choosing different model categories than those used in this study. Different or additional points might change the result of an individual category somewhat, but overall performance still must pass through the points on the current plots.

The use of DEMs with higher resolution than those used in this study may change the relative performance of these topographic models, and the performance of specific models may improve markedly. To test the effect of higher DEM resolution, however, will require new debris-slide mapping, in which slides observed in photographs or in the field are plotted directly on base maps made from the new DEM. Otherwise, tests will be jeopardized by mislocation. The models and mapping reported here used the same topographic base, in that the 10-m DEM used for the models is derived directly from the USGS 7-1/2’ contours used in the mapping.

Conclusions

The topographic models we tested, when run on 10-m DEMs, generally show a similar pattern of prediction, and in many cases perform comparably, in a variety of terrain representative of northwestern California. They generally target past debris-slide sources at moderate levels of success, typically capturing 45 to 75 percent of debris-slide cells in about 20 percent of the ground. When all test results are compared, some differences emerge. Slope, the simplest model, performs consistently and comparatively well in all the study areas. SHALSTAB is fairly consistent but rarely superior, even in terrain dominated by topographic swales. The proximity-to-stream model is competitive in one study area but falls short in the others. The debris-slide-prone landforms model consistently performs as well or better than the others and may excel in capturing large debris slides that deliver directly to streams. In two of the three study areas, debris slides are markedly concentrated within geomorphically fresh, large landslide deposits. Delineation of areas susceptible to debris sliding in northwestern California can be improved by making use of a good topographic model and mapping of large, geomorphically fresh landslide deposits.

Acknowledgments

This work was undertaken for the Mass Wasting Scientific Review Panel, which was convened by the National Marine Fisheries Service, the U.S. Fish and Wildlife Service, the California Department of Fish and Game, and the Pacific Lumber Company. We appreciate the assistance of personnel at Pacific Watershed Associates, the Pacific Lumber Company, the California Geological Survey, and Golder Associates Ltd. in providing maps and data for analysis. Dino Bellugi at the University of California, Berkeley, helped get SHALSTAB up and running.

References

Conover, W.J. 1980. **Practical nonparametric statistics**. New York: John Wiley and Sons.

Dietrich, W.E.; Bellugi, D.; de Asua, R.R. 2001. **Validation of the shallow landslide model, SHALSTAB, for forest management**. In: Wigmosta, M.S.; Burges, S.J., eds. Land use and watersheds: human influence on hydrology and geomorphology in urban and forest areas. Water Science and Application Vol. 2. Washington, DC: American Geophysical Union; 195-227.

Ellen, S.D.; de la Fuente, J.; Falls, J.N.; McLaughlin, R.J. 2006. **Overview of the ground and its movement in part of northwestern California**. In: Standiford, R.B.; Giusti, G.A.; Valachovic, Y.; Zielinski, W.J., technical coordinators. Proceedings of the Redwood Region Forest Science Symposium; 2004 March 15-17; Rohnert Park, CA Gen. Tech. Rep. PSW-GTR-194. Albany, CA: Pacific Southwest Research Station. USDA Forest Service; [this volume].

Kelsey, H.M.; Coghlan, M.; Pitlick, J.; Best, D. 1995. **Geomorphic analysis of streamside landslides in the Redwood Creek basin, northwestern California**. In: Nolan, K.M.; Kelsey, H.M.; Marron, D.C., eds. Geomorphic processes and aquatic habitat in the Redwood Creek Basin, northwestern California. U.S. Geological Survey Professional Paper 1454; J1-J12.

Pacific Watershed Associates (PWA). 1998. **Sediment source investigation and sediment reduction plan for the Bear Creek watershed, Humboldt County, California**. Report prepared for The Pacific Lumber Company, dated April 1998; 42 p.

Pacific Watershed Associates (PWA). 1999a. **Sediment source investigation and sediment reduction plan for the Jordan Creek watershed, Humboldt County, California**. Report prepared for The Pacific Lumber Company, dated January 1999; 63 p.

Pacific Watershed Associates (PWA). 1999b. **Sediment source investigation and sediment reduction plan for the Freshwater Creek watershed, Humboldt County, California**. Report prepared for The Pacific Lumber Company, dated September 1999; 94 p.

Erosion Rates Over Millennial and Decadal Timescales at Caspar Creek and Redwood Creek, Northern California[1]

Ken L. Ferrier,[2] James W. Kirchner,[3] and Robert C. Finkel[4]

Erosion rate measurements are essential for modeling landscape evolution and for understanding how sediment loading affects stream ecosystems. Traditionally, erosion rates have been determined by measuring stream sediment fluxes over timescales of 10 to 100 years. At Caspar Creek and Redwood Creek in northern California, stream sediment fluxes have been measured since 1963 and 1973, respectively (Henry 1998, Nolan and others 1995), and these studies have yielded two of the longest and most detailed records of stream sediment loading in northern California.

We have applied a new technique (Brown and others 1995, Granger and others 1996) at Caspar Creek and Redwood Creek to measure erosion rates averaged over the past several millennia. The concentration of cosmogenic nuclides such as ^{10}Be in stream sediments can be used to estimate whole-catchment erosion rates averaged over thousands of years, a timescale that is unobservable by conventional methods.

Cosmogenic nuclides are isotopes produced through interaction with the cosmic radiation that continually bombards the Earth; ^{10}Be, for example, is an isotope produced in quartz only when quartz is exposed to cosmogenic neutrons and muons (for example, Lal 1991). Since these neutrons and muons only penetrate the upper few meters of soil and bedrock, the concentration of ^{10}Be in quartz reveals how long that quartz was within the penetration depth of cosmic radiation. A high concentration of ^{10}Be in quartz indicates a long exposure time, and hence a slow erosion rate; conversely, a low concentration of ^{10}Be indicates a short exposure time and a fast erosion rate. By measuring the concentration of ^{10}Be in well-mixed stream sediment, we assume that we obtain an erosion rate that is a representative average of the rates of erosional processes upstream of the sample collection site. This method yields an erosion rate that is an average in time (over the past several thousand years) and an average in space (over the drainage basin area).

Comparing long-term (>1000 year) erosion rates from cosmogenic nuclides with short-term (<100 year) sediment yields can shed light on erosional processes and on the effects of land use on sediment delivery to streams. Using cosmogenic ^{10}Be, we measured erosion rates averaged over the past several thousand years at six sites in Caspar Creek and four sites in Redwood Creek, and compared these rates to sediment

---

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] University of California at Berkeley, Department of Earth and Planetary Science, McCone Hall, Berkeley, CA, 94720. email: ferrier@eps.berkeley.edu.
[3] University of California at Berkeley, Department of Earth and Planetary Science, McCone Hall, Berkeley, CA, 94720.
[4] Center for Accelerator Mass Spectrometry, Lawrence Livermore National Laboratory, Livermore, CA, 94551.

yields measured at Caspar Creek since 1963 and at Redwood Creek since 1974. Millennial-scale erosion rates at Caspar Creek agree with each other within error, and long-term erosion rates at our Redwood Creek field sites vary by about a factor of 3 between basins. Our cosmogenic ^{10}Be measurements at Caspar Creek imply an average erosion rate of 0.09 mm/yr, which agrees with short-term sediment yields (Cafferata and Spittler 1998) within error. Our cosmogenic ^{10}Be measurements at Redwood Creek indicate an average long-term erosion rate of ~0.4 mm/yr, which is in rough agreement with measurements of stream sediment flux over the past several decades. These results imply that sediment yields measured at Caspar Creek and Redwood Creek over the past few decades are broadly consistent with long-term average rates of sediment production by hillslope processes.

References

Brown, E.T.; Stallard, R.F.; Larsen, M.C.; Raisbeck, G.M.; Yiou, F. 1995. **Denudation rates determined from the accumulation of in situ-produced ^{10}Be in the Luquillo Experimental Forest, Puerto Rico**. Earth and Planetary Science Letters 129: 193-202.

Cafferata, P.H.; Spittler, T.E. 1998. **Logging impacts of the 1970's vs. the 1990's in the Caspar Creek watershed**. In: Proceedings of the conference on coastal watersheds: the Caspar Creek story; 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 103-115.

Granger, D.E.; Kirchner, J.W.; Finkel, R. 1996. **Spatially averaged long-term erosion rates measured from in situ-produced cosmogenic nuclides in alluvial sediment**. Journal of Geology 104: 249-257.

Henry, N. 1998. **Overview of the Caspar Creek watershed study**. In: Proceedings of the conference on coastal watersheds: the Caspar Creek story; 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 1-9.

Lal, D. 1991. **Cosmic ray labeling of erosion surfaces: in situ production rates and erosion models**. Earth and Planetary Science Letters 104: 424-439.

Nolan, K.M.; Kelsey, H.M.; Marron, B.C. 1995. **Summary of research in the Redwood Creek basin, 1973-1983**. Geomorphic processes and aquatic habitat in the Redwood Creek basin, northwestern California. U.S. Geological Survey Professional Paper 1454-A.

Decision Support for Road Decommissioning and Restoration by Using Genetic Algorithms and Dynamic Programming[1]

Elizabeth A. Eschenbach,[2] Rebecca Teasley,[3] Carlos Diaz,[4] and Mary Ann Madej[5]

Abstract

Sediment contributions from unpaved forest roads have contributed to the degradation of anadromous fisheries streams in the Pacific Northwest. Efforts to reduce this degradation have included road decommissioning and road upgrading. These expensive activities have usually been implemented on a site specific basis without considering the sediment contributions from all roads within a watershed.

This paper describes results from optimization models developed for determining road removal management plans within a watershed. These models consider the tradeoffs between the cost and effectiveness of different treatment strategies to determine a treatment policy that minimizes the predicted sediment erosion from all forest roads within a watershed, while meeting a specified budget constraint.

Two optimization models are developed using dynamic programming and genetic algorithms. Each model accepts road survey data from the Redwood National Park's (RNSP) GIS layers for a watershed with approximately 700 road segments and stream crossings. The models also require treatment effectiveness data, which are derived from previous published studies for the same area. The output from the model is the treatment level for each road segment and crossing and the total cost of the road removal management plan. The output is then exported to the GIS.

The models currently consider only road removal, but could be expanded to include additional road modifications or watershed restoration projects. Our approach is portable to other watersheds.

Key words: optimal watershed management, road removal, sediment

Introduction

Abandoned and unmaintained logging roads are common across the steep, forested landscapes of western North America and present concerns as a major sediment source (Best and others 1995, Janda and others 1975, Megahan and Kidd 1972). Few studies have evaluated long-term and watershed-scale changes to sediment yields as the roads are abandoned, removed or restored. Madej (2001) reported on the post-treatment erosion in Redwood National Park after a 12-year

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Environmental Resources Engineering, Humboldt State University HS 18, 1 Harpst St., Arcata, CA 95521, (707) 826-4348. email: eae1@humboldt.edu
[3] Civil Engineering; University of Texas, Austin. email: rteasley@mail.utexas.edu
[4] Civil and Environmental Engineering, Stanford University. email: cad10@stanford.edu;
[5] U.S. Geological Survey. email: mary_ann_madej@usgs.gov

recurrence-interval storm, and provides a measure of the effectiveness of different road and crossing treatment methods.

Figure 1 summarizes the road treatments evaluated in Madej (2001) that are used in this study. Road segments and stream crossings receive different types of treatments. For road segments (lengths of road between stream crossings) four road treatment alternatives (including no treatment) were assessed which varied in the amount of earth-moving involved *(fig. 1a–d)*. The least intensive treatment decompacts the road surface and constructs drains perpendicular to the road alignment to dewater the inboard ditch—a technique referred to as 'ripped and drained' *(fig. 1b)*. This treatment moves 200 to 500 m^3 of road fill for every kilometer of road treated. More intensive treatment methods include partially outsloping the road surface by excavating fill from the outboard edge of the road and placing the material in the inboard ditch at the base of the cutbank *(fig. 1c)*. This technique requires more earth-moving (1000 to 2000 m^3/km of treated road). Complete recontouring of the road bench is called "total outslope" *(fig. 1d)*. The cutbank is covered by excavated fill, and the original topsoil from the outboard edge of the road is replaced on the road bench where possible. Total outsloping involves moving an average of 6000 m^3/km of treated road.

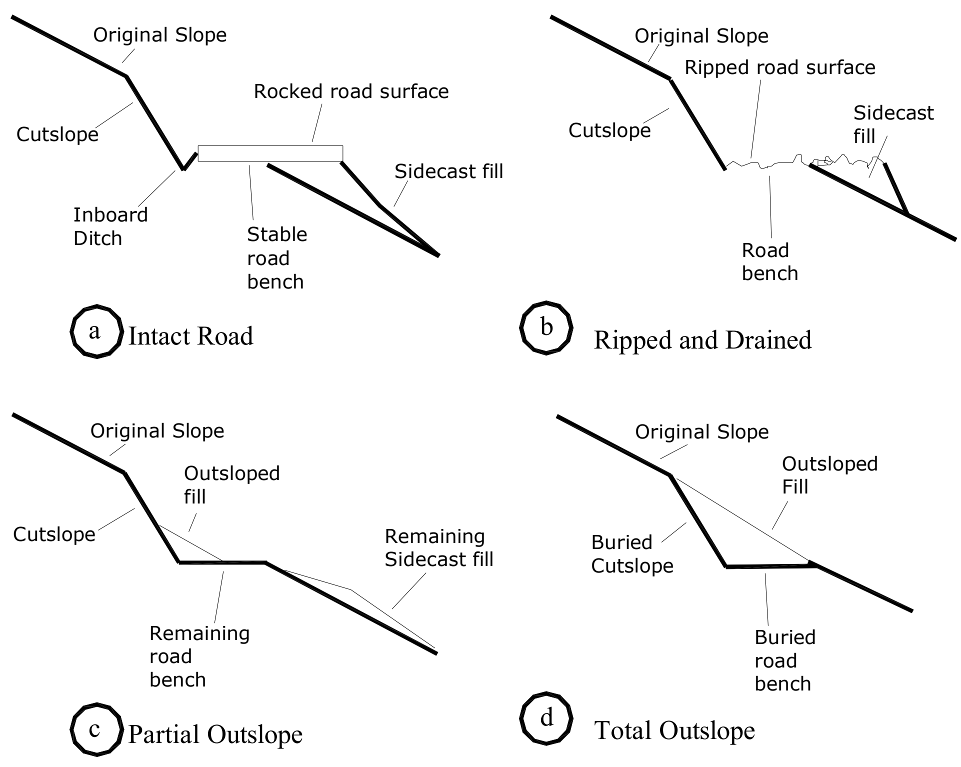


Figure 1—Road treatment methods described in Madej (2001).

Stream crossings are treated by excavating road fill overlying a culvert, removing the culvert and grading a new channel form. "Basic excavation" removes the culvert and establishes a channel in the previous culvert location. "Total excavation" removes more road fill, creates a channel at the elevation of the original stream channel, and excavates sediment deposited upstream of the crossing, if present.

Up to now, few watershed level policies for managing sediment contributions from logging roads have been developed as there has been a lack of information about the effectiveness of different road and crossing treatment methods. Given the effectiveness measures provided by Madej (2001), optimization methods can be implemented to consider trade-offs between cost and sediment savings over an entire watershed. One of the few uses of applied optimization to develop road removal policy was by Tomberlin and others (2002). They report using Stochastic Dynamic Programming to determine if a road in the Casper Creek watershed should be left alone, upgraded or removed based on its erosion potential.

This paper describes the development of two optimization models that are used to determine the level of treatment for removing roads within a watershed, using a strategy that maximizes the sediment saved from critical habitat, while maintaining a specified budget. These two models consider tradeoffs in effectiveness and cost across a watershed.

Methods

Dynamic programming and genetic algorithms are used to determine the best combination of road removal strategies that minimize sediment erosion to a stream (or maximize the sediment saved from entering a stream channel). The problem is formulated with the objective: *Maximize the sediment saved from entering a stream channel as a function of road and crossing treatment levels*. The optimization problem is constrained by the budget and by the existing treatment methods. The problems is stated mathematically as follows

$$\max_{\forall x_r, x_c} z = \{\sum_{\forall r} W_r L_r S_r(x_r) + \sum_{\forall c} W_c V_c S_c(x_c)\} \qquad \textit{Equation} \quad 1$$

subject to

$$TC = \sum_{\forall r} L_r C_r(x_r) + \sum_{\forall c} V_c C_c(x_c) \le B \qquad \textit{Equation} \quad 2$$

$$x_r = 0,1,2,3 \quad x_c = 01,2 \qquad \textit{Equation} \quad 3$$

Where

S_r = sediment saved / mile on road segment r

S_c = sediment saved / cubic yard on crossing c

x_r = treatment level for road r

x_c = treatment level for crossing c

L_r = length of road segment r in miles

V_c = volume of crossing c in cubic yards

W_r = critical habitat weighting factor for road *r*

W_c = critical habitat weighting factor for crossing *c*

TC = total cost of all road and crossing treatments in $

C_r = cost in dollars / mile to treat road segment r

C_c = cost in dollars / cubic yard to treat crossing c

B = budget in dollars

x_r = 4 road treatment methods *(fig. 1)*

x_c = 3 road crossing treatment methods *(table 3)*

The formulation above allows for the weighting of sediment depending on its location or importance to habitat within the watershed via the weighting factors for roads and crossings: W_r. and W_c

Genetic Algorithms (GA) are based on the mechanics of natural selection and genetics, where the most “fit” of randomly generated solutions are allowed to “mate” with the hope of creating more “fit” solutions (Holland 1992). Each solution is a “chromosome” that is made up of a string of “genes” where each gene carries an integer value that represents the level of treatment applied to a road or crossing. The “fitness” of each chromosome (solution) is measured by the objective function. Mating occurs via Selection, Crossover, and Mutation to combine the more fit solutions into a new generation of solutions. In Selection, chromosomes with higher fitness have a higher probability of mating. In Crossover, each member’s chromosome is sliced in two locations and the center pieces are swapped with each other *(fig. 2)*. Mutation is the random alteration of genes in randomly selected chromosomes to diversify the population. Generations of chromosome populations are generated iteratively until a near global optimum is achieved.

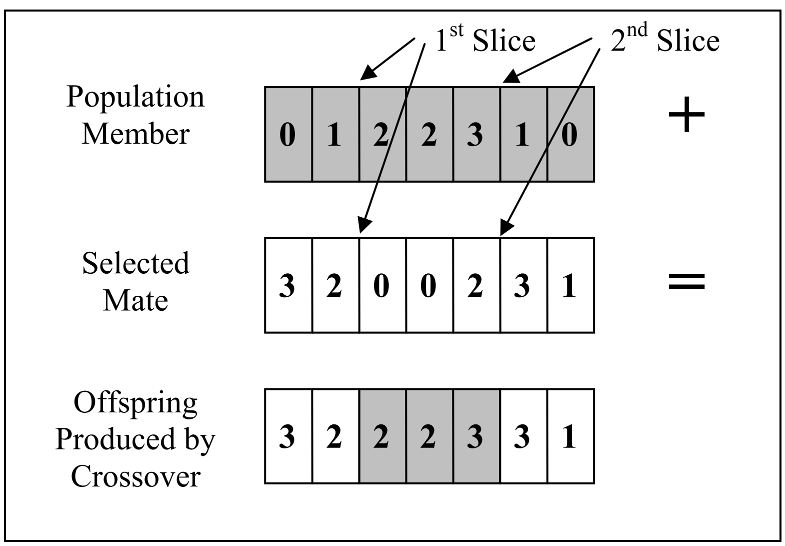


Figure 2—Example of crossover methodology used in genetic algorithms.

One of the strengths of genetic algorithms is they can solve large complex problems that are not solvable with traditional optimization methods that require a differentiable description of the problem. A drawback of GAs is that it is a heuristic method and one cannot prove the optimal solution has been obtained (Goldberg 1989).

We use the Generator™ to build and run the GA. This software is easy to use and runs through an Excel interface. The problem is formulated with a penalty function as provided in Equation 4 in order to meet the requirements of the software. All variables have been defined above, except for P, the penalty.

$$\max_{\forall x_r, x_c} z = \left[\sum_{\forall r} \{W_r L_r S_r(x_r)\} + \sum_{\forall c} \{W_c V_c S_c(x_c)\} \right] - P * |B - TC| \quad \textit{Equation} \quad 4$$

The penalty term is used for numerical stability for the GA. The penalty term pushes the solution toward those solutions that use all the budget B for the total cost *TC* of the solution, i.e. in order to maximize the entire quantity, the penalty term, (the difference between *B* and *TC*) should be small.

The dynamic programming (DP) approach (Bellman 1957) separates the problem into a series of subproblems using stages and states. Each stage has a number of states. The stages are each of the roads and crossings. The states are the amount of remaining budget available to spend to treat that road or crossing. Once each subproblem is solved, one can forward simulate through all the solutions to determine the optimal treatment for each road and crossing that meets the specified budget.

The dynamic program has the following formulation which is a resource allocation DP. Given the End Condition, where $N = N_c + N_r$

$$f_N(R_N) = \max_{x_N} \{W_N V_N S_N(x_N)\} \quad \textit{Equation} \quad 5$$

The recursive equation is solved for $n = N_c + N_r - 1, \ldots, 1$

For $n = N_c + N_r, \ldots, N_{r+1}$, the recursive equation for crossings is

$$f_n(R_n) = \max_{x_n} \left\{ W_n V_n S_n(x_n) + f_{n+1}(R_n - V_n C_n(x_n) \right\} \quad \textit{Equation} \quad 6$$

For $n = N_r, \ldots, 1$, the recursive equation for roads is

$$f_n(R_n) = \max_{x_n} \left\{ W_n L_n S_n(x_n) + f_{n+1}(R_n - L_n C_n(x_n) \right\} \quad \textit{Equation} \quad 7$$

Where

$N = N_c + N_r$= the total number of roads and total number of crossings

R_n= the amount of remaining budget for treatment of road or crossing *n*

$C_m(x_n)$=the cost to treat road or crossing n at treatment level x_n

$f_{n+1}(R_n - L_n C_n(x_n))$=the maximum amount of sediment saved using the budget remaining after treating road or crossing n at treatment level x_n at cost C_n. Other variables are previously defined.

The dynamic program is subject to the following constraint:

$$TC = \sum_{n=N_r+1}^{N_r+N_c} V_n C_n(x_n) + \sum_{n=1}^{N_r} L_n C_n(x_n) \leq B \quad Equation \quad 9$$

A strength of the DP approach is that a global optimum is guaranteed. A drawback of DP is the "curse of dimensionality," where the computation requirements grows exponentially as the problem size increases. However, using a resource allocation formulation, the computational requirements grow linearly in the number of roads and crossings considered.

The optimization algorithms are applied to a sample watershed—the Lost Man Creek Basin in RNSP that has approximately 32 miles of roads and 73 crossings. A field-based road inventory was used to generate a GIS data base with 618 different road segments. *Table 1* provides a summary of the distribution of roads and crossings through the basin. Given four possible road treatments and three possible crossing treatments, the total possible policies for this basin is (4^{618}) x (3^{73}). This number of policies is much too large to examine individually. Optimization algorithms provide a rational method to consider such a large number of policies.

Table 1—*Number of Crossing and Roads for Each Hillslope Position in the Lost Man Creek Basin.*

Hillslope position	**Number of crossings**	**Number of road segments**
Lower	49	196
Middle	19	257
Upper	5	165

Tables 2 and *3* show the sediment saved and associated costs for both roads and crossing treatments. These data are based on Madej (2001) and decommissioning work conducted in RNSP from 1978-1996. Both the potential sediment saved and the cost for treatment increase for roads in the lower slopes of the watershed, i.e. steep slopes closest to the stream Each table provides a cost-benefit ratio, denoting the ratio of money spent to save a cubic yard of sediment. As one might expect, crossing treatments *(table 3)*, in general, have the best cost-benefit ratios. The road treatments with the best cost-benefit ratio occur in the lower slopes *(table 2)*.

Table 2—*Potential sediment savings and associated costs for 4 road treatments.*

Hillslope location and level of treatment	Sediment saved (yd^3/mi)	Cost/mile of road ($/mi)[1]	Cost-benefit ratio ($/$yd^3$)
Upper slopes			
No treatment	0	$0	0.0
Ripped & drained	250	$5,280	21.1
Partial outslope	400	$7,920	19.8
Total outslope	490	$15,840	32.3
Middle slopes			
No treatment	0	$0	0.0
Ripped & drained	300	$5,280	17.6
Partial outslope	650	$7,920	12.2
Total outslope	950	$21,120	22.2
Lower slopes			
No treatment	0	$0	0.0
Ripped & drained	1000	$6,600	6.6
Partial outslope	2000	$7,920	4.0
Total outslope	2500	$26,400	10.6

[1]Costs are based on decommissioning work conducted in RNSP from 1978-1996

Table 3—*Potential sediment savings and associated costs for 3 crossing treatments.*

Crossings - Hillslope location & level of treatment	Sediment saved (yd^3)	Cost/ crossing ($)	Cost-benefit ratio ($/$yd^3$)
Upper slopes			
No treatment	0	$0	0.0
Basic excavation	300	$1,200	4.0
Total excavation	400	$2,100	5.3
Middle slopes			
No treatment	0	$0	0.0
Basic excavation	600	$2,400	4.0
Total excavation	800	$3,500	4.4
Lower slopes			
No treatment	0	$0	0.0
Basic excavation	1000	$3,600	3.6
Total excavation	1200	$5,250	4.4

Results

Table 4 provides a summary of the costs, sediment saved and overall cost-benefit ratio for the Dynamic Program model and a uniform policy of the minimal treatment, where both crossings and roads have the lowest level of treatment of basic excavation *(table 3)* and rip and drain *(fig. 1)* respectively. The uniform policy represents a non-optimized approach to allocate treatments throughout the basin.

Table 4—*Comparison of dynamic program and uniform policy results.*

Policy	**Cost ($)**			**Sediment saved (yd3)**			**Cost/benefit ratio ($/yd3)**
	Roads	Crossings	**Total**	Roads	Crossings	**Total**	
DP	152,451	97,250	**249,701**	24,755	15,062	**39,817**	6.3
Uniform Minimum	178,230	164,960	**343,190**	15,183	14,892	**30,075**	11.4

Table 5 summarizes the policies generated by the dynamic program and the genetic algorithm for two budgets of $250,000 and $500,000 by reporting costs, sediment saved and an overall cost/benefit ratio.

Table 5—*Comparison of dynamic program and genetic algorithm policies for $250K and $500K budgets.*

Budget constraint ($)	**Optimization method**	**Cost ($)**			**Sediment saved (yd^3)**			**Cost/ benefit ratio ($\$/yd^3$)**
		Roads	Crossings	**Total**	Roads	Crossings	**Total**	
250,000	DP	152,451	97,250	**249,701**	24,755	15,062	**39,817**	6.3
	GA	153,783	96,200	**249,983**	23,600	14,476	**38,076**	6.6
500,000	DP	347,036	153,010	**500,046**	32,293	17,352	**49,645**	10.1
	GA	346,789	153,200	**499,989**	32,123	17,029	**49,152**	10.2

Figures 3 and *4* summarize the treatment policies developed by the dynamic program model for 4 budget scenarios: $250K, $500K, $750K and 1 million dollars for crossings and road segments.

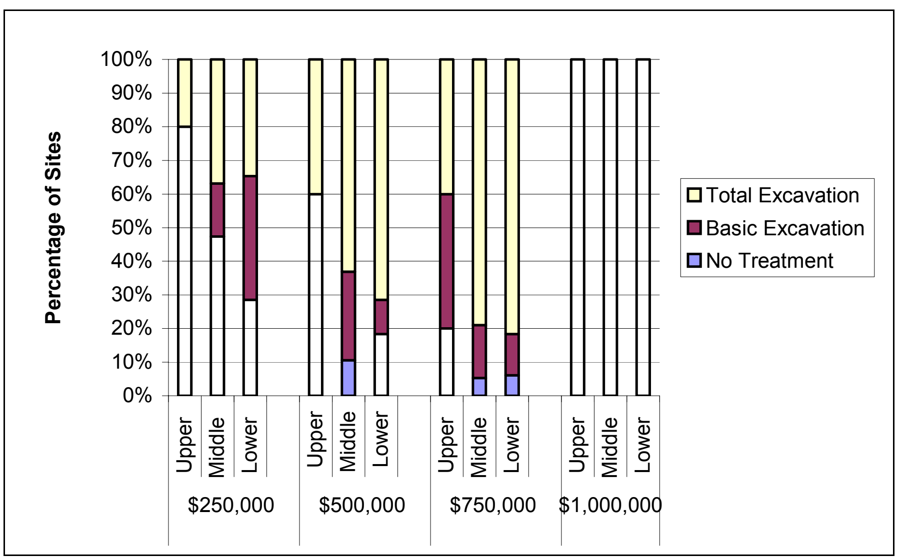


Figure 3—Comparison of dynamic programming crossing treatment policies developed with budgets of $250,000, $500,000, $750,000 and $1,000,000.

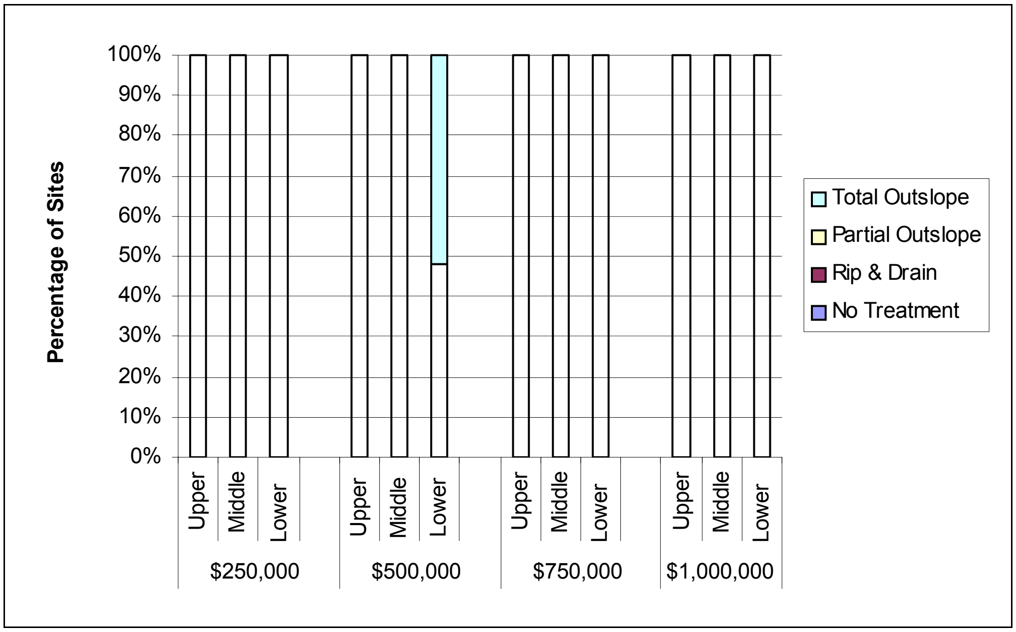


Figure 4—Comparison of dynamic programming road treatment policies with budgets of $250,000, $500,000, $750,000 and $1,000,000.

Figures 5 and *6* compare the treatment polices developed by the GA and the DP models given a $500K budget for crossings and roads respectively.

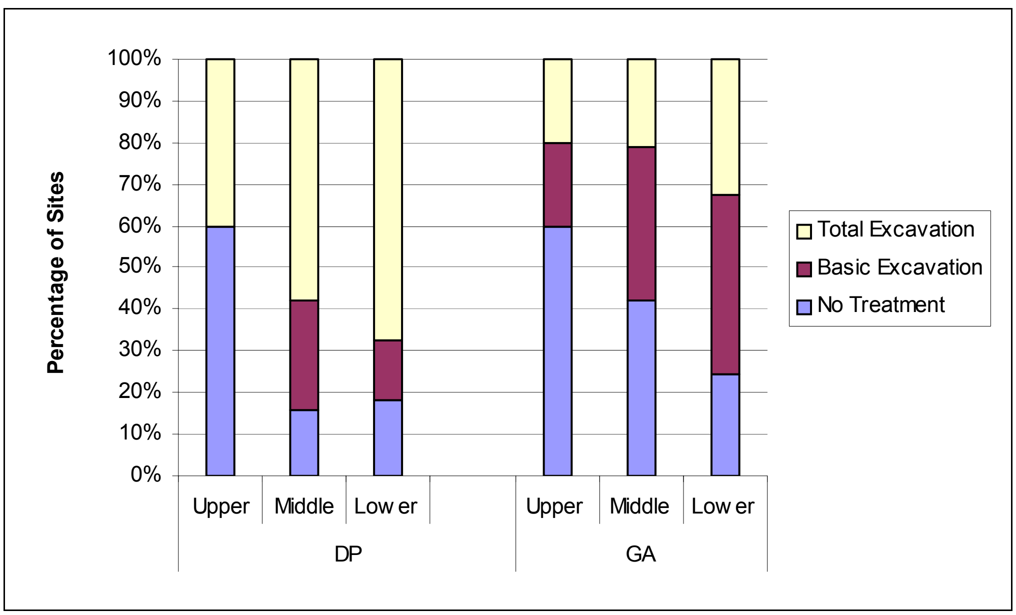


Figure 5—Comparison of DP to GA treatment policies for crossings with a $500,000 budget.

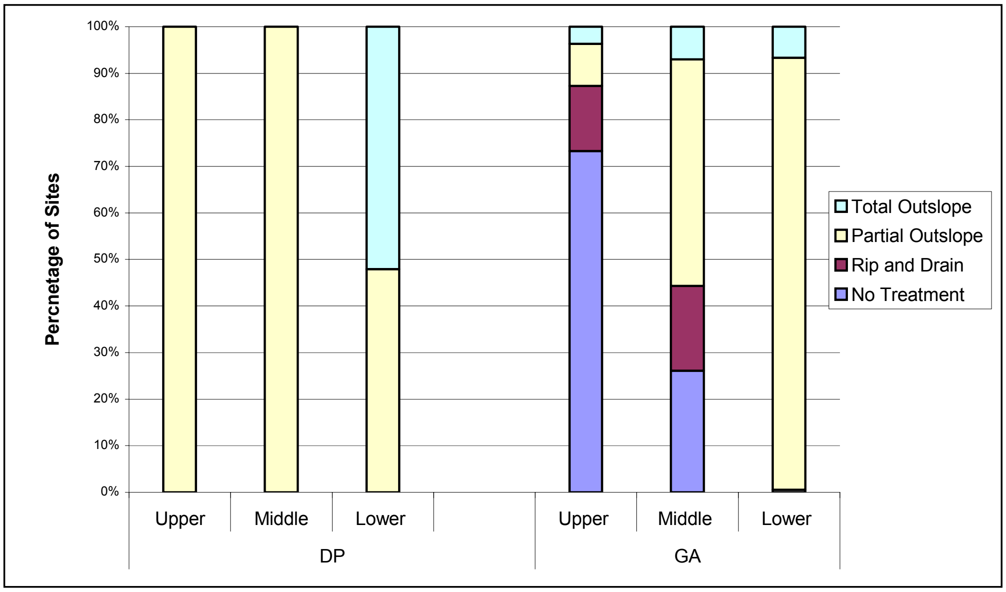


Figure 6—Comparison of DP to GA treatment policies roads with a $500,000 budget.

Discussion

Table 4 results demonstrate that the dynamic program policy allocates financial resources much more effectively than a minimum uniform policy in Lost Man Creek Basin. The DP policy spends almost $100,000 less while saving almost 10,000 yd^3 more sediment than the Uniform Minimum policy. Another measure of the effectiveness of these policies is via the reported cost/benefit ratio.

Table 5 results demonstrate that the GA is obtaining a result that is close to optimal. The summaries of the DP and GA polices are similar *(table 4)*. (As described earlier, no global optimum is guaranteed with GAs while DP results reflect a global optimum.)

Figures 3 and *4* demonstrate that the DP optimization results are rational. As the budget increases, the DP policies include more expensive treatments. In general, the more expensive treatments are used first in the lower basin roads and crossings, as these areas have the best cost/benefit ratio as presented in *tables 2* and *3*.

Figures 5 and *6* compare the GA and DP generated policies for a $500K budget and indicate that the DP policies may be easier to implement as they have less variation. While the total cost and sediment saved for these two policies are similar *(table 5)*, the distribution of the actual treatment types is different for DP and GA policies *(figs. 5* and *6)*. The DP policies have less variation. For example, in *figure 6*, the DP policy only indicates two types of treatments for roads, while the GA indicates four treatment types. This larger variation in treatment types reflects the randomly generated solutions in the GA approach.

Conclusions

Unpaved forest roads can cause erosion and downstream sedimentation in anadromous fish-bearing streams. Although road decommissioning and road upgrading activities have been conducted on many of these roads, these activities have usually been implemented and evaluated on a site-specific basis without the benefit of a watershed perspective. Land managers still struggle with designing the most effective road treatment plan to minimize erosion while keeping costs reasonable across a large land base. We suggest an approach to develop the most cost-effective strategy to treat roads based on field road inventories of erosion potential from roads. The approach can be adapted as more data on erosion and restoration effectiveness become available. In the redwood region, more land managers are using a watershed assessment approach which includes detailed road inventories. Future efforts will also consider the impacts of short-term erosion which occurs immediately following restoration work.

References

Bellman, R.E. 1957. **Dynamic Programming.** Princeton, NJ: Princeton University Press.

Best, D.W.; Kelsey, H.M.; Hagans, D.K.; Alpert, M. 1995. **Role of fluvial hillslope erosion and road construction in the sediment budget of Garrett Creek, Humboldt County, California.** In: Nolan, K.M.; Kelsey, H.M.; Marron, D.C., eds. Geomorphic Processes and Aquatic Habitat in the Redwood Creek Basin, Northwestern California. US Geological Survey Professional Paper 1454: Chapter M.

Goldberg, D.E. 1989. **Genetic Algorithms in Search, Optimization and Machine Learning.** Reading, MA: Addison-Wesly Publishing Company.

Holland, J. 1992 **Adaptation in Natural and Artificial Systems: An Introductory Analysis with Applications to Biology, Control, and Artificial Intelligence**. MIT Press.

Janda, R.J.; Nolan, K.Mm.; Harden, D.R.; Colman, S.M. 1975. **Watershed conditions in the drainage basin of Redwood Creek, Humboldt County, California as of 1973.** Open-File Report 75-568. US Geological Survey: Menlo Park, CA.

Madej, M. 2001. Erosion **and Sediment Delivery Following Removal of Forest Roads.** Earth Surface Processes and Landforms 26:175-190.

Megahan, W.F.; Kidd, W.J. 1972. **Effects of logging and logging roads on erosion and sediment deposition from steep terrain.** Journal of Forestry 7:136–141.

Tomberlin, D.; Baxter, W.T.; Ziemer, R.R.; Thompson, M. 2002. Logging Roads and Aquatic Habitat Protection in the California Redwoods Poster, 2002 SAF National Convention, 5-9 October 2002, Winston-Salem, NC. *Available at* http://www.fs.fed.us/psw/rsl/projects/water/Tomberlin.pdf

Mapping Prehistoric, Historic, and Channel Sediment Distribution, South Fork Noyo River: A Tool For Understanding Sources, Storage, and Transport[1]

Rich D. Koehler,[2] Keith I. Kelson,[3] Graham Matthews,[4] K.H. Kang,[5] and Andrew D. Barron[3]

Abstract

The South Fork Noyo River (SFNR) watershed in coastal northern California contains large volumes of historic sediment that were delivered to channels in response to past logging operations. This sediment presently is stored beneath historic terraces and in present-day channels. We conducted geomorphic mapping on the SFNR valley floor to assess the volume and location of sediment associated with pre-historic terraces, historic terraces, and the active channel along four 1-mi-long stream reaches. Additionally, we established ten streamflow and suspended sediment sampling locations to monitor water and sediment discharges. We estimate 158,000 yds^3 of sediment stored in the active channel, and 68,000 yds^3 of sediment stored beneath historic terraces. These volumes are an order of magnitude less than the volumes estimated for pre-historic terraces. The present-day channel sediment is stored presently in large gravel bars and is mobilized primarily during winter flood events.

Based on channel mapping and hydrologic data, we infer that the largest suspended sediment loads are spatially coincident with the locations of the greatest amounts of stored channel sediment. Re-mobilized historic sediment appears to increase suspended sediment load, and may be a significant, previously unrecognized sediment source. Thus, accurately mapping and quantifying channel deposits is a critical step for assessing sediment budgets, especially in Total Maximum Daily Load (TMDL) studies attempting to relate upslope management to suspended sediment production.

Introduction

The South Fork Noyo River is a major tributary of the Noyo River, which drains to the Pacific Ocean at the town of Fort Bragg in coastal Mendocino County, California *(fig. 1)*. The watershed has been heavily impacted by widespread clearcut logging over the last century. As a consequence, large volumes of sediment have been delivered to watercourses within the basin. Management practices conducted following the 1973 Forest Practice Act have contributed to a decrease in the rate of sediment delivery, although, large volumes of sediment continue to affect the ecology of the watershed (USEPA 1999). Historically, large populations of coho salmon and steelhead reproduced in the river (Brown and others 1994). Drastically declining fish

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] William Lettis & Associates, Inc, 999 Andersen Dr., Suite 120, San Rafael, CA 94901. email: koehler@seismo.unr.edu

[3] William Lettis & Associates, 1777 Botelho Dr., Suite 262, Walnut Creek, CA 94596.

[4] Graham Matthews & Associates, P.O. Box 1516, Weaverville, CA 96093.

[5] National Central Univ., ChungLi, Taiwan.

populations over the past several decades (CDFG 1995a, 1995b) have raised concerns over the cumulative impacts of sediment on water quality, fish habitat, and the aquatic environment.

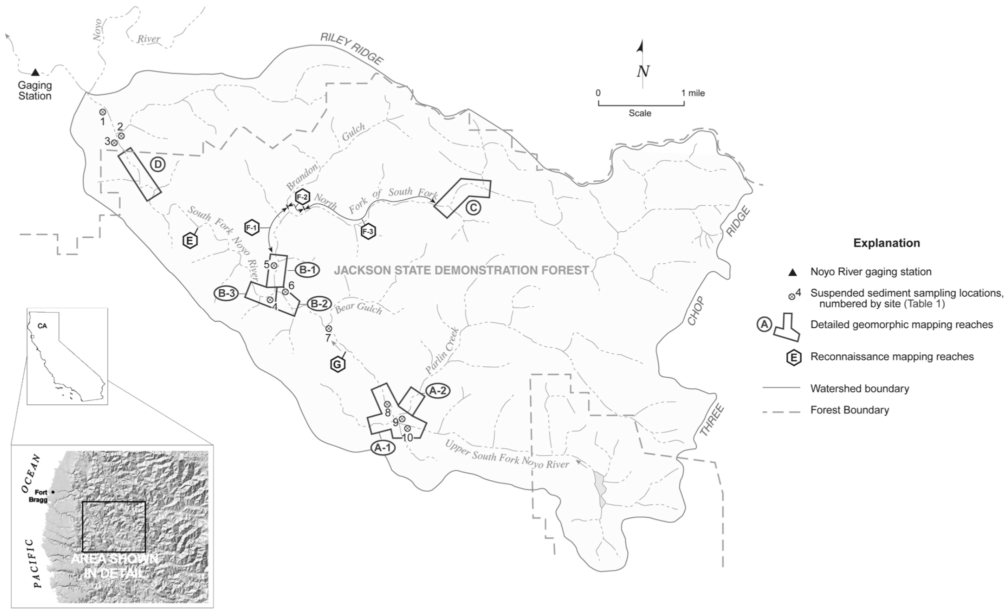


Figure 1—Drainage map of the South Fork Noyo River watershed showing detailed geomorphic mapping locations, reconnaissance mapping reaches, suspended sediment sampling locations, cross section locations, watershed boundary, and property boundary of Jackson State Demonstration Forest.

In response to these concerns, the Noyo River watershed was listed as a sediment impaired waterbody and included in the 1998 Section 303(d) list adopted by the State of California North Coast Regional Water Quality Control Board. In 1999, the Environmental Protection Agency (EPA) established the Noyo River TMDL for sediment, and identified sediment loading allocations aimed at improving water quality criteria for sediment. Accurately determining sediment loads for large watersheds is non-trivial, and office-based estimates are often associated with large uncertainties. The EPA acknowledges that large uncertainties in sediment input/storage estimates may be due to incompatibilities between field and office-based analyses (USEPA 1999). We believe that field-based sediment storage estimates are needed to improve office-based estimates. Thus, quantifying reasonable ranges of sediment input from, and storage in, these watershed sources is critical to understanding the sediment transport processes within the SFNR watershed, and to evaluating the long-term impacts of sediment transport within the SFNR ecological system.

The primary objectives of this assessment are to: (1) collect basic data on volumes of sediment stored and transported within the SFNR watershed over the past approximately 110 years, and (2) collect present-day stream flow and sediment transport data from the main stem SFNR and its major tributaries. These data provide information on long- and short-term storage and transport within the SFNR watershed and illustrate the importance of field-based information in sediment budget analyses.

Approach and Methods

We assessed the historic and current influences on channel morphology by conducting detailed geomorphic field mapping along four stream reaches (Areas A, B, C and D; *fig. 1*). Within these study reaches, we developed detailed geomorphic maps of current channel conditions showing the locations of fluvial terrace, gravel bar, and channel deposits. For field mapping, a string line painted at 25 foot intervals was tied tight along a straight line of sight in the channel thalweg. The compass bearing of the string line was plotted on the field map and tape and compass methods were used to map the dimensions of geomorphic units.

The field maps were converted into a Geographic Information System (GIS) format and used to calculate the area of all of the mapped deposits. These data were combined with field observations of deposit thickness to estimate the sediment volume for each deposit. Cumulative terrace and channel storage volume for each stream reach was calculated as a sum of individual terrace and channel deposits. Sediment thickness is the largest source of error in estimating storage volume.

Individual terrace and gravel bar deposit thickness was assumed to be the distance from the deepest scour in the active channel to the top of the surface. Field evidence used to determine the minimum thickness of channel storage included the depth of scour pools, depth measured at the downstream side of debris dams, the diameter of logs partially buried in the channel, and where available, the surface of bedrock. We infer that the estimates of the sediment volume associated with channel deposits represent minimum reasonable values. Additionally, because information usually is not available on the depth to bedrock beneath gravel bar or historic terrace deposits, we estimate thickness for these deposits as the sum of the sediment thickness estimated in the channel and the height of the respective surface. Because of this, estimates of the sediment volume associated with gravel bars and historic terraces combined represent minimum storage values associated with the active channel. In addition, sediment volume was quantified similarly in channel reaches between the detailed stream reaches (Areas E, F, and G; *fig. 1*), with the exception that surface area was estimated using pace measuring techniques.

To assess present-day hydrology and sediment transport within the major sub-watershed areas in the SFNR watershed, we established ten streamflow and suspended sediment sampling locations (numbered 1 to 10, *fig. 1*) and monitored these stations through WY2001. A standard staff plate and fence posts driven into the streambed were used to measure stream flow stage. Continuous stage recorders with pressure transducers were installed at four locations (Sites 1, 5, 9, and 10; *fig. 1*). Stream flow measurements were taken at all sites with a Price AA or Pygmy current meter and an AquaCalc 5000-Advanced Stream Flow Computer. Depth-integrated turbidity and suspended sediment sampling was performed at most locations. At locations where it was not possible to get a true depth-integrated sample, grab samples or modified depth-integrated samples were taken. The streamflow and sediment data were used to develop relations between stage, discharge, suspended sediment load, suspended sediment concentration, turbidity, and suspended sediment load per unit area (Koehler and others 2001). Total suspended sediment loads calculated for each sampling station were used to compare sediment loads between sub-watershed basins and to assess present-day sediment transport through the watershed.

Results

Locations and Amounts of Stored Sediment

Within each study area along the SFNR, we identified three distinct geologic map units, including deposits associated with pre-historic terraces, historic terraces, and the active channel *(fig. 2)*. Pre-historic terraces were identified by the presence of old-growth redwood stumps in growth position on the terrace surface. This map unit approximates the terrace configuration in the SFNR watershed prior to logging initiation in the late 1800s. Bedrock strath exposures along the channel margin indicate that the terraces are associated with three to eight feet of sediment, which probably is in permanent storage on the basis of deep incision (five to 20 feet). Historic terraces were delineated based on the presence of chainsawed logs within terrace deposits and an absence of old-growth stumps. Based on abundant logging debris only in historic terrace deposits, we infer that the historic terraces represent the maximum amount of channel aggradation that has occurred since the initiation of logging. Historic terraces are most common near the confluence of major tributaries *(fig. 1)*. The deposits associated with these terraces are approximately three to six feet thick. The terraces are a relatively constant height along the stream profile and are inset into pre-historic terraces and bedrock. Sediment stored in historic terrace deposits is subject to bank erosion but is trapped primarily in long-term storage.

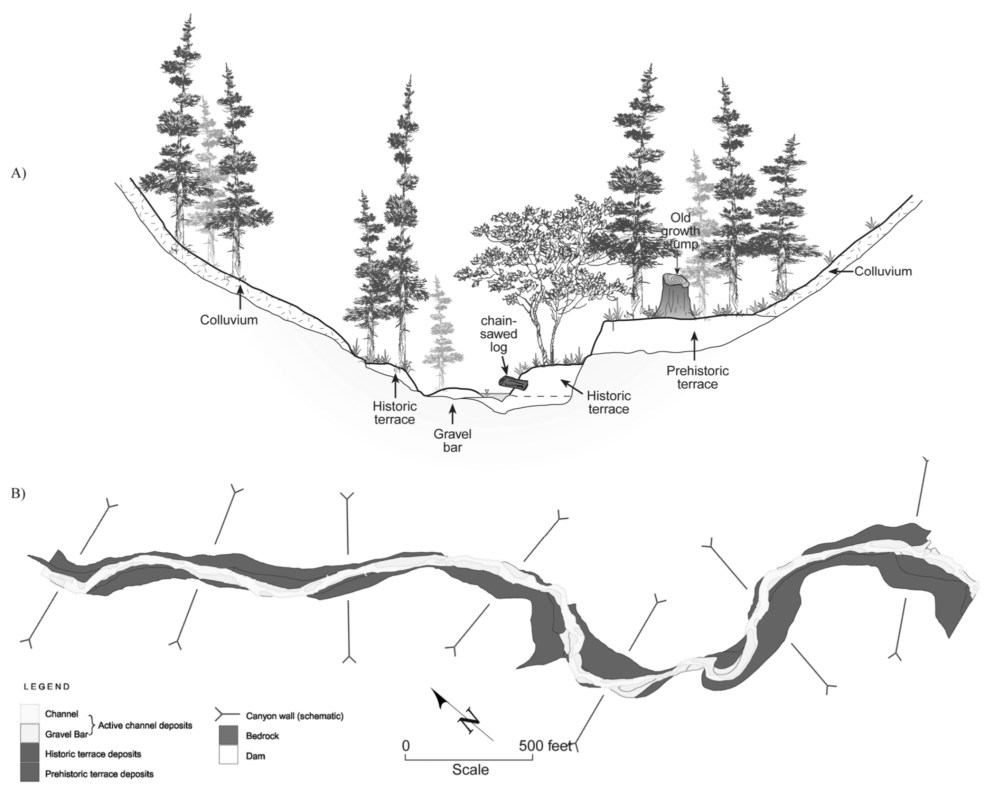


Figure 2—A) Schematic sketch of typical South Fork Noyo River channel showing valley margin, prehistoric terrace, historic terraces, gravel bar, and channel. Historic terrace deposits are observed on bedrock in some locations (left) and on channel deposits in other locations (right). Old growth redwood stumps are diagnostic of prehistoric deposits and embedded chain-sawed logs are diagnostic of historic deposits. Prehistoric terraces typically support second-growth redwood trees and ferns, historic terraces typically support alder trees and grasses. B) Detailed geologic map of mapping area D.

Active channel deposits exist throughout the study area, but are more extensive in downstream locations (Areas B, D, and E on *fig. 1*). These deposits are composed of both gravel bar and channel deposits. Channel deposits are submerged by the river throughout the year and range in thickness from approximately 0.5 to four feet, with occasional pockets as deep as 10 feet. Gravel bar deposits are submerged only during storm events, and range in thickness from approximately 0.5 to three feet. Based on the presence of chainsawed logs buried in the channel, we infer that the active channel deposits post-date the initiation of logging in the SFNR and represent transport of historic sediment.

Table 1 summarizes the total volume of each type of deposit within the detailed and reconnaissance mapping areas. Because individual mapping areas are different sizes, the total volume associated with each deposit in each stream reach is averaged over river distance for comparative purposes *(table 2)*. *Figure 3* shows active channel storage and historic terrace storage volumes for each stream reach. Map Areas A, F, and G have similar active channel storage (less than 13,300 yds^3/mile), whereas Areas B, C, D, and E have active channel storage of more than 20,000 yds^3/mile. Historic terrace sediment distribution is similar for areas D, E, F, and G (less than 5,000 yds^3/mile), however areas A, B, and C have considerably more stored historic terrace sediment *(table 3* and *fig. 3)*. Overall, the volume of sediment stored in the active channel is much more than the volume of the historic terrace deposits, with the exception of Area A. These data show that a large amount of the sediment in the SFNR watershed is stored along the main channel downstream of the North Fork of the SFNR. From these relations, we infer that there has been sufficient time since the logging operations and subsequent terrace deposition to erode the historic terrace deposits and redistribute this material downstream. We also infer that the combined volume of sediment stored in the active channel and historic terrace locations represents the minimum amount of material introduced to the South Fork Noyo river system by logging operations.

Table 3 shows the total post-logging sediment (in other words, active channel and historic terrace) remaining in the SFNR study area. The total post-logging sediment volume in storage over the entire study area is estimated at 225,000 yds^3 or approximately 22,000 yds^3/mile *(table 3)*. Areas F and G, which contain the least post-logging sediment, are located directly upstream of the confluence of the SFNR and the North Fork of the SFNR, and have bedrock exposed along much of their distance. The scarcity of historic terrace remnants and the low volume of active channel sediment within Areas F and G imply that much of the post-logging sediment has been transported downstream. This relationship may be related to the narrow confined valley (between pre-historic terraces) in Areas F and G and the comparatively wider valleys in Areas B, D, and E. Alternatively, the low sediment storage in Areas F and G may be related to a lesser amount of debris left by past logging operations. Notably, areas directly downstream from Areas F and G (in other words, Areas C and A, respectively) have considerably more post-logging sediment in storage than the stream reaches located directly upstream (Areas G and F, respectively). This probably is related to a wider channel in Area C, and a channel confluence in Area A.

Present-Day Hydrology WY2001

Streamflow measurements and sediment transport data included most of the significant storm events in WY2001, although few large storms provided relatively few opportunities to collect high-flow discharge measurements and sediment samples. One hundred and fifteen sediment transport measurements were made at the 10 sampling stations in WY2001 (Koehler and others 2001). *Table 4* shows the total suspended sediment load (tons) and the unit rate (tons/mi^2) for each sampling station, and *figure 3* shows the suspended sediment load distribution. Suspended sediment loads computed for each sampling station ranged from 14 tons at the mouth of Bear Gulch (Site 7, *fig. 1*) to 685 tons on the SFNR at the downstream end of the study area (Site 1, *fig. 1*). Total suspended sediment load increased downstream as the drainage area increased from Bear Gulch (Site 7) through the SFNR at Site 4. However, between this site and the mouth of Kass Creek suspended sediment load dramatically increased from 12.5 to 25.4 tons/mi^2, suggesting that a readily mobilized source of sediment exists within this stream reach. We infer that the source for this sediment is the large volume of sediment stored in the active channel that is remobilized during storm events.

Table 1—*Total volume of sediment stored in active channel deposits, historic terrace deposits, and pre-historic terrace deposits for each detailed mapping area (Area A-1 to Area D) and reconnaissance mapping area (Area E to Area G).*

Stream reach	River dist. (miles)	Active channel deposits (yds^3) *			Historic terrace deposits (yds^3) *	Pre-historic terrace deposits (yds^3) * ≠
		Gravel bar deposits (yds^3) *	Channel deposits (yds^3) Ø	Total active channel deposits (yds^3) *		
Area A-1	1	3,900	5,400	9,300	19,200	199,400
Area A-2	0.3	500	700	1,200	1,300	N.D. ∞
Area B-1	0.5	5,500	4,400	9,900	4,500	68,300
Area B-2	0.4	5,400	3,300	8,700	3,200	82,300
Area B-3	0.4	5,700	4,400	10,100	4,300	34,100
Area C	0.8	9,700	7,100	16,800	10,100	26,100
Area D	0.8	9,500	7,200	16,700	2,700	44,500
Area E	2.2	29,500	26,700	56,200	7,000	3,316,300
Area F-1	0.4	1,600	2,000	3,600	1,800	22,100
Area F-2	0.3	100	600	700	0	3,700
Area F-3	1.9	8,300	4,600	12,900	6,200	93,500
Area G	1.5	4,500	7,000	11,500	7,600	65,900
All areas	**10.27**	**84,200**	**73,400**	**157,600**	**67,900**	**3,956,200**

* Reported values represent minimum potential storage volume due to uncertainties in terrace thickness at the back edge of the deposit.

Ø Reported values represent minimum storage volume.

≠ Pre-historic terrace sediment volumes are based on an assumed 5 foot thickness except for Area A which is calculated based on 4 foot thickness determined from field observation. (Range of depth error is +/- 3 feet).

∞ N.D.; no data. Prehistoric terrace volume for Area A-2 is included in the volume calculated for A-1.

Table 2—*Sediment storage in active channel deposits, historic terrace deposits, and pre-historic terrace deposits averaged per river mile for each detailed mapping area (Area A-1 to Area D) and each reconnaissance mapping area (Area E to Area G).*

Stream reach	River dist. (miles)	Active channel deposits (yds^3/mile)*			Historic terrace deposits (yds^3/mile)*	Pre-historic terrace deposits (yds^3/mile)*
		Gravel bar storage (yds^3/mile)*	Summer channel storage (yds^3/mile)^Ø	Total active channel storage (yds^3/mile)*		
Area A-1	1	3,900	5,400	9,300	19,200	199,400
Area A-2	0.3	1,600	2,300	4,000	4,300	N.D.$^\infty$
Area B-1	0.5	11,000	8,800	19,800	9,000	136,600
Area B-2	0.4	13,500	8,300	21,800	8,000	205,800
Area B-3	0.4	14,300	11,000	25,300	10,800	85,300
Area C	0.8	12,100	8,900	21,000	12,700	32,600
Area D	0.8	11,900	9,000	20,900	3,400	55,600
Area E	2.2	13,400	12,100	25,500	3,200	1,507,400
Area F-1	0.4	4,000	5,000	9,000	4,500	55,300
Area F-2	0.3	300	2,000	2,300	0	12,300
Area F-3	1.9	4,400	2,400	6,800	3,300	49,200
Area G	1.5	3,000	4,700	7,700	5,100	43,900
All Areas	**10.3**	**8,200**	**7,100**	**15,300**	**6,600**	**384,100**

* Reported values represent minimum potential storage volume due to uncertainties in terrace depth at the back edge of deposit.

Ø Reported values represent minimum storage volume.

∞ N.D.; no data, pre-historic terrace volume for Area A-2 is included in the volume calculated for A-1.

Table 3—*Total amount of post-logging sediment remaining in the South Fork Noyo River and tributaries by stream reach. The values represent the sum of sediment stored in the active channel and historic terrace deposits.*

Stream reach	River distance (miles)	Total volume of post-logging sediment (yds^3)*	Total volume of post-logging sediment averaged for river distance (yds^3/mi)*
Area A-1	1	28,500	28,500
Area A-2	0.3	2,500	8,300
Area B-1	0.5	14,400	28,800
Area B-2	0.4	11,900	29,800
Area B-3	0.4	14,400	36,000
Area C	0.8	26,900	33,600
Area D	0.8	19,400	24,200
Area E	2.2	63,200	28,700
Area F-1	0.4	5,400	13,500
Area F-2	0.3	700	2,300
Area F-3	1.9	19,100	10,100
Area G	1.5	19,100	12,700
All Areas	**10.3**	**225,500**	**21,900**

*Reported values represent minimum potential storage volume.

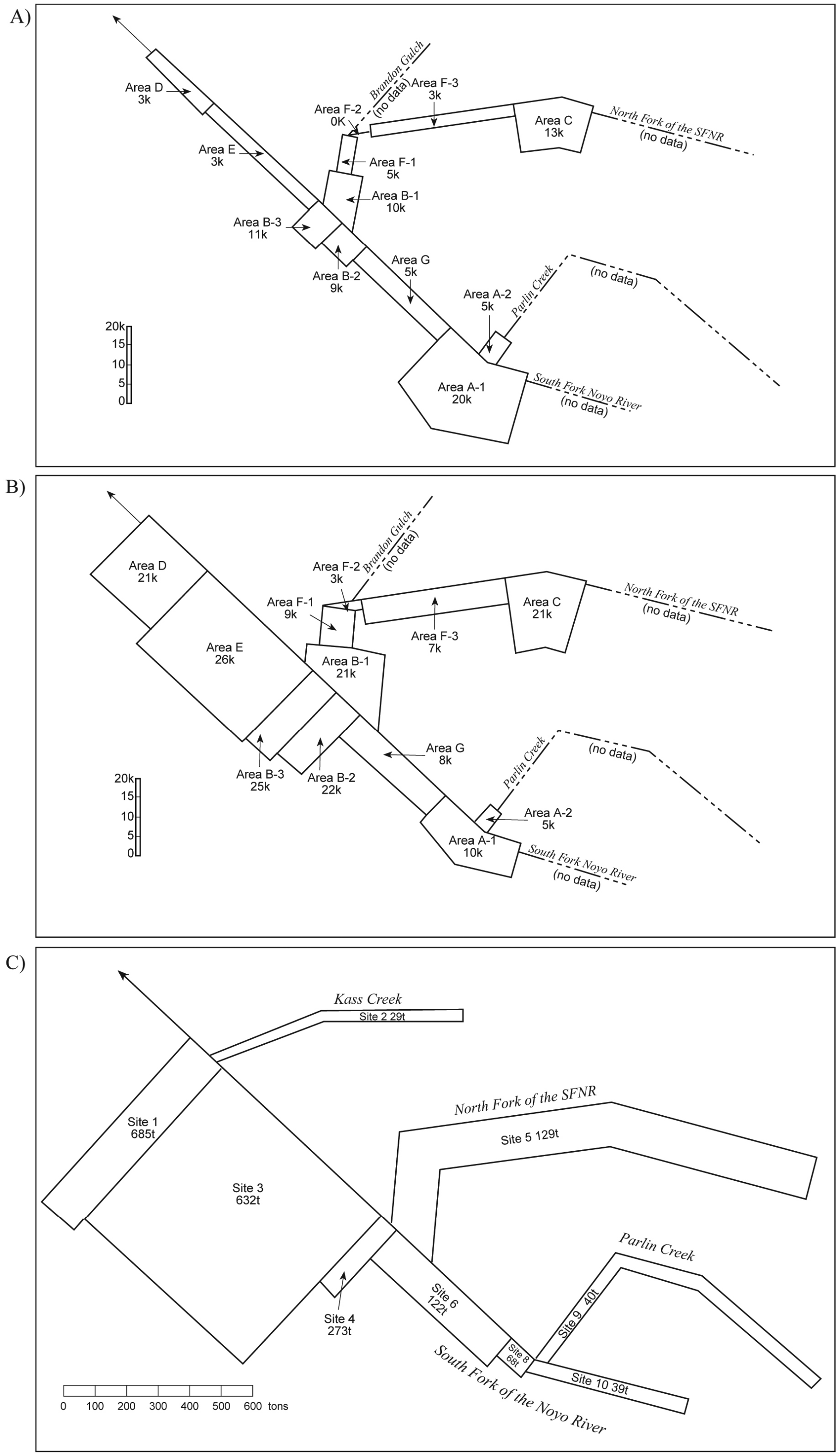


Figure 3—Schematic box diagrams of the South Fork Noyo River showing; A) total volume of historic terrace storage in yds^3/mile, B) total volume of active channel deposits in yds^3/mile, and C) total suspended sediment for each sampling station in tons.

Table 4—*WY 2001 total suspended sediment load (SSL) in tons and tons per square mile for each sampling station.*

Station Number	Area (mi^2)	SSL (tons)	Unit SSL (tons/mi^2)
1	27	685	25
2	2.21	29	13
3	25	632	25
4	22	273	13
5	9.9	129	13
6	12	122	10
7	1	14	13
8	9.2	68	7
9	4.4	40	9
10	3.7	39	11

Discussion

Our detailed channel mapping identified 158,000 yds^3 of sediment stored in the active channel and 68,000 yds^3 of sediment stored in historic terraces *(table 2)*. This sediment likely is mobilized during winter storm flows. The greatest amount of active channel storage occurs between Kass Creek and the mouth of the North Fork SFNR (Areas B, D, and E). In contrast to upstream areas, suspended sediment measured in this area showed a dramatic increase in the volume of sediment produced, where approximately 360 tons of suspended sediment were delivered from only 2.9 mi^2. Thus, the greatest amount of stored channel sediment is spatially coincident with the location of the largest amount of suspended sediment load *(fig. 3)*.

The source for this suspended sediment is most likely sediment stored in the active channel that is re-mobilized during storm events, rather than eroded from historic terrace deposits. The volume of sediment stored in historic terraces along this reach (Areas B, D, and E; *figs. 1* and *3*) is less than along reaches upstream, suggesting that suspended sediment eroded from historic terraces by bank erosion is a minor component of the total suspended sediment load. We interpret that other possible sources of suspended sediment load (in other words, landslides, road erosion) are minor contributors, based on scarcity of slides along the channel margin and adjacent side slopes, and the consistent road density in the area. Thus, land management practices probably do not cause the relatively high suspended sediment load in Areas B, D, and E.

Short-term sediment budgets, evaluated over decadal time scales, generally rely on the assessment of sediment inputs determined from inspection of multiple sets of aerial photographs and limited field observation. The office-based sediment budget approach for the Noyo River TMDL, which included the SFNR, states that fluvial-induced alluvial storage change is a relatively minor term in the overall sediment budget (USEPA 1999). However, the TMDL notes that the discrepancy between inputs and outputs in the Noyo River watershed may be a result of sediment input volume errors or time lags from sediment delivery to transport through the system. In contrast to previous assumptions, our sediment storage and transport study shows that the amount of sediment stored in the SFNR for various lengths of time has a major influence on the assessment of the present-day sediment transport and the short-term sediment budget.

The addition of suspended sediment eroded from active channel deposits to watercourses appears to result in a dramatic increase in the overall suspended sediment load. Areas that contain large amounts of sediment stored in active channels likely are large contributors to the suspended sediment measured during present-day high-discharge events. Therefore, this research has demonstrated that the amount of sediment in long-term storage is a significant contributor to short-term suspended sediment load. Clearly, a distinction must be made between the amount of sediment introduced to the system over the short-term and the amount of sediment re-introduced to the system from long-term channel storage locations.

Future field-based sediment budget analyses for watersheds in the North Coast will benefit from accurate mapping and quantification of channel deposits. An understanding of the volume and timing of sediment stored in the channel is necessary to relate upstream management practices to suspended sediment production and to evaluate cumulative effects. By not addressing long-term sediment storage and relying solely on present-day suspended sediment sampling, suspended sediment load entering the watercourse by modern management practices can be substantially over-estimated.

Conclusions

We assessed the volume of past and present sedimentation within the SFNR by quantifying the volume associated with pre-historic terraces, historic terraces, and the active channel in four detailed mapping reaches and three reconnaissance surveys. Additionally, we assessed present day streamflow and sediment transport throughout the SFNR watershed by establishing and monitoring a stream gauge network for WY 2001.

Total post-logging sediment volume (active channel and historic terrace) in storage over the entire study area is estimated at 225,000 yds^3 or approximately 22,000 yds^3/mile. Comparison of the volume associated with historic terraces and the volume associated with the active channel indicates that a large portion of the sediment originally deposited in historic terraces has been eroded and transported downstream. A significant portion of this sediment presently is stored in the lower SFNR channel between its confluence with the North Fork of the SFNR and the mouth of the SFNR.

Suspended sediment loads computed for each sampling station ranged from 14 to 685 tons. Overall, most sites produced sediment at a fairly consistent rate with discharge, although a large increase in sediment transport occurred between the mouth of the North Fork SFNR (Site 4) and the site upstream of Kass Creek (Site 3). This implies that significant sources of readily accessible sediment are located in this reach. This readily accessible sediment is most likely the active channel sediment stored in Areas B, D,and E.

The detailed maps and hydrologic data produced in this research provide a snap-shot of the distribution of stored sediment and present day sediment transport within SFNR. These data represent a baseline datum from which to monitor future channel recovery and assess the effects of upslope management practices. This research suggests that past logging practices contributed many thousands of cubic yards of sediment to channels in the SFNR watershed, and that the river has the ability to transport this material downstream. This research also demonstrates the need for an understanding of in-channel sediment storage and transport for relating upslope forest management practices to suspended sediment load.

Acknowledgments

This research was supported by the State Forest Research and Demonstration Program (RFP # 8CA99038) of the California Department of Forestry and Fire Protection (CDF). We thank Tim Robards (CDF-Sacramento) and Bill Baxter (CDF-Fort Bragg) for logistical support throughout the project.

References

Brown, L.R.; Moyle, P.B.; Yoshiyama, R.M. 1994. **Historical decline and current status of coho salmon in California.** North American Journal of Fisheries Management (4)2: 237-261.

California Department of Fish and Game (CDFG). 1995a. **Stream Inventory Report: South Fork Noyo River, Northern California and North Coast Region.** Fortuna, California.

California Department of Fish and Game (CDFG). 1995b. **Stream Inventory Report: Parlin Creek, Northern California and North Coast Region.** Fortuna, California. 35 p.

Environmental Protection Agency (USEPA). 1999. **Noyo River Total Maximum Daily Load for Sediment: Region IX Water Division.** San Francisco, California. 82 p.

Koehler, R.D.; Kelson, K.I.; Matthews, G.; Kang, K.H.; Barron, A.D. 2001. **Sediment storage and transport in the South Fork Noyo River watershed, Jackson State Demonstration Forest: Final Technical Report to California Department of Forestry and Fire Protection.** Sacramento, California. Contract # 8CA99253. 80 p.

The Significance of Suspended Organic Sediments to Turbidity, Sediment Flux, and Fish-Feeding Behavior[1]

Mary Ann Madej,[2] Margaret Wilzbach,[3] Kenneth Cummins,[4] Colleen Ellis,[2] and Samantha Hadden[2]

For over three decades, geologists, hydrologists and stream ecologists have shown significant interest in suspended load in running waters. Physical scientists have focused on turbidity, the development of sediment-rating curves and estimation of sediment yields, often as an indicator of changing land uses (Beschta 1981). Stream ecologists, on the other hand, have focused on 1) the role of suspended sediments in water quality degradation and its deleterious impacts on biological communities (for example, Waters 1995); or 2) its beneficial roles in providing food resources for filter-feeding invertebrates and as the major pathway of organic matter transport and export, linking upstream and downstream reaches and affecting such ecosystem processes as nutrient spiraling (Minshall and others 1983, 1985; Wallace and Grubaugh 1996). The focus of these interests has dictated the way in which sediment samples are examined. In many cases, the organics in suspended load samples are removed by ashing or chemical digestion. But physical scientists and stream ecologists concerned with the deleterious role of suspended sediments tend to discard data on the organic fraction (ash-free or carbon digested), while ecologists interested in its beneficial role discard information on the mineral fraction (ash or digestion residue).

Failure to distinguish between organic and inorganic components of the suspended load or to consider the full suite of information present in suspended sediment samples has hindered full understanding of sediment dynamics as it affects stream health and reflects watershed condition. The present study establishes the contribution of size-specific, inorganic and organic components to turbidity and sediment flux in streams in the redwood region (in Redwood National and State Parks and Jackson State Forest). We investigated the influence of organic suspended load on the feeding efficiency and condition of juvenile salmonids and their invertebrate food base through field observations and flume experiments *(fig. 1)*.

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] USGS Western Ecological Research Center, 1655 Heindon Rd., Arcata, CA 95521. email: mary_ann_madej@usgs.gov
[3] USGS California Cooperative Fishery Research Unit, Humboldt State University, Arcata CA. email: paw7002@humboldt.edu
[4] Humboldt State University Institute for River Ecology, Arcata, CA. email: kwc7002@humboldt.edu

Figure 1—A biologist makes underwater observations of fish feeding behavior during low-turbidity conditions in northern coastal California. Photograph by Samantha Hadden.

Suspended sediment was sampled in several watersheds for two years. Watershed characteristics are listed in *table 1.* Samples were analyzed for particle sizes and organic content. Organic particles were abundant in samples with turbidity readings of up to about 60 NTU (nephelometric turbidity units), typically occurring during times of rising and falling streamflows. At higher turbidity levels and peak flows, the sediment samples had a higher proportion of inorganic materials. Organic materials tended to be more abundant in samples collected during early-season storms. Particulate organic matter constituted 10 to 65 percent of the total annual suspended sediment load for in Water Years 2002 and 2003 (Madej 2005). The influence of organics on specific streams depended on floodplain conditions and underlying geology. The percentage of organics in the suspended load varied with riparian condition, which also affects benthic macroinvertebrate communities. In previous studies of suspended sediment, researchers have tended to assume that the remaining material is all inorganic. Our results show that in four streams, the organic fraction of the sediment load is high at low-to-moderate turbidity levels and should be included in analyses of suspended-sediment concentrations.

Results of the biological phase of the study showed that the biomass of invertebrate prey sampled from the foreguts of juvenile coho salmon and steelhead trout declined with increasing turbidity. Our field observations also revealed a decline in the rate of prey capture by juvenile salmonids with increasing turbidity. These data are unique in that other studies of salmon feeding efficiency in response to turbidity or suspended-sediment concentrations have all been flume- rather than field-based. Although the efficiency of prey capture decreased at higher turbidity levels, limited fish feeding activity was still observed at the highest turbidity levels (45 NTU) in which underwater observations were made. These observations are important because many previous studies have assumed that 30 NTU is a turbidity threshold above which fish cannot feed.

Table 1—*Drainage area, gradient, riparian composition, and dominant substrate characterizing the study sites.*

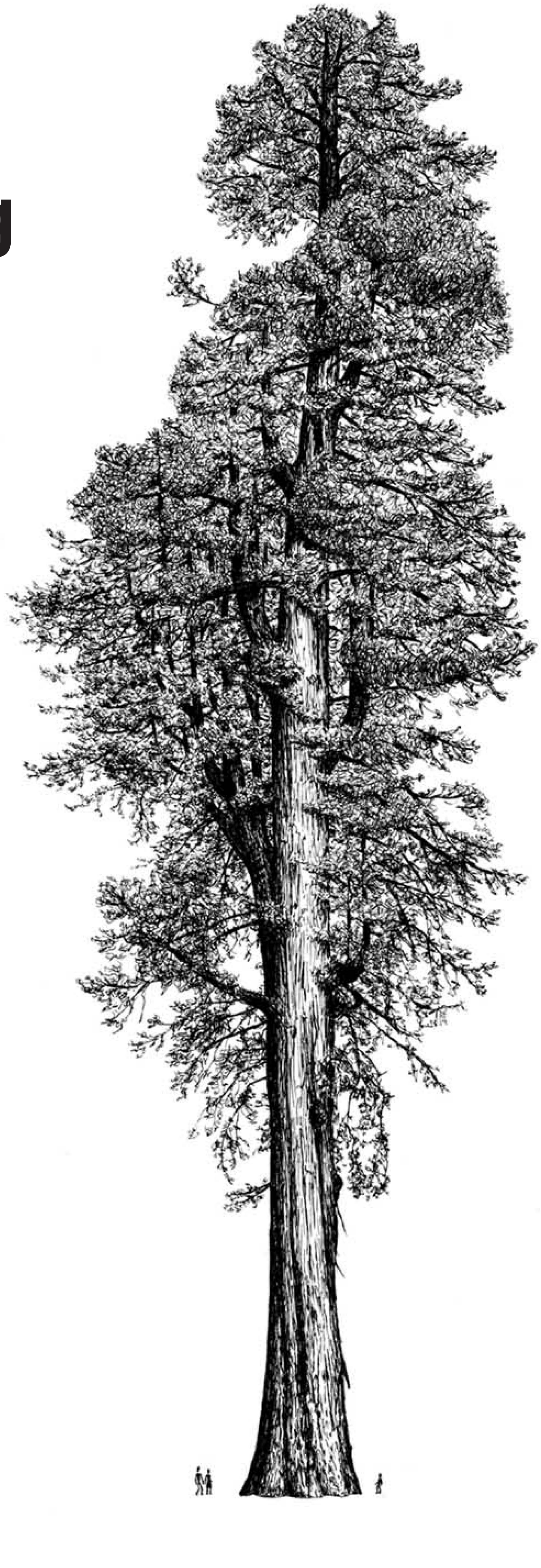
Site	North Fork Caspar Creek	South Fork Caspar Creek	Upper Prairie Creek	Little Lost Man Creek
Area (km^2)	3.8	4.2	10.5	9.9
Stream gradient (percent)	1.5	0.8	1.0	2.6
Dominant overstory riparian composition	Second-growth redwood (*Sequoia sempervirens)*	Red alder *(Alnus rubra)*	Old growth redwood *(Sequoia sempervirens)*	Old growth redwood *(Sequoia sempervirens)*
Dominant substrate	Pebble, cobble	Pebble	Pebble, sand	Cobble, pebble

Reference

Madej, Mary Ann. 2005. **The role of organic matter in sediment budgets in forested terrain.** In: Proceedings of the international symposium on sediment budgets; 2005 April 3-9; Foz do Iguaco, Brazil. IAHS Public. 292 Sediment Budgets 2. Wallingford, UK: International Association of Hydrological Sciences; 9–15.

SESSION 9

Forest Policy and Modeling



Forest Certification in the Redwood Region[1]

Sheila Helgath[2] and Mike Jani[3]

Two major forest certification schemes have emerged in the Redwood region of California: the Sustainable Forestry Initiative® (SFI) program and the Forest Stewardship Council. Both of these certification systems are influenced by the California Forest Practice Rules. This paper provides an overview of the extent certification systems are being utilized within the Redwood Region. It discusses the interaction of these programs with the California Forest Practice Rules. Following a format similar to the Meridian Report on SFI and FSC certification systems, it reviews actual forest practices and policies followed by companies using different systems of certification. Inventory planning, harvest rate, herbicide use and long term applicability, suitability of silvicultural techniques and road construction/ rehabilitation and abandonment best management practices are reviewed. The management of unique, rare and old growth forest communities identified under the different systems as forests of exceptional conservation value or forests of high conservation value is discussed. The paper goes on to discuss how scientific research activities interact with the certification programs within Redwood Region and the research needs being driven by certification standards. The perceived benefits and costs of forest certification programs for forest products companies and the public are addressed as well as how each company addresses the social interactions within the communities in which they operate. Finally, current and future issues for these certification programs in the Redwood Region are identified and discussed.

The authors of this paper are uniquely qualified to present a paper on certification programs in the Redwood Region. Mr. Jani has participated both with the FSC standard setting for the Redwood region and the Pacific Coast and also was a panel member during the development of the Meridian Report in 2001 which, compared the SFI and FSC programs as they existed then . He manages the largest FSC certified redwood company in the Redwood Region and was the first forest manager to be FSC certified within the redwood region. Dr. Helgath manages the SFI certification program and the development of an ISO 14001 registration program for one of the major SFI certified forest companies in the Redwood Region. She has also worked with FSC certification programs in South America.

---

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Manager Environmental Certification Systems PALCO, P.O. Box 37, Scotia, CA 95565-0037, (707)764-4143. email: shelgath@palco.com
[3] Chief Forester Mendocino Redwoods, Mendocino Redwood Company, LLC, 6500 Durable Mill Road, P.O. Box 390, Calpella, CA 95418, (707) 485-6751. email: mikejani@mendoco.com

Implementation of Uneven-Age Forest Management Under the Santa Cruz County/California Forest Practice Rules[1]

Douglas D. Piirto,[2] Walter R. Mark,[3] Richard P. Thompson,[4] Cheryl Yaussi,[5] Jessica Wicklander,[5] and Jesse Weaver[6]

The current, relatively even-aged second-growth forest of Santa Cruz county largely resulted from clearcutting that occurred in the late eighteen hundreds through the 1920s. Much of the lumber produced from those harvest operations between 1906 and 1920 went to rebuild San Francisco following the 1906 earthquake. The current Santa Cruz County/California Forest Practice Rules limit cutting and re-entry intervals. These rules have come to be known as the 60/50 rules; allowing cutting of up to 60 percent of the trees over 18 inches and 50 percent of the trees between 12 and 18 inches. Studies at Cal Poly's School Forest are underway to compare strict implementation of the 60/50 diameter limit rules to uneven-age management prescriptions. Continuous Forest Inventory data collected before and after harvest will be utilized in FVS, CRYPTOS and Stand Visualization system (SVS) models to compare the sustainability of both practices.

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] RPF, Professor and Department Head, Natural Resources Management Department, California Polytechnic State University, San Luis Obispo, CA 93407, (805) 756-2968. email: dpiirto@calpoly.edu

[3] RPF, Professor and Executive Faculty Director, Swanton Pacific Ranch and School Forest, Natural Resources Management Department, California Polytechnic State University, San Luis Obispo, CA 93407, (805) 756-5028. email: wmark@calpoly.edu

[4] RPF, Professor, Natural Resources Management Department, California Polytechnic State University, San Luis Obispo, CA 93407, (805) 756-2898. email: rpthomps@calpoly.edu

[5] Student, Natural Resources Management Department, California Polytechnic State University, San Luis Obispo, CA 93407.

[6] Forester , Mendocino Redwood Company.

The New Economies of the Redwood Region in the 21st Century[1]

William Stewart[2]

Abstract

The redwood region of California has experienced a number of major land use changes over the past one hundred and fifty years. A review of recent economic trends in the redwood region suggests the emergence of three new themes. First, it appears that the transition from an old growth to a young growth redwood industry is essentially complete. Lower revenues and relatively high operating costs may reduce landowner's interest to maintain large areas of expensive real estate in sustainable forest products based operations. A continued decline in the timber-based economy will probably not be compensated by a growth in economic activity associated with redwood parks. The recreational and tourism economy of the region has been, and will continue to be dominated by the coast rather than the forests. One of the most significant economic and environmental trends is the increase in rural residential land use in redwood forests. Most of the redwood forests in four areas in four separate counties now effectively have an understory of houses and associated residential land uses. The environmental impacts in terms of altering wildlife habitats and new sources of water pollutants may be greater than those typical of current best forest management practices.

Key words: economics, land use, parks, timber, tourism

Introduction

The redwood region of California stretches from the Oregon border to scattered canyons in Monterey. The region has experienced a number of major land use changes over the past one hundred and fifty years that have significantly affected regional economies. For the first one hundred and twenty-five years, the harvesting of massive inventories of old growth redwood and associated Douglas-fir dominated the economics of land use throughout the rural parts of the region and built much of the houses in the interspersed metropolitan areas. Over the past twenty-five years, an emerging young growth based timber industry, new approaches to conserving fish and wildlife habitats, additional park acquisitions, and the expansion of rural residential land use in the redwood region signal that future economic issues will not simply be limited to the 'jobs v parks' or 'jobs v owls' debates that dominated the 1980s and 1990s. The goal of this paper is to highlight a few emerging economic trends that will have considerable importance in this century.

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Assistant Deputy Director, California Department of Forestry and Fire Protection, P.O. Box 944246, Sacramento, CA 94244-2460. email: bill.stewart@fire.ca.gov

Methods

To identify potential trends that could have a significant economic impact on the redwood region in the current century, a number of spatially explicit data sets were used to illustrate patterns and trends that could have increasing importance in future decades. The main data sets used are the detailed vegetation coverage (FRAP 2002), county level economic data from the 2000 Census and the California Economic Development Department, county level redwood harvest statistics, park visitor statistics, and block level data from the 2000 Census (FRAP 2003a, FRAP 2003b, U.S. Census 2000).

Results

Redwood forests occur in twelve counties of California. *Table 1* summarizes redwood area, total area and county population for the counties in the region. The redwood forest area is based on the FRAP (2002) vegetation databases and excludes interspersed areas of Douglas fir dominated stands, tanoak, other hardwoods, other vegetation types, and urban areas within what is commonly considered to be the redwood region. More than three quarters of the redwood forests are in the three northern counties—Del Norte, Humboldt, and Mendocino—while the rest of the counties have both fewer acres of redwoods and considerably larger populations. Although some of the more central and southern counties have relatively small acreage, they also have the parks with the greatest numbers of visitors and the redwood forests with the more neighbors who value them as scenic open space.

Table 1—*Redwood area, total area, and population of counties in the Redwood Region.*

County	Redwood acres	Total acres	Population (2000)
Del Norte	42,000	649,000	27,000
Humboldt	424,000	2,293,000	127,000
Mendocino	542,000	2,248,000	86,000
Sonoma	94,000	1,015,000	458,000
Napa	500	505,000	124,000
Marin	5,000	336,000	247,000
Alameda	500	477,000	1,443,000
Contra Costa	600	475,000	948,000
San Mateo	49,000	291,000	707,000
Santa Clara	11,000	833,000	1,682,000
Santa Cruz	114,000	285,000	255,000
Monterey	15,000	2,120,000	401,000
Total	1,297,600	11,527,000	6,505,000

Sources: FRAP 2002, FRAP 2003a, US Census 2000

With economic diversification of all local economies, it is rare for any single land use based industry to dominate the county employment patterns in the manner that was common in rural California just a few decades ago. The following table compares 2000 Census data for the major employment sectors related to redwood forests for the six counties with substantial redwood to the state as a whole. Most of the jobs in the woods are combined into the broad category of Agriculture, Forestry, and Fisheries, while sawmilling jobs are classified under Manufacturing. Construction employment is a good measure of overall residential land use expansion and the recreational category captures jobs in the travel and tourism sectors. Not surprisingly, all the non-metropolitan counties (Del Norte, Humboldt and Mendocino) have a considerably higher percentage of their work force involved in the Agriculture, Forestry and Fisheries sector. The most surprising pattern from the 2000 Census is the relatively small size of the manufacturing employment in Del Norte and Humboldt Counties. This area historically had many large sawmills to match the large timber land base, but recent downsizing in timber industry employment has resulted in these counties now being far below the statewide average in terms of the relative role of manufacturing employment. The lack of relatively high wage manufacturing jobs and the greater distance from the San Francisco Bay regional economy are two reasons for the relatively low median household incomes.

Table 2—*Sectoral employment and median household incomes for California and selected counties in 2000.*

Area	**Agriculture, Forestry, Fisheries**	**Construction**	**Manufacturing**	**Recreation & Accommodations**	**Median Household Income (2000)**
	----------	----------	*percent*	----------	
California	1.9	6.2	13.1	8.2	$ 47,493
Del Norte	6.2	5.4	4.4	13.0	$ 29,642
Humboldt	4.9	5.8	8.7	9.8	$ 31,226
Mendocino	7.1	7.9	10.1	12.0	$ 35,996
Sonoma	2.6	8.5	12.7	7.9	$ 53,076
Santa Cruz	4.4	7.9	12.4	8.6	$ 53,998
San Mateo	0.4	6.2	10.3	7.4	$ 70,819

Source: US Census 2000

It is important to note that the economic data captured in the 2000 Census were a relative high water mark in the larger economic cycles. Focusing on annual data for the three counties where economic activity related to redwood forests is not overshadowed by much larger economic factors illustrates the trends over a whole decade. There was continuous improvement from 1991 to 2002 in the total number of jobs and a decline in the overall unemployment rate. Since then, there has been a slight increase in the unemployment rate.

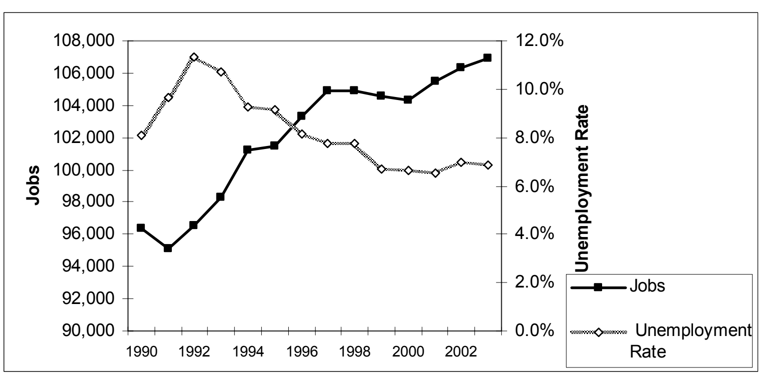


Figure 1—Jobs and Unemployment Rates for the combined Del Norte, Humboldt and Mendocino region, 1990 to 2003. Source: California Economic Development Department 2004

Current redwood harvests now are less than half of the levels seen twenty five years ago *(fig. 2)*. It is often assumed that the redwood timber harvests will occupy a progressively smaller portion of all local economies. A more detailed review of harvest levels suggests a different interpretation. *Figure 2* shows that nearly all the decline in the overall redwood harvest is due to the decline in the volume of old growth redwood harvests. Young growth harvests, on the other hand, have been relatively stable for twenty five years with most of the variation due to changes in market prices. In 1999, the Board of Equalization stopped using different harvest value schedules for old growth and young growth due to the declining volumes of old growth. The harvest declines over the past five years are partly related to lower prices compared to the high levels of 1999. The harvest declines have had noticeable impacts on employment levels within the industry.

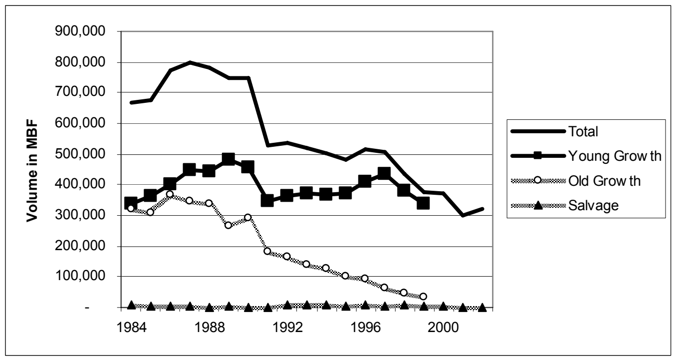


Figure 2—Total, Young, Old, and Salvage Harvest Trends for Redwood, 1984- 2002. Source: Board of Equalization 2004

Within the larger region, employment within the redwood timber industry is a substantial fraction of the county economy in Del Norte, Humboldt, and Mendocino. *Figure 3* shows job numbers for three sectors related to redwood forests from 1990 to 2003. While employment in sectors related to tourism have been relatively stable since the 1996, employment in the wood product manufacturing and natural resources sectors have declined significantly.

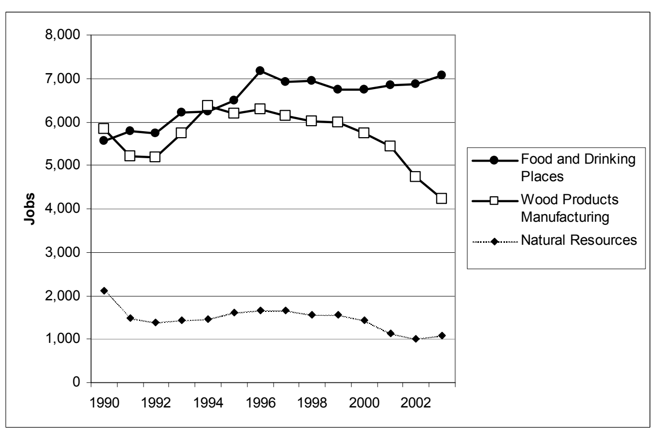


Figure 3—Forest-related employment in Del Norte, Humboldt and Mendocino counties from 1990 to 2003. Source: California Economic Development Department 2004

Since the expansion of Redwood National Park in the 1970s, increases in park visits and related tourism has been counted on to reduce the negative employment impacts of reduced timber harvests. In most cases, the increase in redwood park tourism has not matched increases in statewide tourism. The following data suggest two possible factors for the divergence between redwood-specific tourism and overall tourism. *Figure 4* shows use trends for 16 State beach parks and 13 State redwood parks in Mendocino, Humboldt and Del Norte Counties from 1991 to 1999. While both types of parks have millions of visitors, since 1991, the redwood parks experienced a 13 percent decline in visits while beaches experienced a six percent increase.

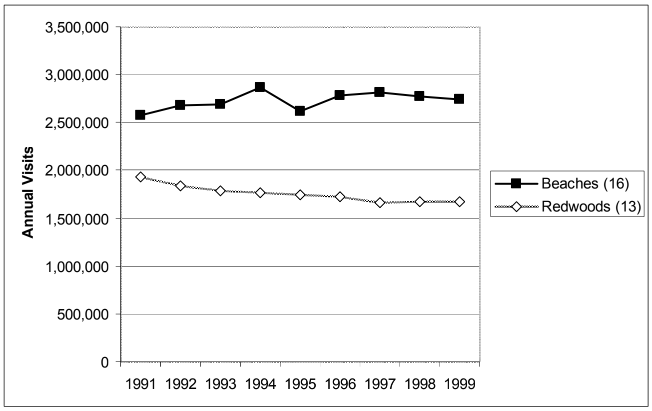


Figure 4—State Park Visits to Redwood and Beach Parks in Del Norte, Humboldt, and Mendocino counties. Source: FRAP 2003b

Table 3 compares use data for a number of redwood parks across the larger redwood region during the 1990s. One of the most significant relationships is the much higher use levels in the parks in close proximity to the populated San Francisco Bay Area.

Table 3—*Visits, area, use intensity, and distance from the San Francisco Bay Area for major redwood parks.*

Park	Acres	Annual visits	Visits/Acre	Distance from San Francisco
Muir Woods	549	1,311,000	2388	20
Armstrong Redwoods	780	200,000	256	80
Samuel P. Taylor	2,792	187,000	67	40
Henry Cowell Redwoods	4,376	292,000	67	50
Big Basin	17,478	907,000	52	40
Jedidiah Smith	10,165	177,000	17	130
Del Norte Coast Redwoods	6,325	84,000	13	380
Humboldt State	53,672	637,000	12	210
Redwood National and State Park	80,665	401,234	5	340

Source: FRAP 2003b

Figure 5 compares the increasing number of visits to the seven redwood state parks near the San Francisco Bay Area to the decreasing number of visits to the 13 redwood state parks in the Mendocino, Humboldt and Del Norte. This trend corresponds to California-wide trends of increases in day use of parks near metropolitan areas compared to flat or declining use patterns for more remote parks (FRAP 2003b). From an economic perspective, the major conclusion is that visitation and tourism related employment for redwood forests will continue to be an important aspect of our redwood forests but will probably not be an economic growth sector for areas not close to the San Francisco Bay Area.

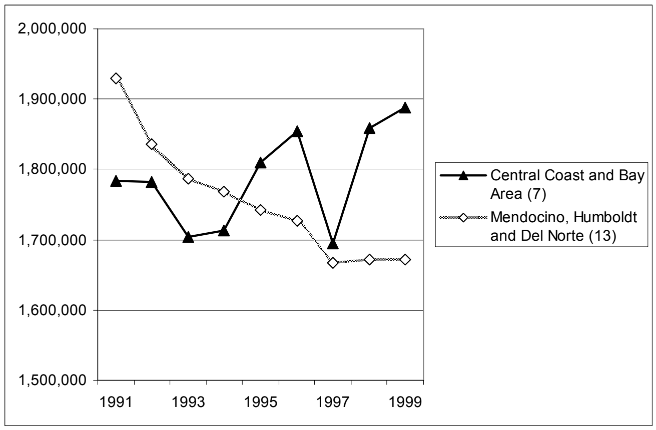


Figure 5—Redwood State Parks Visits for Metropolitan (Central Coast and Bay Area) and Non-metropolitan regions.

In comparison to declines in overall redwood timber harvests and park visits in the northern end of the redwood region, a very different trend is the significant increase in the area of redwood forests that also have an understory of houses. *Table 4* summarizes an overlay of 2000 Census block population densities with redwood forest area. Approximately 17 percent of the total redwood vegetation type is now in census blocks where there is at least one house per 40 acres. While these housing densities will often have limited impact on the number of trees per acre (a single house typically covers a quarter acre, 100 feet by 100 feet, with the main building, immediate driveway and irrigated yard) they do signify a shift in land use away from unfragmented forest management towards a mix of forest management and residential land use parcels. Nearly all of the residential lands are concentrated in four distinct areas. The most unique county is Santa Cruz, where over half of all the redwood forests in Santa Cruz are also part of the rural residential landscape. As figure 6 illustrates, the same pattern exists in the Russian River region of Sonoma, the Mendocino to Fort Bragg region of Mendocino County, and the Humboldt Bay region of Humboldt County.

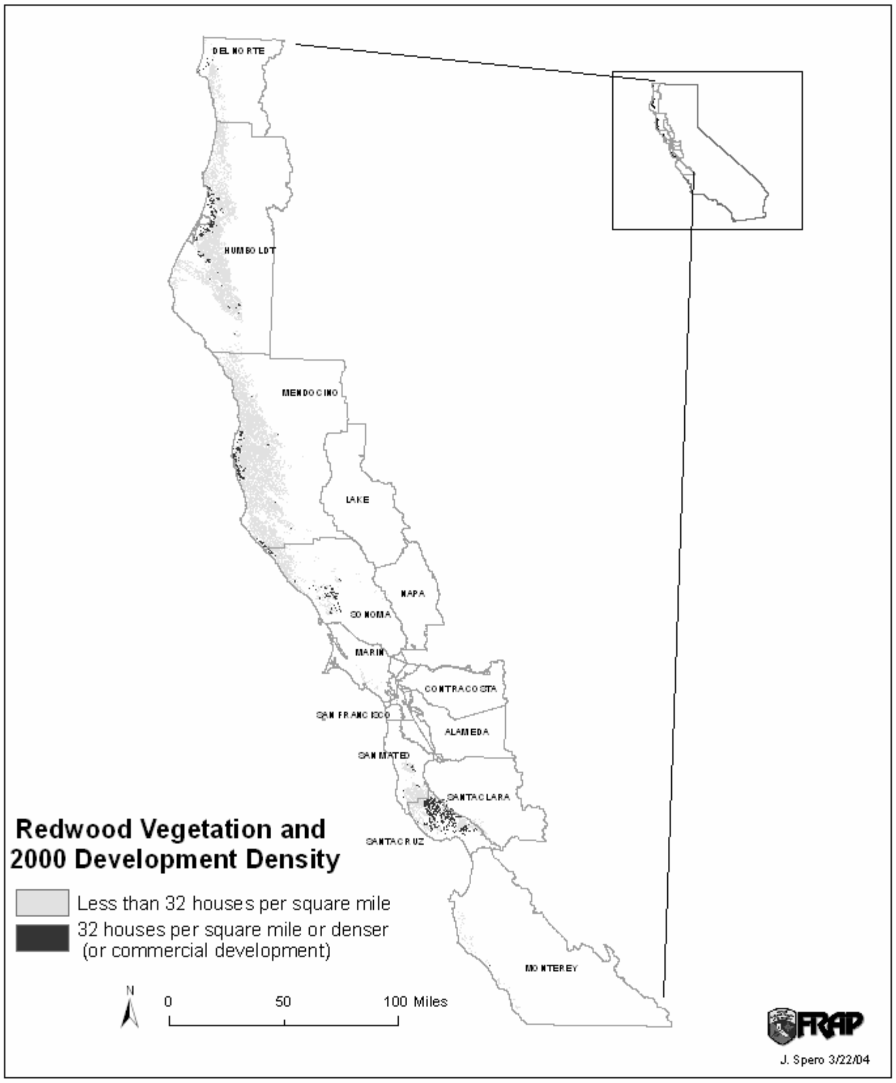


Figure 6—Residential housing densities in the redwood vegetation types.

Table 4—*2000 Census based analysis of average residential parcels in the redwood region.*

		Average Parcel Size based on 2000 Census block data				
Total acres	**County**	**> 1/40**	**1/40- 1/20**	**1/20- 1/5**	**1/5 - 1/1**	**< 1/1**
541,959	Mendocino	490,717	33,233	16,573	1,426	10
424,216	Humboldt	356,786	36,987	25,751	3,287	1,406
114,252	Santa Cruz	48,697	5,795	52,233	6,351	1,176
93,482	Sonoma	64,114	15,538	9,590	3,976	264
123,501	Others	111,106	3,138	7,240	1,927	89
1,297,410	Total	1,071,419	94,691	111,388	16,966	2,945
	Percent of total	83	7	9	1	0.2

Sources: FRAP 2002, US Census 2000

During the 1990s, significant shifts towards smaller lots occurred in four counties. *Table 5* shows that from 1990 to 2000 relatively few acres reached urban densities of more than one house per acre, but that the total acreage that moved into the rural residential category of more than one house per forty acres was more than the total acreage of new park acquisitions during the same period. *Table 6* shows the same data as a percentage of total acres in the counties. Santa Cruz had the largest percentage change, but the changes in the much larger county of Mendocino affected three times as many acres.

Table 5—*1990 to 2000 change in redwood acreage at different housing densities.*

		Average Parcel Size based on 2000 Census block data				
Total acres	**County**	**> 1/40**	**1/40 to 1/20**	**1/20 to 1/5**	**1/5 to 1/1**	**< 1/1**
541,959	Mendocino	-32,223	32,190	25	0	7
424,216	Humboldt	-13,504	11,658	1,502	-554	897
114,252	Santa Cruz	0	-12,311	12,244	-381	447
93,482	Sonoma	-3,170	991	1,611	568	0
123,501	Others	-54	-383	183	242	12
1,297,410	Total	-48,952	32,146	15,565	-124	1,364

Sources: FRAP 2002, US Census 2000

Table 6—*1990 to 2000 percentage change in redwood acreage at different housing densities.*

		Average Parcel Size based on 2000 Census block data				
Total acres	**County**	**> 1/40**	**1/40 to 1/20**	**1/20 to 1/5**	**1/5 to 1/1**	**< 1/1**
		-----------------------------------*percent*-----------------------------------				
541,959	Mendocino	-6	6	0	0	0
424,216	Humboldt	-3	3	0	0	0
114,252	Santa Cruz	0	-11	11	0	0
93,482	Sonoma	-3	1	2	1	0
123,501	Others	0	0	0	0	0
1,297,410	Total	-4	2	1	0	0

Sources: FRAP 2002, US Census 2000

Conclusion

A review of recent economic trends in the redwood region suggests the emergence of three new themes that will have major impacts on the future of redwood forests. First, it appears that the transition to a young growth redwood industry is essentially complete. Most of the major land owners also have forest management plans that include institutional arrangements to address fish and wildlife habitats, biodiversity, and watershed restoration investments. Habitat conservation plans, third party certification, and conservation easements are some of the institutional arrangements that can bring stability to long term forestland ownership and ensure sustainable management practices. However, continued decline in the amount of high value, very large redwood logs and relatively high operating costs required by California's overlapping regulatory systems (Dicus and Delfino 2003) may reduce landowner's interest to maintain large areas of expensive real estate in sustainable forest products based operations.

A continued decline in the timber-based economy will probably not be compensated by a growth in economic activity associated with redwood parks. Unlike visits to beach parks, visits to redwood parks declined in the 1990s in the non-metropolitan counties. Statewide, there are many indications that the benefits of day use and associated open space values will continue to increase where the forest is in close proximity to large population centers. However, shorter trips by local residents will not generate the level of jobs associated with overnight visitors.

One of the most significant trends is the increase in the extent of rural residential land use in redwood forests. Most of the redwood forests in four areas—most of Santa Cruz county, the Russian River region in Sonoma, the Fort Bragg to Mendocino region of central Mendocino coast and the southeast side of Humboldt Bay in Humboldt county—now effectively have an understory of houses and associated residential land uses. The environmental impacts in terms of altering wildlife habitats and new sources of water pollutants may be greater than those typical of current best forest management practices. In addition, there will be new economic costs associated with environmental mitigations, increased public safety requirements, and increased residential infrastructure requirements.

References

California Board of Equalization. 2004. **California timber harvests by county 1993 to 2002**. *Available at* http://www.boe.ca.gov/proptaxes/pdf/ytr3692to01.pdf

California Economic Development Department. 2004. **2000 Census – demographic profiles** *Available at* http://www.calmis.cahwnet.gov/htmlfile/subject/DP2000.htm

Dicus, C.A.; Delfino, K. 2003. **Comparison of the California forest practice rules and two major certification systems**. Technical report to the California Forest Products Commission.

Fire and Resource Assessment Program (FRAP). 2002. **Multi-source land cover, v02_1**. Sacramento, CA. *Available at* http://frap.cdf.ca..gov/projects/frapgisdata/select.asp.

Fire and Resource Assessment Program (FRAP). 2003a. **The changing California: forest and range 2003 assessment: assessment summary.** *See* http://www.frap.cdf.ca.gov /assessment2003/

Fire and Resource Assessment Program (FRAP). 2003b. **The changing California: forest and range 2003 assessment: wildland outdoor recreation assessment.** *See* http://www.frap.cdf.ca.gov/assessment2003/Chapter6_Socioeconomic/recreation.html

U.S. Census Bureau. 2000. **Summary file 1**. Web site accessed April 22, 2003. *See* http://www.census.gov/Press-Release/www/2001/sumfile1.html

Upland Log Volumes and Conifer Establishment Patterns in Two Northern, Upland Old-Growth Redwood Forests, A Brief Synopsis[1]

Daniel J. Porter[2] and John O. Sawyer[3]

Abstract

We characterized the volume, weight and top surface area of naturally fallen logs in an old-growth redwood forest, and quantified conifer recruit densities on these logs and on the surrounding forest floor. We report significantly greater conifer recruit densities on log substrates as compared to the forest floor. Log substrate availability was calculated on a per hectare basis using the line-intersect method. This method is used to estimate the amount of coarse woody debris present, by measuring log diameters intersected along randomly oriented transects. Conifer recruitment on intersected logs was characterized by recording recruit species, height class and substrate for each established individual. Conifer recruit densities on the forest floor were obtained for two old-growth redwood stands by establishing narrow strip plots. We found redwood logs of intermediate decay to be the most abundant woody substrate available for conifer recruitment. Where conifers had established on the forest floor, we found them most frequently on exposed mineral soil. These results confirm the use of logs by conifers in early establishment and growth, and suggest these substrates may affect stand composition and structure over the course of centuries. Differences in life-history traits between redwood and a potential competitor, western hemlock, along with the reported recruit densities, suggest redwood will remain the dominant conifer species in these stands, in the absence of significant disturbance.

Key words: coarse woody debris, disturbance, logs, nursery, recruitment

Introduction

Redwood *(Sequoia sempervirens)* is a long-lived (up to 2,000+ years) conifer that can attain great height (100+ meters tall) and massive girth (>3 meters dbh). In the last century, interest in the dwindling collection of remaining old-growth redwood stands has increased. Now, at less than 9.5 percent of their original extent (Fox 1989), these old-growth stands hold immeasurable aesthetic, cultural and spiritual values, in addition to their unique ecological qualities (Noss 2000).

The last three decades have produced a number of discoveries about redwood ecology. For example, researchers have found old-growth redwood forests contain

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15 – 17, 2004, Rohnert Park, California.

[2] Forest Ecologist. Save-the-Redwoods League. 114 Sansome Street, San Francisco, CA 94104. email: dporter@savetheredwoods.org

[3] Professor of Botany, Emeritus. Humboldt State University. 1 Harpst Street, Arcata, CA 95521. email: jos2@humboldt.edu

some of the largest volumes of standing and downed wood biomass in the world (Bingham 1984, Bingham and Sawyer 1988, Fujimora 1977) and may serve as significant carbon sinks, thereby influencing landscape-scale carbon cycles. Old-growth redwood forests contain little known assemblages of species including epiphytic lichens in the canopy (Sillett 1999), rich invertebrate collections on the forest floor (Olson 1992), and a highly diverse collection of fungal species (Largent 1998).

The Northern portion of redwood's range most closely resembles the coastal forests of Washington and Oregon with a notable exception; redwood takes the place of Douglas-fir *(Pseudotsuga menziesii)* as the dominant overstory tree while Douglas-fir assumes a sub-dominant role in newly cleared areas or canopy gaps. Both regions are underlain by sedimentary formations, where mass wasting and slide-and-flow processes are common (Aalto and Harper 1989). Fire and windthrow also are modes of disturbance in upland settings. Signs of low intensity surface fires can be found in old-growth redwood stands (Stuart 1987), and select old-growth trees have fire scars that extend 30 to 70 meters up the boles. Severe windstorms hitting the coast of northern California and southern Oregon uproot old-growth trees and break off the trunks of others (Sawyer and others 2000). Massive logs (>two meters in diameter) can cover as much as 19 percent of the redwood forest floor. Like the coastal forests of Oregon and Washington, the predominant disturbance processes (low intensity fire and wind throw) tend to accumulate logs. Redwood logs, because of their sheer size and resistance to decay (Espinosa-Garcia and others 1996), can last for centuries on the forest floor, making them important components of ecosystem structure and function (Franklin and others 1981).

Upland forests of northern Humboldt and Del Norte counties provide the setting for this study. These stands appear to contain very few redwood seedlings and saplings. Young trees can be found along a trail's edge, or on a mound of upturned road, but among undisturbed ground, young redwoods appear to exist at extremely low densities. By contrast, western hemlock seedlings and saplings are fairly abundant, growing on logs, the forest floor, and even on other trees. Redwood seedlings and saplings also can be found on logs, but less frequently than western hemlock (Bingham 1984).

How it is then that redwood remains the dominant conifer in these forests? What disturbances, if any, affect stand composition over the course of centuries? Some ecologists have suggested that western hemlock is overtaking redwood as the dominant conifer, because pre-European disturbance regimes have been altered (Cooper 1965, Daubenmire and Daubenmire 1975). What disturbances might managers introduce to maintain the desired stand structure and species composition if it is changing due to a lack of redwood regeneration? Such questions are beyond the scope of this study. However, this study seeks to describe conifer establishment patterns in two northern old-growth upland forests to lay the groundwork for addressing these questions in the future.

This study was designed as a continuation of earlier research conducted in 1982 to 1984 (Bingham 1984). This earlier work accomplished two objectives: (1) estimate, by species and decay class, the quantity of naturally fallen decaying logs on the forest floor and (2) characterize conifer regeneration on logs in an upland old-growth redwood forest.

Naturally fallen logs play a central role in the abiotic and biotic functions of

Pacific Northwest forests (Franklin and others 1981, Norse 1990). Logs help maintain the productive capacity of the soil by helping maintain higher moisture levels (Bernsten 1960, Maser and others 1984), preventing surface erosion, moderating soil surface temperatures and providing a source of nutrient inputs (Maser and others 1984, Triska and Cromack 1980). Large coarse woody debris also store significant amounts of carbon and nitrogen (Grier 1978, Larsen and others 1978, Roskoski 1977), which gradually become available to other forest organisms.

Logs are an important substrate for tree establishment in a number of forests (Stewart and Burrows 1994) including Appalachian spruce-fir forests (White and others 1985), tropical forests (Nakashizuka 1989), and New Zealand temperate beech (Nothofagus) forests (Stewart and Burrows 1994). The phenomenon is well documented in northern temperate rainforests (Bingham and Sawyer 1988, Gray and Spies 1997, Thornburgh 1969). Logs serve as water reservoirs in forests that experience a summer dry period (Gray and Spies 1995, Harmon and Cromack 1987). One study found logs of intermediate decay to have almost three times the moisture content than the forest floor in August, and less seasonal variation in moisture (Marra and Edmonds 1994).

Higher nutrient levels on logs may also facilitate conifer establishment on these substrates compared to the forest floor. Nutrients are generally abundant on the forest floor in the northern and central redwood forests (Popenoe 1987, Zinke and others 1996), however, nitrogen may be limiting (Sawyer and others 2000). Logs may provide a level of nitrogen not available on the forest floor. For example, in the western hemlock/spruce forests of coastal Oregon, Grier (1978) reported a rapid increase in log nitrogen and other nutrients the first 20 years on the ground. A separate study of respiration rates on logs and the forest floor in temperate rainforests, indicate higher respiration rates on newly decaying logs (Marra and Edmonds 1994) compared to the forest floor. These researchers suggest the higher rates can be attributed to the release of more labile forms of carbon and nitrogen in the early stages of log decay.

Methods

We selected two upland, old-growth redwood stands in northwestern California to study conifer establishment patterns on logs and the forest floor. The northern Yurok site was chosen because it resembles Prairie Creek Redwoods State Park, where Bingham's study (Bingham 1984) was conducted. At the Yurok site, we calculated conifer recruit densities on logs and the forest floor. At the southern Prairie Creek site, we calculated conifer recruit densities on the forest floor, so that we could compare these to conifer recruit densities on logs calculated by Bingham (1984) at this same site. Combining Bingham's study with this study gives two estimates of coarse woody debris volumes for upland old-growth redwood forests, and two descriptions of conifer establishment patterns on logs and the forest floor.

To estimate the area's volume of coarse woody debris, we used the same line intersect technique (Van Wagner 1968) as Bingham (1984). A temporary system of 13 transects were established oriented in southeasterly and southwesterly directions down the slope. Line length varied with topography (range 180 meters to 420 meters, mean = 304.29 meters). Along each line we established intercept points every 60 meters. At each of these intercept points, we chose a random direction, and established a short (30 meter) transect. Along this transect, we measured diameter and length for every log (diameter >25 centimeters and length >4 meters). This method has been shown to best estimate wood volumes with the least effort (Brown and Roussopoulos 1974). Estimates obtained for the Yurok site are directly comparable to those generated by Bingham (1984).

The line intersect method for estimating coarse woody debris volumes also provided an opportunity to assess conifer establishment on decaying logs. The sampled logs were inspected on the top and sides for established (>5 centimeters tall) conifers. Species and height classes were recorded and later collapsed into three life stages: seedlings (>5 cm to 50 cm), saplings (>50 cm to 6m) and trees (>6m). Redwood sprouts originating from redwood logs were not included. The underlying logs were identified to species using anatomical features of the wood and placed into three decay classes:

1. Slightly decayed logs–bark is entirely or mostly present. Large branches may be intact. Plant growth on the log, if any, is light and can include young, understory shrub seedlings.

2. Intermediately decayed logs–broadest category. Materials are mostly sound, may contain some rot, but can support their own weight. Branch material is often gone but bark may still be present. Plant growth on the log can be quite extensive, including mature shrubs and young trees.

3. Highly decayed logs–material are mostly rotten. Logs are oval shaped and cannot support their own weight. Since this category of log resembles the forest floor, plant growth on the log is variable.

A log species and decay class together constitute a substrate; thus three predominant log species (redwood, Douglas-fir, hemlock) and three decay classes gave a total of nine log substrates.

Established conifer density on the forest floor was sampled for both sites using narrow strip plots (2 meters by 60 meters). At the Yurok site, these strip plots were oriented parallel to the lines used to establish the intercepts. We established the strip plots from topographic high points at both sites, and worked downhill for varying distances. Line lengths varied because of variable distances between site boundaries though the sampling units used to compare recruit densities (“strip plots”) were always the same.

We calculated log volume per hectare for each of the nine log substrates using Van Wagner’s equation (1) (Brown and Roussopoulos 1974, Van Wagner 1968):

$$\text{Equation 1: } v = \frac{\Pi^2 \sum d^2}{l} ac$$

Where,

v = volume (m3) of logs per m2 of forest floor.

d = diameter of each log intersected (m).

l = total length of all transects (m).

a = correction for non-horizontal orientation bias.

c = the average slope correction factor.

We calculated top surface area (TSA) estimates of log substrates per hectare of forest floor using the following equation (Lamberson 1984):

$$\text{Equation 3: TSA} = \frac{2.13\sum d}{l} ac$$

where,

TSA = the upper ½ surface area of logs, excluding ends, per unit area of forest floor (m^2/m^2).

d = the diameter of each log intersected (m).

l = the total length of all transects (m).

c = the slope correction factor.

a = correction for non-horizontal orientation bias.

Estimates of volume, weight and top surface area are all converted to a per hectare basis by multiplying the results by 10,000 m^2/ha.

Results

Log volume and weight estimates between stands were highly variable. The log total volume estimate for the Yurok site (1,457 m^3/ha) is one and a half times larger than the Prairie Creek site estimate (957 m^3/ha) *(tables 1, 2)*. Redwood logs make up the largest percentage of the overall log volume at both sites and Douglas-fir constitutes a notable percentage at the Prairie Creek site. Redwood logs of intermediate decay make up the majority of log volume at the Yurok and Prairie Creek sites, 64.8 percent and 55.2 percent respectively.

Table 1—*Volumes (m^3/ha) and weights (mt/ha)*[a] *of decaying logs on the forest floor at the Yurok site. Decay classes: 1- slightly decayed, 2- intermediately decayed, 3- highly decayed.*

	Redwood		W. hemlock		Douglas-fir		Totals	
Decay Class	Volume (m^3/ha)	Weight (mt/ha)	Volume (m^3/ha)	Weight (mt/ha)	Volume (m^3/ha)	Weight (mt/ha)	Volume (m^3/ha)	Weight (mt/ha)
1	55.15	22.06	0.00	0.00	7.66	3.68	62.81	25.74
2	943.66	358.59	2.76	1.05	41.59	18.30	988.01	377.94
3	394.16	118.25	4.90	1.47	7.43	2.23	406.49	121.95
Total	1,393	499	8.00	3.00	57.00	24.00	1,457.00	525.63

[a] mt = metric tons

Table 2—*Volumes (m^3/ha) and weights (mt/ha) of decaying logs on the forest floor at Prairie Creek (Bingham 1984). Decay classes: 1- slightly decayed, 2- intermediately decayed, 3- highly decayed.*

	Redwood		**W. hemlock**		**Douglas-fir**		**Totals**	
Decay Class	**Volume (m^3/ha)**	**Weight (mt/ha)**	**Volume (m^3/ha)**	**Weight (mt/ha)**	**Volume (m^3/ha)**	**Weight (mt/ha)**	**Volume (m^3/ha)**	**Weight (mt/ha)**
1	56	23	22	9	16	8	94	40
2	528	202	10	4	68	30	606	236
3	236	71	4	1	17	5	257	77
Total	820	296	36	14	101	43	957	353

Redwood logs represent most of the log area available for conifer colonization at both sites: 94 percent at the Yurok site and 78 percent at the Prairie Creek site *(tables 3, 4)*. Redwood logs of intermediate decay are the most readily available of all substrate types representing 62.1 percent of the available log area at Yurok site and 45.6 percent at the Prairie Creek site.

At both sites, conifer establishment on logs is patchy. Recruits were found on 64 of the 160 logs (40 percent) at the Yurok site, and Bingham (1984) found recruits on 130 of 276 logs (47 percent) at the Prairie Creek site. A total of 415 recruits were found on logs at the Yurok site, and 1,048 recruits were found on logs at the Prairie Creek site (Bingham 1984).

At both sites, of the number of recruits by log size and decay class was highly variable. For example, at the Yurok site a 1.5 meter diameter redwood log had 46 recruits on its upper surface, and a 1.7 meter diameter redwood log of similar decay and length had no recruits. Bingham (1984) had similar results. Overall recruitment on logs of all species was higher at Prairie Creek (197 recruits/ha,) than at the Yurok site (119 recruits/ha) despite more available log substrate at the Yurok site.

As in Bingham's study, western hemlock was the most abundant conifer species on logs, with an estimated 103 recruits/ha at the Yurok site. There are six redwood recruits per hectare at the Yurok site (third most common species) and 56 redwood recruits per hectare at the Prairie Creek site. Redwood logs of intermediate decay hold the highest percentage of conifers of all log substrates (72.5 percent Yurok site; 44.2 percent Prairie Creek site).

Table 3—*Top surface areas of each decay class available for conifer establishment at the Yurok site. Values represent the upper one-half surface area, excluding log ends, per hectare of forest floor (m^2/ha). Decay classes: 1- slightly decayed, 2- intermediately decayed, 3- highly decayed.*

Decay Class	Redwood	W. hemlock	Douglas-fir	Total	Percent
1	47.66	0.00	18.54	66.20	3
2	1206.16	7.94	67.52	1281.62	66
3	568.00	10.59	17.21	595.80	31
Total	1821.82	18.53	103.27	1943.62	
Percent	94	1	5		100

Table 4—*Top surface areas of each decay class available for conifer establishment at the Prairie Creek site (Bingham 1984). Values represent the upper one-half surface area, excluding log ends, per hectare of forest floor (m^2/ha). Decay classes: 1- slightly decayed, 2- intermediately decayed, 3- highly decayed***.**

Decay Class	Redwood	W. hemlock	Douglas-fir	Total	Percent
1	86	50	44	180	12
2	681	31	135	847	57
3	403	18	45	466	31
Total	1170	99	224	1493	
Percent	78	7	15		100

This study confirms the observations of researchers and park managers that redwood forest floor recruit densities are low to non-existent (12 redwood recruits/ha at the Yurok site and 16 redwood recruits/ha at the Prairie Creek site). Nearly all of these recruits were greater than six meters tall, making it difficult to determine their origin (seed or sprout).

Despite low recruit densities on logs and the forest floor at the Yurok site, differences between these two major substrate categories are still apparent. The proportion of forest floor sampling units (in other words, strip plots) containing no recruits is nearly twice as high (84.5 percent; n = 58) as the same proportion for the log sampling units (46.6 percent; n = 71). Similarly, 72 percent of ground sampling units at Prairie Creek had no recruits, and 53 percent log sampling units had no recruits (Bingham 1984). When recruits are present on either of these two major substrate categories, differences in mean conifer densities (# recruits/m^2) between logs and the forest floor at the Yurok site are statistically significant (p = 0.001) *(table 5)*. An analogous comparison for the Prairie Creek site cannot be made because the raw data from Bingham's original study are not available. However, the overall number of recruits on logs per hectare at Prairie Creek site (197) is clearly higher than the number of recruits per hectare of forest floor (38). Differences between recruit densities on the forest floor between sites were not significant (p = 0.506).

Table 5—*Average number of conifer recruits when recruits present, per square meter of substrate (± S.D.). Recruit densities on logs vs. the forest floor are significantly different at the Yurok site (p = 0.001). Forest floor recruit densities between sites are not significantly different (p = 0.506).*

Substrate Category	Yurok site	Prairie Creek site
Logs	0.082 (0.10)	Not available
Forest floor	0.012 (0.011)	0.015 (0.010)

Conclusions

Upland old-growth redwood forests in the northern portion of the species range contain some of highest quantities of coarse woody debris in the world (Bingham and Sawyer 1988, Franklin and Waring 1980, Fujimora 1977). This is in large part due to the massive size of live trees, redwood's slow rate of decomposition (Espinosa-Garcia and others 1996) and a combination of historic forest processes that favored the accumulation of log material.

The role of fire in the redwood region has been debated for decades. Early studies (for example, Veirs 1980, Stuart 1987) relied on stump records or indirect measurements (sprouting frequency) and tended to over-estimate fire return intervals. More recent studies (Brown and Baxter 2003; Brown and others 1999; Brown and Swetnam 1994; Finney and Martin 1989, 1992) have taken direct measurements nearer ground level, thereby recording more frequent fires (two 22-year intervals) characterized by low flame length typical of many old-growth fires. Together, these studies suggest that high severity fires have historically been relatively infrequent and that low severity fires occurred every two to three decades. Though the volume of log material consumed by old-growth wildfires has not been recorded, these studies in combination with the log volumes reported in this study suggest that rates of log accumulation (windthrow) exceed the rates of log export (via fire) in northern old-growth redwood forests.

Considerable variation in the quantity of coarse wood debris exists between apparently similar old-growth sites. Slope, aspect and distance to the ocean effect overstory species composition (Mahony 2000), which ultimately influences log volume. Where geographic locations are similar, mid and low slope positions generally support a higher proportion of large redwood trees than ridge and upper slope positions, where white woods become more common.

Conifer recruit densities are highest on redwood logs of intermediate decay. This substrate type is the most abundant in the stands studied. Other log substrates were not present in sufficient quantity to resolve substrate preferences between log species. Conifer recruit densities on the forest floor are highest on exposed mineral soil, as also indicated anecdotally in previous studies (for example, Metcalf 1924), but in general, the forest floor was found to be an inhospitable place for recruitment. Conifer recruit densities on logs are significantly higher than those found on the forest floor though on both general substrate categories, the distribution of recruits was highly variable. Studies have suggested that this variability may be related to microsite variables that effect establishment (for example, Gray and Spies 1997).

Western hemlock establishes frequently on log substrates (Christy and Mack 1983, Thornburgh 1969, Zobel and Antos 1991) and appears to survive in a co-dominant and suppressed form on redwood logs of intermediate decay in northern old-growth redwood forests. Differences in gross life history traits (for example, size, longevity) between redwood and western hemlock and the recruit densities presented here, suggest redwood will remain dominant in these stands, even in the absence of significant disturbance (for example, fire). This conclusion pertains only to northern upland old-growth redwood forest.

References

Aalto, K.R.; Harper, G.D. 1989. **Geologic evolution of the northernmost coast ranges and western Klamath mountains, California**. Washington, DC: American Geophysical Union.

Bernsten, C.M. 1960. **Planting Sitka spruce and Douglas-fir on decayed wood in coastal Oregon**. Res. Note PNW-197. Portland, OR: Pacific Northwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture.

Bingham, B.B. 1984. **Decaying logs as a substrate for conifer regeneration in an upland old-growth redwood forest**. Arcata, CA: Humboldt State University. M.S. thesis.

Bingham, B.B.; Sawyer, J.O. 1988. **Volume and mass of decaying logs in an upland old-growth redwood forest**. Canadian Journal of Forest Research 18: 1649-1651.

Brown, J.K.; Roussopoulos, P.J. 1974. **Eliminating biases in the planar intersect method for estimating volumes of small fuels**. Forest Science 20: 350-356.

Brown, P.M.; Baxter, W.T. 2003. **Fire history in coast redwood forests of the Mendocino Coast**. Northwest Science 77: 147-158.

Brown, P.M.; Swetnam, T.W. 1994. **A cross-dated fire history from coast redwood near Redwood National Park, Californi**a. Canadian Journal of Forest Research 24: 21-31.

Bunnell, F.L.; Chan-McLeod, A.C. 1997. **Terrestrial vertebrates**. In: Schoonmaker, P.K.; von Hagen, B.; Wolf, E.C., eds. The rainforest of home: profile of a North American bioregion. Washington, DC: Island Press; 103–130.

Christy, J.E.; Mack, R.N. 1983. **Variation in demography of juvenile *Tsuga heterophylla* across the substratum mosaic**. Journal of Ecology 71: 75–93.

Cooper, D.W. 1965. **The coast redwood and its ecology**. Eureka, CA: Humboldt County Agricultural Extension Service.

Daubenmire, R.F.; Daubenmire, J. 1975. **The status of coastal redwood, *Sequoia sempervirens***. Unpublished report prepared for the USDI, National Park Service. Arcata, CA: Redwood National Park.

Espinosa-Garcia, F.J.; Rollinger, J.L.; Langeheim, J.H. 1996. **Coastal redwood leaf endophytes: their occurrence, interactions, and response to host volatile terpenoids**. In: Redlin, S.C.; Carris, L.M., eds. Endophytic fungi in grasses and woody plants: systematics, ecology and evolution. St. Paul, MN: American Phytopathology Society; 101–130.

Finney, M.A.; Martin, R.E. 1989. **Fire history in a *Sequoia sempervirens* forest at Salt Point State Park, California**. Canadian Journal of Forest Research 19: 1451–1457.

Finney, M.A.; Martin, R.E. 1992. **Short fire intervals recorded by redwoods at Annadel State Park, California**. Madrono 39: 251-262.

Fox, Lawrence III. 1989. **A classification, map and volume estimate for the coast redwood forest in California**. Final report. Interagency agreement number 8CA52849. Sacramento, CA: The Forest and Rangeland Resources Assessment Program, California Department of Forestry and Fire Protection.

Franklin, J.F.; Waring, R.H. 1980. **Distinctive features of the northwestern coniferous forest: Development, structure and function**. In: Waring, R.H., ed. Forests: fresh perspectives from ecosystems analysis. Annual Biology Colloquium, 1979. Corvallis, OR: Oregon State University: 59–85.

Fujimori, T. 1977. **Stem biomass and structure of a mature *Sequoia sempervirens* stand on the Pacific Coast of northern California**. Journal of the Japanese Forestry Society 59: 431-441.

Gray, A.N.; Spies, T.A. 1995. **Water content measurement in forest soils and decayed wood using time domain reflectometry**. Canadian Journal of Forest Research 25: 376-385.

Gray, A.N.; Spies, T.A. 1997. **Microsite controls on tree seedling establishment in conifer forest canopy gaps**. Ecology 78: 2458–2473.

Grier, C.C. 1978. **A *Tsuga heterophylla-Picea sitchensis* ecosystem of coastal Oregon: decomposition and nutrient balances of fallen logs**. Canadian Journal of Forest Research 8: 198–206.

Harmon, M.E.; Cromack, K., Jr. 1987. **Course woody debris in mixed-conifer forests, Sequoia National Park, California**. Canadian Journal of Forest Research 17: 1265–1272.

Largent, D.L. 1998. **Species of mushrooms, boletes, chanterelles, coral fungi, and tooth fungi found in the Redwood National and States Parks between 1993 and 1997**. Unpublished report. Redwood National Park, Arcata, CA.

Larsen, M.J.; Jurgensen, M.F.; Harvey, A.E. 1978. **N_2 fixation associated with wood decayed by some common fungi in western Montana**. Canadian Journal of Forest Research 8: 341-345.

Mahony, T.M.; Stuart, J.D. 2000. **Old-growth forest associations in the northern range of coastal redwood**. Madrono 47: 53-60.

Marra, J.L.; Edmonds, R.L. 1994. **Coarse woody debris and forest floor respiration in an old-growth coniferous forest on the Olympic Peninsula, Washington, USA**. Canadian Journal of Forest Research 24: 1811-1817.

Maser, C.; Trappe, J.M.; Cline, S.P.; Cromack, K., Jr.; Blaschke, H.; Sedell, J.R.; Swanson, F.J. 1984. **The seen and unseen world of the fallen tree.** Gen. Tech. Rep. PNW-GTR-164. Portland, OR: Pacific Northwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture.

Metcalf. W. 1924. **Artificial reproduction of redwood (*Sequoia sempervirens*)**. Journal of Forestry 22: 873-893.

Nakashizuka, T. 1989. **Role of uprooting in composition and dynamics of an old-growth forest in Japan**. Ecology 70: 1273–1278.

Norse, E.A. 1990. **Ancient forests of the Pacific Northwe**st. Washington, DC: Island Press.

Noss, R.F. 2000. **The redwood forest: history, ecology and conservation of the coast redwoods**. Covelo, CA: Island Press.

Olson, D.F.; Roy, D.F.; Walters, G.A. 1990. ***Sequoia sempervirens* (D. Don) Endl**. In: Burns, R.M.; Honkala, B.H., eds. **Silvics of North America**. **Volume 1. Conifers**. Agric. Handbook. 654. Washington, DC: U.S. Department of Agriculture; 541-551.

Olson, D.M. 1992. **The northern spotted owl conservation strategy: implications for Pacific Northwest invertebrates and associated ecosystem processes**. Final Report prepared for the Northern Spotted Owl EIS Team. USDA Forest Service, Order number 40-04HI-2-1650.

Popenoe, J.H. 1987. **Soil series descriptions and laboratory data from Redwood National Par**k. Technical Report 20. Orick, CA: Redwood National Park.

Roskoski, J.P. 1977. **Nitrogen fixation in northern hardwood forests**. New Haven, CT: Yale University. Ph.D. dissertation.

Sawyer, J.O.; Sillett, S.C.; Popenoe, J.H.; LaBanca, A.; Scholars, T.; Largent, D.L.; Euphrat, F.; Noss, R.F.; Van Pelt, R. 2000. **Characteristics of redwood forests**. In: Noss, R.F. ed. The redwood forest: history, ecology and conservation. Covelo, CA: Island Press; 39-79.

Sillett, S.C. 1999. **Tree crown structure and vascular epiphyte distribution in *Sequoia sempervirens* rain forest canopies**. Selbyana 20: 76-97.

Snyder, J.A. 1992. **The ecology of *Sequoia sempervirens*: an addendum to On the edge: Nature's last stand for coast redwoods**. San Jose, CA: San Jose State University. M.S. thesis.

Stewart, G.H.; Burrows, L.E. 1994. **Coarse woody debris in old-growth temperate beech (Nothofagus) forests of New Zealand.** Canadian Journal of Forest Research 24: 1989-1996.

Stuart, J.D. 1987**. Fire history of an old-growth forest of *Sequoia sempervirens* (Taxodiaceae) forest in Humboldt Redwoods State Park, CA**. Madrono 34: 128–141.

Thornburgh, D.A. 1969. **Dynamics of the true fir-hemlock forests of western Washington**. Seattle: University of Washington. Ph.D. dissertation.

Triska, F.J.; Kromack, K., Jr. 1980. **The role of wood debris in forests and streams**. In: Waring, R.H., ed. Forests: fresh perspectives from ecosystem analysis. Corvallis, OR: Oregon State University Press; 171-190.

Van Wagner, C.E. 1968. **The line intersect method in forest fuel sampling**. Forest Science 14: 20-26.

Viers, S.D. 1981. **The influence of fire in coastal redwood forests**. In: Proceedings of the fire history workshop; 1980 October 20-24; Tucson, AZ. Gen. Tech. Rep. RM-GTR-81. Fort Collins, CO: Rocky Mountain Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 93-95.

White, P.S.; MacKenzie, M.D.; Busing, R.T. 1985. **Natural disturbance and gap phase dynamics in southern Appalachian spruce-fir forests**. Canadian Journal of Forest Research 15: 233-240.

Zinke, P.J.; Stangenberger, A.G.; Bertenshaw, J.L. 1996. **Pattern and process in forests of *Sequoia sempervirens* (D.Don) Endl**. In: LeBlanc, J. ed. Proceedings of the conference on coast redwood forest ecology and management; June 18-20, 1996; Arcata, CA. University of California, Cooperative Extension, Forestry.

Zobel, D.B.; Antos, J.A. 1999. **Growth and development of natural seedlings of Abies and Tsuga in old-growth forests**. Journal of Ecology 79: 985-998.

Tree Biomass Estimates on Forest Land in California's North Coast Region[1]

Tian-Ting Shih[2]

Tree biomass is one essential component in a forest ecosystem and is getting more attention nowadays due to its close relationship to estimating forest fuel, biogenic volatile organic compound (BVOC) emission, carbon sequestration, energy production, and other natural and social resources uses and impacts.

A biomass estimator, BMTree, is constructed to embed a large collection of biomass equations based on tree species, diameter, height, and other tree variables to calculate oven-dried biomass weight of each sampling tree by tree components including boles, barks, live branches, and foliages. This study uses the most current Forest Inventory and Analysis (FIA) data as the input for the analysis on California's North Coast region and formulates the output into a biomass density map using Wildlife Habitat Relationship (WHR) types and into biomass weight tables by forest type, ownership, size class, and other forest variables. TimSuM, a timber supply model, is then employed to project the region's tree biomass weights for the next two decades under various scenarios on forest growth, mortality, harvest, and land base.

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Fire and Resource Assessment Program, California Department of Forestry and Fire Protection, 1920 20th Street, Sacramento, CA 95814, (916) 227-2653. email: tian-ting.shih@fire.ca.gov

Using Scientific Information to Develop Management Strategies for Commercial Redwood Timberlands[1,2]

Jeffrey C. Barrett[3]

Abstract

In 1999, PALCO (Pacific Lumber Company), a private landowner, and the state and federal governments agreed to implement a unique Habitat Conservation Plan (HCP) on 89,000 hectares of commercial redwood and Douglas-fir timberlands in Humboldt County, California. The aquatics portion of the PALCO HCP contains a set of "interim" conservation strategies developed using regional studies of habitat conditions and threats to species. The HCP, however, also requires PALCO and the state and federal governments to undertake extensive landscape level studies, and in particular, watershed analysis, for the purpose of cooperatively developing and testing the effectiveness of land management prescriptions tailored to address local watershed specific conditions.

In the first five years of implementing the PALCO HCP, the interdisciplinary team of PALCO and agency representatives successfully developed an analytical framework for conducting watershed analysis, completed such analysis for three watersheds, and developed new operating prescriptions which target watershed specific issues while maintaining operational flexibility and economic viability. This paper discusses these studies, and in particular, the application of specific scientific findings to the development of management approaches. In the Freshwater watershed, results of the sediment budget supported easing mass wasting avoidance operating prescriptions and instead increasing prescriptions to reduce road surface erosion. Conversely, the Van Duzen and Lower Eel River watershed sediment budgets led the interdisciplinary team to refine mass wasting prescriptions by utilizing a new risk based model which provides greater protections for near stream environments. Uniformly, high stream canopy levels combined with surveys of existing large woody debris and development of local recruitment curves support the easing of no harvest restrictions in streamside riparian management zones in all three watersheds.

Key words: HCP, sediment budgets, watershed analysis

Introduction

This paper discusses my experience in the implementation of the PALCO Habitat Conservation Plan (HCP), and in particular, the development and use of site-specific watershed studies to develop a "custom" set of management prescriptions for PALCO's lands. What sets these experiences apart from watershed work done by other parties is its location within the Redwood Zone, and the very significant differences in dominant watershed processes and management effects upon these

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] The opinions expressed within this paper are those of the author and do not represent the official positions or findings of my organization.

[3] PALCO, PO Box 37, Scotia, CA 95565. email: jbarrett@scopac.com

processes on relatively small scales. In addition, I discuss some observations on how to successfully implement site-specific watershed studies within the context of protecting listed aquatic species.

Study Area

PALCO owns approximately 89,000 hectares of coastal redwood *(Sequoia sempervirens)* and Douglas-fir *(Pseudotsuga menziesii)* forests in northwestern California (approximately 165 km south of the Oregon border). Company lands are zoned for commercial timber production as their primary use. Although various research and monitoring studies are being conducted across the ownership, this paper focuses on efforts in three basins: Freshwater Creek, the Van Duzen River, and the Lower Eel River *(fig. 1)*.

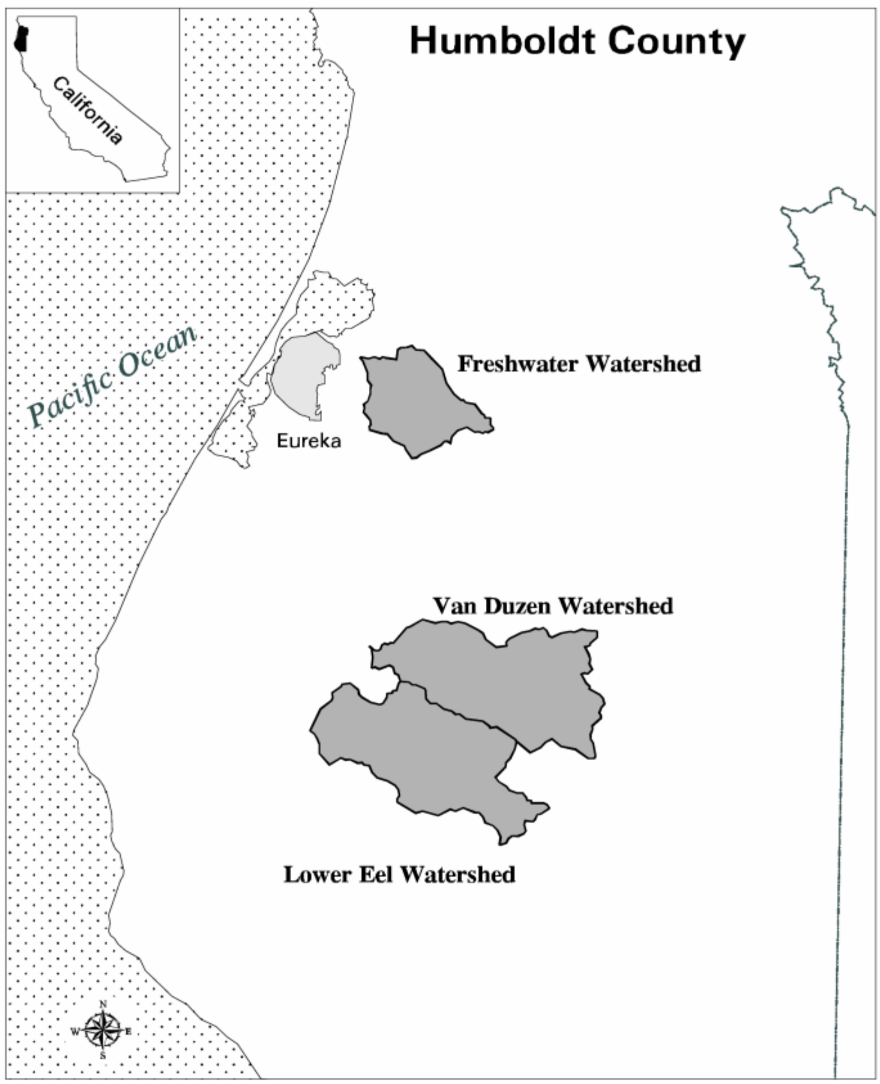


Figure 1—Map of PALCO's ownership the Freshwater, Van Duzen, and Lower Eel watersheds.

The Freshwater Creek watershed is an 81 km^2 drainage basin located approximately eight km east of Eureka, California in Humboldt County. Approximately 77 percent of the watershed is owned and managed for timber by PALCO. Small, private residences and several ranches comprise most of the remainder of the lands in the basin.

The Van Duzen River watershed area is a 185 km^2 portion of the lower Van Duzen River basin located approximately 25 km southeast of Eureka. PALCO owns the majority of the lands within the study area. Other commercial timberlands, state and county parks, ranches, cattle and dairy, and rural development parcels make up the remainder of the study area.

The Lower Eel watershed area is a 179 km^2 portion of the lower Eel River basin, located approximately 35 km south of Eureka. PALCO owns over 80 percent of the lands within the study area, which is composed almost entirely of commercial timberlands.

The geology in all three basins is dominated by marine sediments of relatively recent origin that have been uplifted through tectonic activity, with uplift rates on the order of 6.1 to 10.2 mm per year (Carver and Burke 1992). These high tectonic uplift rates have resulted in relatively extensive folding, faulting, and associated seismic activity. Freshwater has a relatively lower uplift rate, and is dominated by a rolling typography with more limited incised topographic features, and a relatively lower incidence of mass failure processes. By contrast, the Van Duzen and Lower Eel areas experience a higher tectonic uplift rate resulting in much steeper topography, deeply incised features, and a higher prevalence of mass wasting processes.

The relatively recent origin of these marine sediments results in geologies that are weak, highly erodible, and susceptible to mass failure. Soils derived from these parent geologies are characteristically deep (>3 m) and are dominated by silts, clays, and other fine-grained materials.

Methods

A PALCO/agency team modified the watershed analysis methods developed by Washington DNR (WADNR 1994) to make them more applicable to PALCO's lands (PALCO 2000). The work presented here focuses on findings relative to sediment budgets and riparian condition. All such work was done at a level equaling or exceeding so-called "Level II" studies within the DNR methodology.

Sediment budgets were developed utilizing field-based measurements and computer modeling of hillslope and road mass wasting, road and harvest unit surface erosion, and streamside and bank-related landsliding. The sediment budgets developed estimates for both background and/or natural levels of sediment input and management-related sources.

Assessment of riparian function focused on levels of canopy closure that provide shade and the potential for streamside forests to provide large woody debris to streams. Canopy levels were estimated from aerial photographs and were ground truthed. The potential for large wood recruitment was assessed based on the size and density of conifer trees along streams.

Criteria for development of site-specific management prescriptions from the watershed analyses included the requirement for habitat to maintain, or achieve over time, a matrix of "properly functioning conditions" (PFCs) contained in PALCO's HCP (PALCO 1999). Analyses important to the prescription development phase include: 1) estimated historic sediment delivery as a function of distance from streams, slope, and stream type (in other words, fish bearing, non-fish bearing, ephemeral); and 2) large woody debris source distance curves using empirically derived field data.

Results

Watershed Specific Sediment Budgets

In Freshwater, road-related erosion constituted more than 80 percent of all management-related sediment inputs *(fig. 2)*. Of this, by far the most important was road-related surface erosion. This finding reflects the high density of roads in this basin (>2.5 km/km^2), the erodibility of the soils underlying many of the road prisms, and the low incidence of mass wasting as a sediment source. Overall, annual inputs of management-related sediment were a little more than double the estimate of natural sediment inputs (161 metric tons/km^2 versus 68 metric tons/km^2).

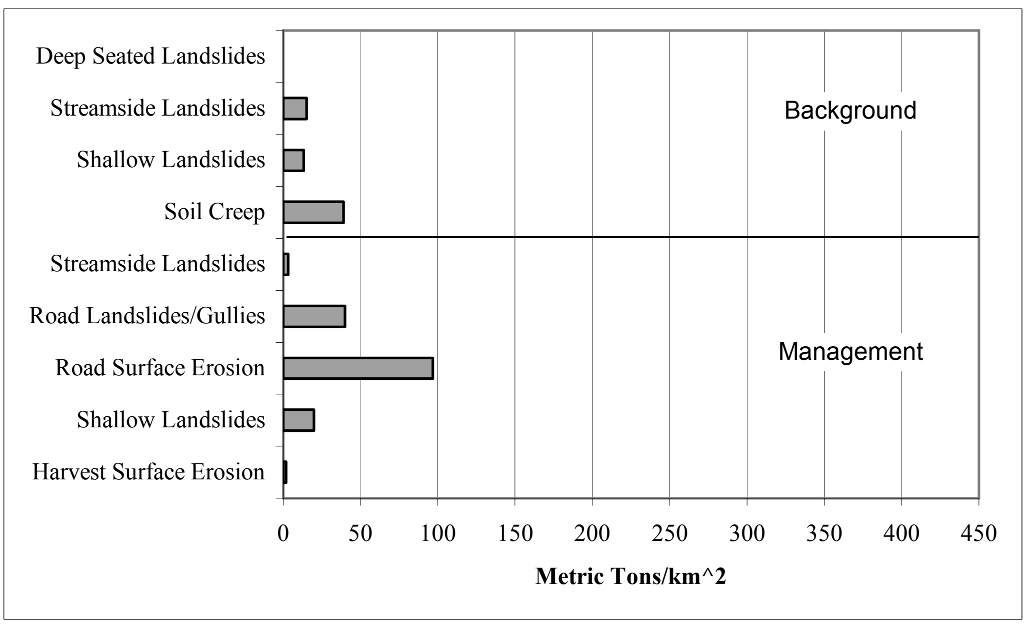


Figure 2—Annual sediment budget for the Freshwater Watershed Analysis Area.

In contrast, the Van Duzen River management-related sediment sources were less than half as much as the natural sediment inputs (441 metric tons/km^2/year versus 1,058 metric tons/km^{2}year). Streamside landslides were, by far, the largest management-related sediment source *(fig. 3)*, most resulting from historic tractor logging in steep, near stream areas. Although management sources were small compared to natural sediment, the total annual input of management sediment was 2.7 times greater than that observed in Freshwater.

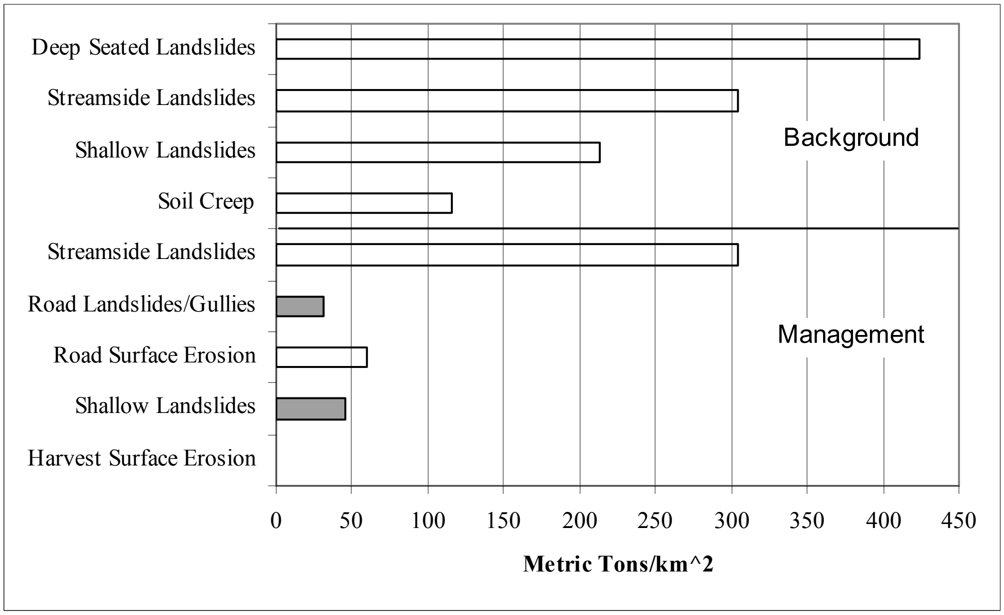


Figure 3—Annual sediment budget for the Van Duzen Watershed Analysis Area.

For the Lower Eel, management-related sediment inputs again dominated the sediment budget (928 metric tons/km^2/year versus 394 metric tons/km^{2}year). All types of management-related sediment were significant sources, except harvest-related surface erosion *(fig. 4)*. In common with the Van Duzen, streamside landslides were the single greatest management-related sediment source. Lower Eel had, by far, the greatest management-related sediment yield, at 2.1 times the yield in the Van Duzen and 5.8 times the yield in Freshwater.

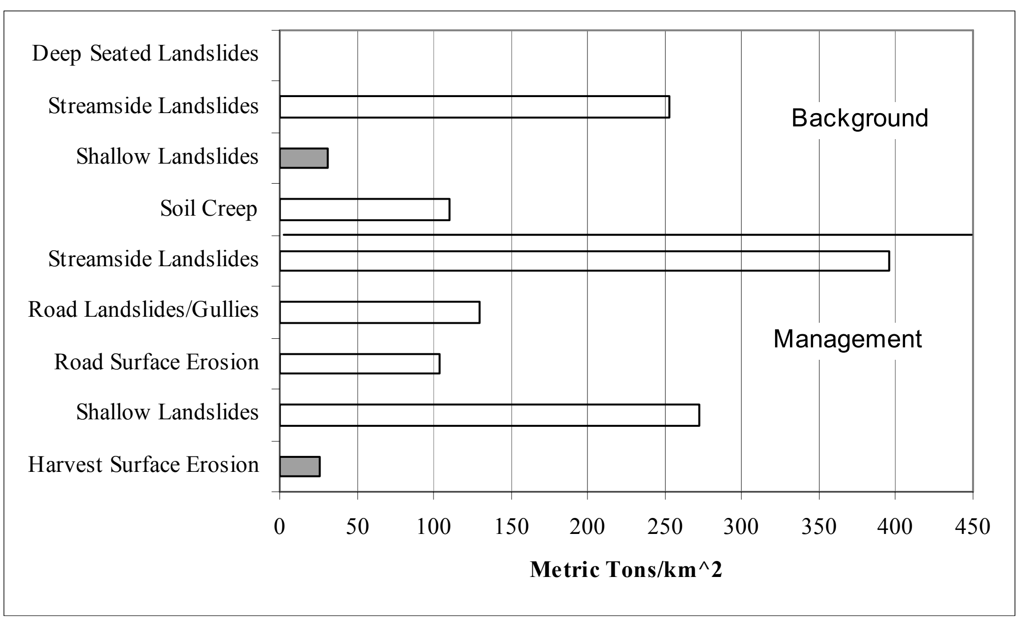


Figure 4—Annual sediment budget for the Lower Eel Watershed Analysis Area.

Riparian Condition

All three watershed analyses found that streamside canopy levels were high regardless of management history, vegetation type, or vegetation age. Canopy levels exceeding the 85 percent PFC level were present along 90, 77 and 72 percent of stream segments in the Freshwater, Van Duzen, and Lower Eel areas, respectively.

Source recruitment curves for large woody debris delivery in the watersheds were similar. In all cases the majority of wood delivery to streams occurred from within 50 feet of the stream's edge *(fig. 5)*. In some cases wood delivery from greater distances was noted, particularly in steep incised features where landslides originating at some distance from streams were capable of delivering wood to the channels. However, such areas were relatively rare, and the effectiveness of any delivery mechanism (for example, landslides, wind throw, bank erosion, and so forth) decreased dramatically with distance.

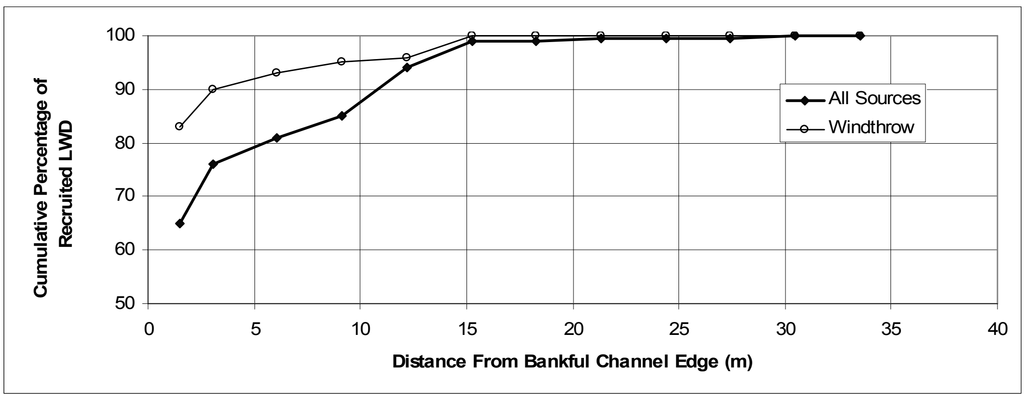


Figure 5—LWD recruitment curve for the Freshwater Watershed Analysis Area.

Development of Management Prescriptions

As noted, PALCO's HCP contains a series of interim conservation measures. Prescription writing used the sediment budgets and large woody debris recruitment relationships to determine whether the interim mass wasting and riparian prescriptions could be modified. For mass wasting, a two-part question guided prescription writing: 1) is mass wasting related sediment an important sediment source; 2) what specific combination of features (slope, distance from streams, stream type, and geomorphic indicators) were associated with that sediment delivery? Ultimately, a three-tiered, empirically derived, risk based approach was developed: Tier 1 = no harvest in the areas with the highest historic mass wasting sediment delivery rates, Tier 2 = required geologic review by a licensed geologist prior to harvest, with a mandatory "floor" that retains 50 percent canopy closure, and Tier 3 = required geologic review by a licensed geologist, but with no mandatory retention standard.

For Freshwater, hillslope related mass wasting was a relatively unimportant sediment source, and so the less restrictive Tier 3 strategy was instituted watershed wide. For road-related mass wasting, continuation of the requirement that roads must be stormproofed to remove at-risk culverts, fill slopes, and so forth, was retained along with requirements to upgrade surfacing and road drainage for all roads used for specific harvesting operations.

For the Van Duzen, sub-basin specific sediment budgets identified three sub-basins (Cummings, Root, and Hely creeks) with relatively higher road and hillslope failures. For these three sub-basins all three tiers of the mass wasting strategy were instituted with the locations and slope triggers of the individual tiers being dictated by the location and relative level of sediment delivery *(table 1)*. On the ground, this strategy was much more site-specific and stream oriented than the bulbous, and qualitative science-based polygons typically delineated by the interim mass wasting strategy of PALCO's HCP *(fig. 6)*.

Table 1—*Percent of total mass wasting related sediment delivery to streams that occurred as a function of stream type, slope, and distance from streams in the Van Duzen Watershed Analysis area.*

Van Duzen Watershed Mass Wasting Sediment Delivery				
	Stream type	**Percent slope**	**Distance from stream**	**Percentage[1]**
1987-1997				
	Class I & II	>40	0 - 30 m	12.2/75
	Class I & II	>40	30 - 61 m	23.5/15
	Class I & II	>40	61 - 91 m	3.3/0
	Class III	>60	0 - 30 m	8.3/0

[1]First number is percentage sediment from hillslope landslides, second number is estimate for streamside landslides.

For the Lower Eel watershed, sub-basin specific sediment budgets demonstrated that management-related mass wasting was significant in all areas, but that the location of at-risk areas differed among sub-basins. Ultimately we modeled sediment delivery to three sub-basins in detail (Bear, Jordan, and Stitz creeks) and then used the results to develop separate mass wasting strategies for different portions of the Lower Eel analysis area. As an example, for the Jordan Creek sub-basin, we again instituted all three tiers with the selection of tier level based on the magnitude of sediment delivery *(table 2)*.

Table 2—*Percent of total mass wasting related sediment delivery to streams that occurred as a function of stream type, slope, and distance from a representative sub-basin (Jordan Creek) in the Lower Eel Watershed Analysis area.*

Lower Eel-Jordan Creek Watershed Mass Wasting Sediment Delivery				
	Stream type	**Percent slope**	**Distance from stream**	**Percentage[1]**
1988-2000				
	Class I & II	>50	0 - 30 m	6.7/70
	Class I & II	>50	30 - 61 m	2.3/5
	Class I & II	>50	61 - 91 m	2.7/5
	Class III	>50	0 - 30 m	3.1/0

[1]First number is percentage sediment from hillslope landslides, second number is estimate for streamside landslides.

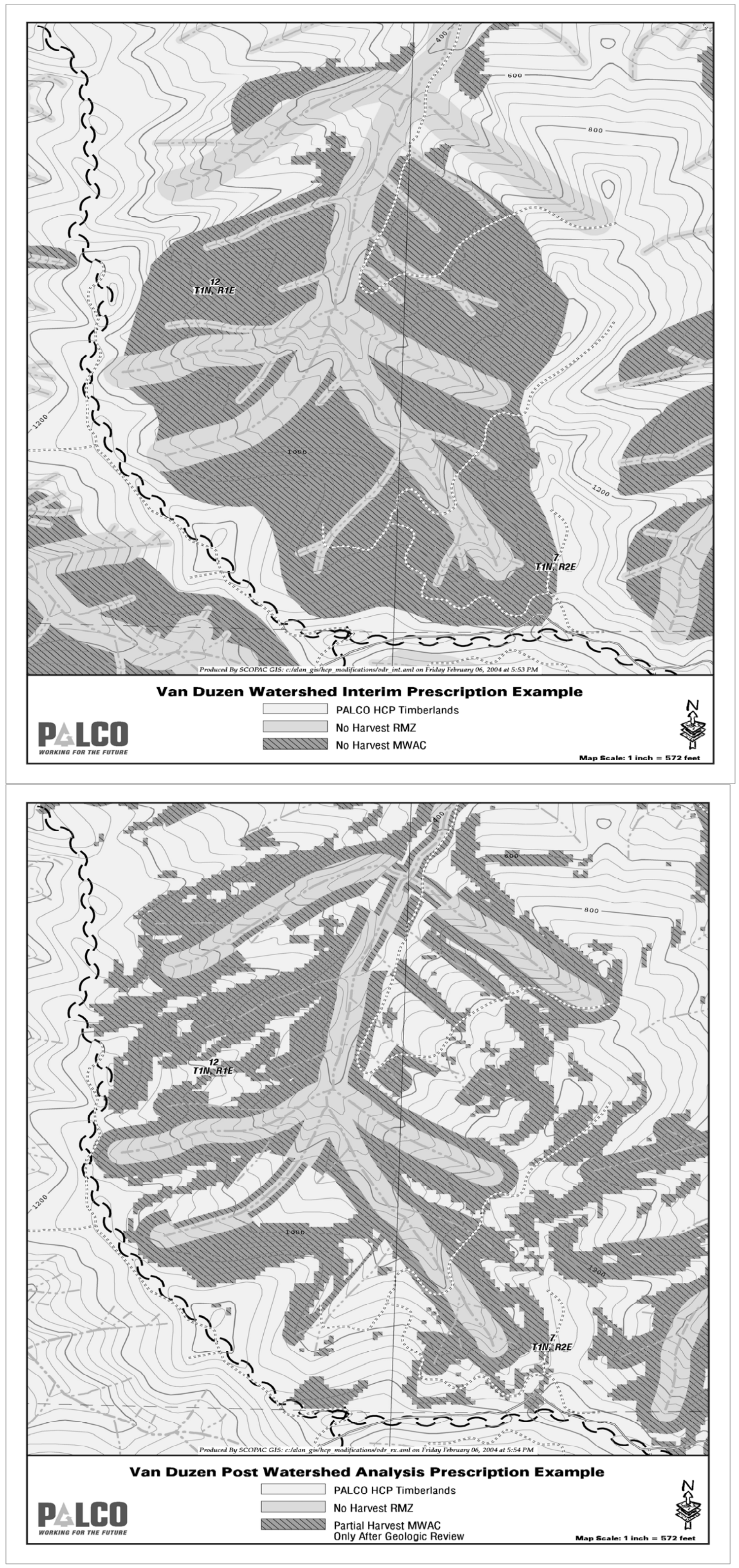


Figure 6—Areas of restricted harvest under the interim (left) and post watershed analysis mass wasting avoidance strategies.

In contrast to management strategies to control mass wasting, which are often site specific, we have found that prescriptions for riparian areas on PALCO's lands can be very similar. Watershed analysis and trend monitoring data demonstrate that high canopy closure conditions dominate virtually all channels, and that all the source-recruitment curves were comparable to that depicted in *figure 5*. The combination of high canopy levels and the near absence of wood recruitment from distances greater than 30m from streams led to significant reductions in the required buffer widths along streams: (*table 3*). These general guidelines were supplemented by site specific prescriptions for locations in need of additional corrective or protective prescriptions. For example, required buffer widths were increased along low gradient (< 4 percent) higher order Class I streams in the Van Duzen due to currently low large wood levels, and the importance of these areas for salmonid spawning.

Table 3a

Interim Riparian Restrictions

	Class I		**Class II**		**Class III**	
Percent Slope	<50	>50	<50	>50	<50	>50
No Cut buffer width (meters, slope distance)	0 - 30	0 - 30	0 - 9	0 - 9	3	3
Outer band Selective Entry Band (meters, slope distance)	30 -52	30 - 52	9 - 40	9 - 40	3 - 9	3 - 9
Total Equipment Exclusion Zone Width (meters, slope distance)	52	52-122	52	52 - 122	15	30

Table 3b—*Riparian Management Strategies following watershed analysis.*

Revised Riparian Restrictions

	Class I			**Class II**			**Class III**		
Percent Slope	<20	20-50	>50	<20	20-50	>50	<20	20-50	>50
No Cut buffer width (meters, slope distance)	0 - 15	0 - 15	0 - 15	0 - 9	0 - 9	0 - 9	0	0	0
Outer band Selective Entry Band (meters, slope distance)	15 - 23	15 - 30	15 - 38	9 - 15	9 - 23	9 - 30	0	0	0
Total Equipment Exclusion Zone Width (meters, slope distance)	23	30	38	15	23	30	15	23	30

Discussion

Operationally and economically, the development of new management strategies for Freshwater, Van Duzen, and Lower Eel led to significant improvements over the interim measures of the HCP *(table 4*) while providing for similar or improved environmental protections. In total, over 2053 hectares and 922,000 cubic meters of timber previously unavailable for harvest under the interim measures were made available following watershed analysis (*table 4*). On average, the relief amounted to 15 to 20 percent of the total land area and standing timber volume in each basin. Of interest is that site-specific prescriptions sometimes put more land and more timber volume under restriction than the interim measures (*table 4*). Thus, although the net direction of our prescriptions was toward greater economic and operational flexibility, the process was also able to increase protection levels above interim standards when site specific conditions warranted it.

Table 4—*Operational and economic improvements achieved through watershed analysis based prescription writing ("new") compared to the original HCP restrictions ("Interim"). Volumes are expressed as cubic meters of timber. Riparian and MW refers to land area and timber volume affected by riparian and mass wasting restrictions, respectively.*

	Interim Riparian	New Riparian	Difference in percent	Interim MW	New MW	Difference in Percent
Freshwater Hectares	2032	986	-51.5	525	592	12.7
Freshwater Volume	470255	232624	-50.5	121159	90918	-25.0
Van Duzen Hectares	1312	1035	-21.1	2200	2638	19.9
Van Duzen Volume	144819	123264	-14.9	212044	194716	-8.2
Lower Eel Hectares	2397	4032	68.2	6919	4675	-32.4
Lower Eel Volume	464565	608166	30.9	987191	228284	-76.9
Total Hectares	5740	6053		9645	7905	
Total Volume	1079639	964055		1320394	513918	

These experiences lead me to conclude that the PALCO HCP's dependence upon an interim set of conservation measures and subsequent watershed analysis studies to develop final management strategies works scientifically, but requires extensive staff time and money by both the landowner and the participating agencies. As a scientist I have been surprised to find how often *a priori* expectations about watershed conditions and functions were wrong, and by how successfully the teams were able to leverage information unique to the watersheds to achieve comparable levels of ecological protection. Both observations are strong endorsement for watershed studies to inform selection of watershed specific management strategies.

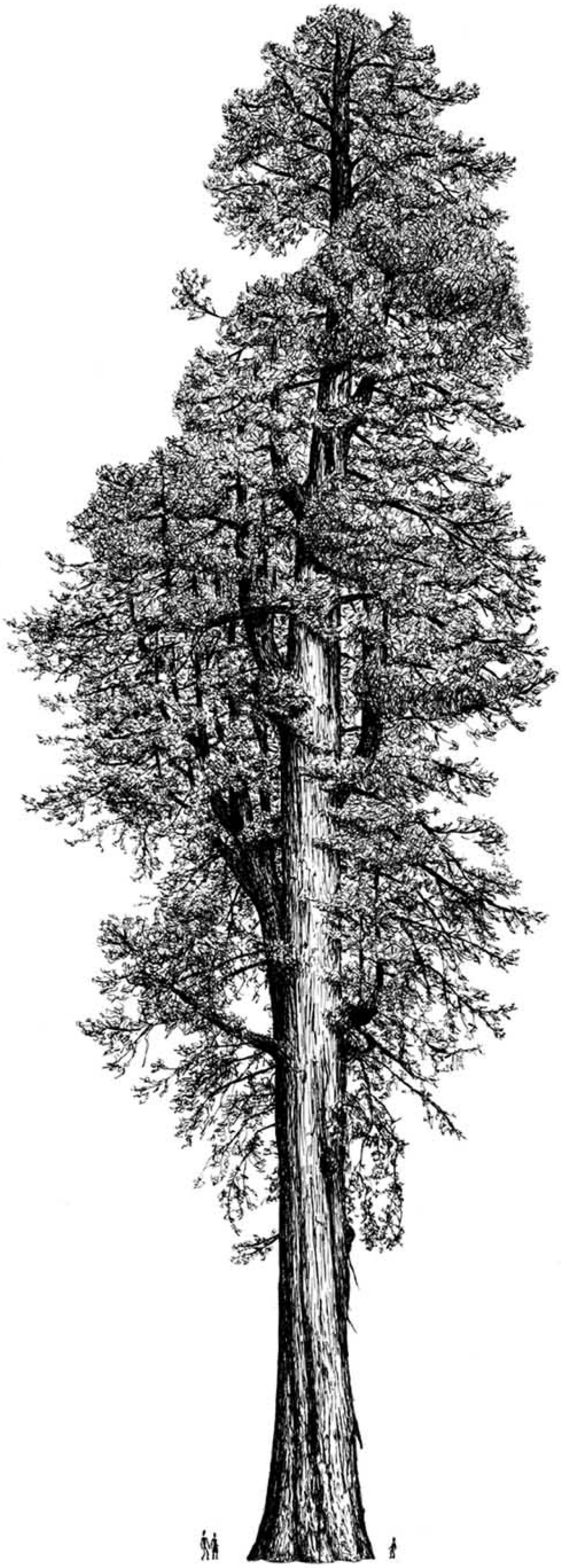
This conclusion is reinforced by the large variation among watersheds in some key variables. For example, large variations in sediment yields and dominant delivery mechanisms were noted in these three watersheds, all of which are within just 20 km of each other. It would have been difficult to develop management strategies that simultaneously provide high levels of protection for the environment, and maximize operational and economic opportunities, without the scientific information we had in hand. However, it is also true that the teams are beginning to see commonalities in results, and in the prescriptive approaches to protecting against management-related impacts. Thus, I expect, and look forward to the prospect that watershed analysis studies and prescription writing for PALCO's lands will continue to get more efficient and less costly as efforts continue to refine and apply a consistent approach.

Conclusions

PALCO's experience demonstrates that industrial timberland owners can maintain operational flexibility and economic viability while simultaneously protecting or enhancing aquatic habitat conditions by leveraging watershed specific information in the development of their land management plans and best management practices. Watershed specific information characterizing current conditions and driving watershed mechanisms allowed the teams to focus environmental protections on dominant local risks to the resources and relax protections associated with regionally identified risks which were not as relevant locally. In addition, when the teams integrated an analysis of costs to the landowner, in terms of acres and timber volume encumbered, they were able to identify those prescriptive measures which provided for the greatest level of environmental protection or enhancement at the least cost to the landowner. Finally, these watershed studies demonstrate that an interdisciplinary team comprised of private landowner and state and federal government representatives can successfully work together in the implementation of a habitat conservation plan which addresses both environmental and economic goals.

SESSION 10

Erosion and Physical Processes II



Sediment Yield From First-Order Streams in Managed Redwood Forests: Effects of Recent Harvests and Legacy Management Practices[1]

M.D. O'Connor,[2] C.H. Perry,[3] and W. McDavitt[2]

Abstract

According to the State of California, most of North Coast's watersheds are impaired by sediment. This study quantified sediment yield from watersheds under different management conditions. Temporary sedimentation basins were installed in 30 randomly chosen first-order streams in two watersheds in Humboldt County, California. Most treatment sites were clearcuts, but two types of clearcut harvest occurred: sites harvested under strict regulations of a Habitat Conservation Plan (HCP) and sites harvested prior to implementation of the HCP. Second-growth stands not recently entered served as the control. Neither geologic substrate nor management were significant predictors of sediment yield. The pre-HCP sites contributed more sediment, but some control sites had sediment yields comparable to these sites. The possible management effect on sediment yield may be influenced by the fact that the HCP sites have experienced only one or two post-harvest winters, while the pre-HCP sites had sediment mobilized in relatively severe winters over a longer post-harvest period. The mean sediment yield from control sites was higher than from HCP sites suggesting that legacy effects of management may be important.

Key words: erosion and sedimentation, forest management, headwater channels

Introduction

Headwater stream channels with ephemeral flow represent a narrow band of fluvial process that occurs on a portion of the landscape dominated by hillslope processes. The proximity of hillslopes (particularly steep ones) to a relatively dense stream channel network creates a zone in which hillslope materials may be readily transferred to the channel network (Swanson and others 1982, Vannote and others 1980). Although channel morphology is recognizable in headwater areas, it is often discontinuous and weakly expressed over channel lengths of up to a hundred meters or more. A discrete channel head may or may not be present. The strength of fluvial process may also vary temporally as well as spatially, so the channel head migrates up or down slope depending on climate and landscape disturbance (Dietrich and Dunne 1993). The size distribution of sediment in these channels reveals weak fluvial

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Principal and Fluvial Geomorphologist, O'Connor Environmental, Inc., PO Box 794, Healdsburg, CA 95448, (707) 431-2810. email: matto@oe-i.com

[3] Assistant Professor, Humboldt State University, Department of Forestry and Watershed Management, 1 Harpst St., Arcata, CA 95521. Current affiliation: Research Soil Scientist, USDA Forest Service, North Central Research Station, 1992 Folwell Ave., St. Paul, MN 55108. email: charleshperry@fs.fed.us

sorting (Dietrich and Dunne 1978) leading to geomorphic classification of these channels as "colluvial" (Montgomery and Buffington 1997). Moving down-slope and increasing contributing area, the fluvial process domain becomes more fully established, manifested by a continuous channel with well-defined channel banks and increased fluvial sorting of sediment.

These areas at the head of the watershed channel network are referred to as zero-order basins. In mountain landscapes where debris flows are the dominant erosion process, fluvial processes in headwater channels are thought to be of little significance in the long-term sediment budget (Benda and Dunne 1987, 1997). These hillslope hollows are evacuated by debris flow and then gradually refill by colluvial processes before failing again (Reneau and Dietrich 1991). Not all mountain landscapes, however, are dominated by debris flow processes. There has been relatively little investigation of morphological or fluvial sediment routing characteristics of colluvial headwater streams (O'Connor 1993). In such areas, the magnitude and frequency of fluvial processes in colluvial headwater channels may be relatively important in long-term sediment routing from low-order basins.

The northern California Coast Range is well known for high sediment yield (Lisle 1990) and deeply weathered soil and bedrock profiles (McLaughlin and others 2000). Although not unknown in the region, debris flow processes are less prevalent than large-scale deep-seated landslides and debris slides on inner gorge slopes. Consequently, the role of colluvial headwater channels in sediment routing and watershed scale sediment budgets may be better characterized by weak fluvial processes than episodic mass wasting processes. In addition, the California Coast Range supports extensive redwood *(Sequoia sempervirens)* forests that have been intensively logged over the last century, and most areas are in their second or third harvest rotation. Early clearcut logging was followed with fire to clear small debris prior to log yarding. Soil disturbance occurred on the surface and in draws and stream channels. Harvest techniques following World War II relied increasingly on bulldozers to yard logs by ground skidding and resulted in extensive soil disturbance to greater depth. Since the State of California began to regulate timber harvest practices in 1974, the permissible extent of disturbance to stream channels has declined.

Under the California Forest Practice Rules, colluvial headwater streams are typically categorized as Class III waters for regulatory purposes. The primary definition of a Class III channel is the ability or potential to transport sediment downstream, consistent with the geomorphic concept of channels. Determination of the headward extent of Class III channels and the boundary with Class II channels (defined as supporting non-fish aquatic life and loosely defined by the extent of intermittent or perennial stream flow) is more problematic and integral to the determination of the extent of Class III channels.

According to present regulations, Class III channels are protected from widespread mechanical disturbance by heavy equipment. In addition, retention of at least 50 percent of the understory vegetation present before timber operations is required (California Department of Forestry and Fire Protection 2004, §916.5). The efficacy of these and related regulatory protections with respect to erosion has been investigated through monitoring programs (Cafferata and Munn 2002) and experimental studies at Caspar Creek (Henry 1998). The Caspar Creek study compared sediment yield from clearcut-logged second-growth and unlogged second-growth redwood forest over a six-year post-logging period (Lewis 1998). Suspended sediment yield increased between 123 and 203 percent in three 10- to 25-ha clearcut watersheds. Observed peak flow increases of about 25 percent for two-year recurrence interval storms (Ziemer 1998) attributed to reduced canopy interception and evapotranspiration in clearcuts may have induced surface erosion and accelerated channel erosion to produce the observed increase in sediment yield (Lewis 1998).

Habitat Conservation Plans (HCPs) to protect endangered aquatic species either in effect or being developed under the authority of the Federal Endangered Species Act include increased regulatory protections for Class III channels in portions of the redwood region. The Pacific Lumber Company (PALCO) has operated under an HCP since 1998 that provides for a minimum 10-m-wide equipment exclusion and limited harvest zone adjacent to Class III channels. It also requires a 3-m-wide no-harvest zone adjacent to each bank of the channel. Given the limited information on headwater colluvial (Class III) channels and the significant harvest restrictions under its HCP, PALCO initiated this pilot study to develop additional data on sediment yield from Class III channels and to test whether HCP regulations accomplished the desired goal of reducing sediment yield.

Methods

Study Design

The erosion processes of interest in this study were surface erosion and channel erosion that could reasonably be modified by the presence or absence of riparian forest cover. We designed our study to exclude the direct influence by roads and landslides that would introduce excess runoff or sediment that would be expected to be of much larger magnitude than the effect on the erosion and runoff processes of interest. We focused particularly on comparing management practices under PALCO's HCP with recent pre-HCP harvest practices. Our fundamental hypothesis was stated as follows:

H_0: There is no effect of geology or management on sediment yield, and

H_a: There is an effect of geology and/or management on sediment yield.

A randomized block design was developed to test our hypothesis. The design included four geologic types (Franciscan units `co1` and `co2`, Yager unit `y1`, and Wildcat unit `Qtw`), and three forest management categories; these include clearcut areas harvested after implementation of the HCP in 2000 and 2001, clearcut areas harvested prior to adoption of the HCP typically between 1994 and 1997, and a group of control sites where no harvest had occurred since at least 1985. We attempted to locate at least five sites in these management categories distributed across each of the four geologic types, for a desired total sample size of sixty sites.

Study Area

This study was conducted on PALCO timberlands in Humboldt County in the lower Van Duzen River and lower Eel River watersheds. These study areas correspond to watershed analysis units developed by PALCO for its HCP. Watersheds drained by mapped Class III channels in the study area are small, with a median contributing area of five ha (n = 350). About 80 percent of Class III streams drains watersheds smaller than 10 ha.

The three most common bedrock formations underlying Class III channels in these areas are the Franciscan, the Wildcat and the Yager (McLaughlin and others 2000). The Franciscan is generally characterized as deformed and sheared sedimentary rocks of Tertiary to Cretaceous age. McLaughlin and others (2000) re-mapped the Franciscan according to topographic criteria and subdivided it into four units. Our study sites in the Franciscan were ultimately located in map unit `co2` which is described as Coastal Belt Franciscan rocks with roughly equal parts sandstone and clay-rich mélange lacking well-incised side hill drainage systems. This unit also comprised the largest acreage of the four subunits of the Franciscan in the study area. The Yager is of similar age and sedimentary composition, and varies widely locally in strength; McLaughlin and others (2000) describe map unit `y1` as relatively well-lithified sedimentary rocks of the Yager terrane. The Wildcat Group is of sedimentary origin, of Pliocene-Miocene age, and is more uniformly weak; it is denoted as map unit `Qtw` and described as weakly lithified sedimentary rock (McLaughlin and others 2000). All of these rocks weather to form soils rich in silt and clay; the Wildcat generally does not produce competent gravel that withstands fluvial transport (Pacific Lumber Company 2003).

Site Selection Procedure

A map of potential field sites was developed from PALCO's geographic information system to identify areas of recent harvest and to exclude sites likely to receive direct runoff from roads. Potential sites were grouped according to geologic types. Field crews were given reconnaissance assignments selected by a random draw. These field visits determined whether a sediment basin could be installed and, if so, confirmed the forest management treatment. When treatment sites were identified, a nearby control site was also identified. Field reconnaissance continued until the necessary number of sites was identified or until the pool of potential sites was exhausted.

We found a large number of sites where sediment basins could not be installed. This was primarily a result of small channel size, discontinuous channels, or extensive woody debris. Some sites were easily accessible by road while others were reached by a combination of ATV and hiking. Ultimately, it was not possible to maintain equal sample size in each of the combinations of geology and management class. Sites in Franciscan map unit `co1` were so few that this geologic type was removed from the study.

Sedimentation Basins

We established sedimentation basins to capture sediment transported in ephemeral colluvial headwater drainages. Sedimentation data was collected over the 2002 and 2003 water years.

A simple sedimentation basin formula using basin surface area and estimated discharge was applied to provide a target for basin dimensions and to impose some uniformity of trapping efficiency (Goldman and others 1986, p 8.15). We expected the basins to retain sediment about 0.5 mm and coarser.

Sedimentation basins were constructed in channels by installing a flashboard dam of marine plywood with a trapezoidal notch to control spilling stream flow. The floor of the sedimentation basin was covered with plastic sheeting in the first year; this was replaced by a porous erosion control fabric for the second year of the study to reduce ponding. These liners provided a sampling surface upon which sediment in transport could settle and be positively distinguished from material on the channel floor and banks.

Most dams were composed of 2.4-m wide plywood with the base of the notch about 0.75 m above the channel floor. The basins typically extended three to four m upstream, but the shape varied considerably. Median basin volume below the spillway elevation (sediment storage capacity) was approximately 0.9 m^3. Five basins had capacities less than 0.5 m^3.

After each winter, field crews measured the total volume of sediment in each basin. Differences in moisture content of sediment were disregarded. The sediment in the basins was removed, and subsamples of sediment were preserved for particle size analysis. To estimate the magnitude of sediment yield including sizes not deposited, we used a technique that compares the probable size distribution of the source material to the size distribution of the deposit (Reid and Dunne 1996, p. 49).

Results

Thirty sedimentation basins were installed prior to runoff producing rainstorms in autumn of 2001. Two sediment basins were directly affected by mass wasting and were subsequently excluded. Two other sites were damaged by runoff and were decommissioned immediately to avoid potential erosion. Two additional sites added to represent recent harvest on lands not subject to HCP requirements were excluded from analysis because of unique bedrock geology. The final data set contained 24 sites with an unbalanced distribution among experimental blocks *(table 1)*. The data were collected at the same sites over a two-year period and analyzed with respect to unit-area sedimentation (L/ha) *(table 2).*

Table 1—*Distribution of sample sites among treatment types.*

	Bedrock geology		
Management	**Franciscan**	**Wildcat**	**Yager**
Control	3	2	3
Pre-HCP	1	1	4
HCP	3	3	4

We used a split-plot treatment of repeated measures (MathSoft 1999) to structure our analysis of variance. Exploratory data analysis revealed that the data were extremely skewed with an overabundance of small sediment yields *(fig. 1)*. We used the Box-Cox function to determine the ideal transformation (Crawley 2002). Box-Cox fits positive and negative exponents, so a non-significant number (0.01) was added to each observation. Given the average magnitude of trapped sediment volume, this small addition should not influence the final result. It also made sense from a geomorphic perspective. We did not expect any observations of absolutely no sediment yield over the course of an entire water year, but we did anticipate sites with yields small enough to be undetectable. The best transformation was the fifth root of unit-area sedimentation, and the transformed data more closely approximated a normal distribution *(fig. 2).*

Table 2—*Summary of collected sediment basin data.*

Site	Mgmt	Geology	Drainage area	Channel length	Sediment volume 2002	Sediment volume 2003	Sediment size[1] d50	Sediment size[1] d84
			ha	*m*	*L*	*L*	*mm*	*mm*
699B	Control	Franciscan	2.5	101	595	889	3.2	10.9
713A	Control	Franciscan	0.8	147	0	575	--	--
717A	Control	Franciscan	0.8	114	0	0	--	--
600A	Control	Wildcat	4.8	56	11	114	2.0	8.0
702D	Control	Wildcat	0.4	37	0	11	--	--
463A	Control	Yager	1.9	170	0	8	--	--
470A	Control	Yager	1.0	78	22	208	3.7	11.5
721A	Control	Yager	1.0	15	175	238	4.0	13.0
699	HCP	Franciscan	1.9	116	114	131	--	--
713	HCP	Franciscan	1.3	52	0	168	--	--
717	HCP	Franciscan	3.6	324	42	132	0.2	0.6
571	HCP	Wildcat	0.8	61	0	0	--	--
600	HCP	Wildcat	3.6	472	224	265	0.2	0.9
702	HCP	Wildcat	4.8	300	55	163	1.5	13.0
463	HCP	Yager	7.5	392	7	17	4.7	11.5
470	HCP	Yager	10.9	140	65	45	3.0	15.0
721	HCP	Yager	2.1	71	0	61	--	--
734	HCP	Yager	1.9	154	36	178	0.1	0.5
460	Pre-HCP	Franciscan	1.3	129	735	265	0.1	0.9
798	Pre-HCP	Wildcat	2.5	196	8	112	1.6	5.2
781	Pre-HCP	Yager	2.9	165	962	2,120	0.2	0.9
795	Pre-HCP	Yager	0.8	41	4	30	0.4	1.5
821	Pre-HCP	Yager	1.5	152	256	606	5.0	13.0
823	Pre-HCP	Yager	1.5	129	21	360	0.1	0.2

[1]Particle size analysis of trapped sediment was completed only for the 2002 water year.

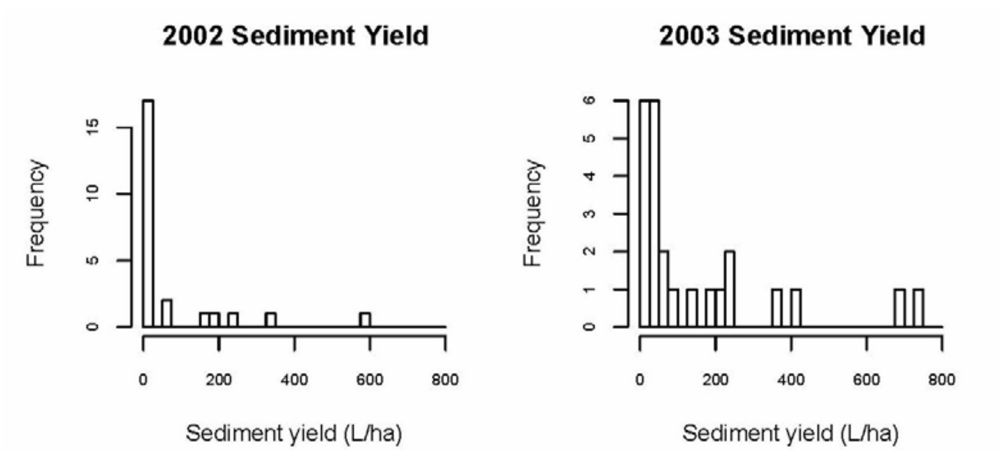


Figure 1—Observed sediment yield (basin sedimentation) was highly skewed. There were more zeroes observed in the dry water year of 2002.

Figure 2—A 1/5-root transformation created a nearly normal distribution of sediment yield (basin sedimentation). A small, non-significant number (0.01) was added to every observation to facilitate the transformation.

Graphical assessment of the central tendency and dispersion of the data suggested there was tremendous variation in sediment yields in Class III streams *(figs. 3, 4* and *5).*

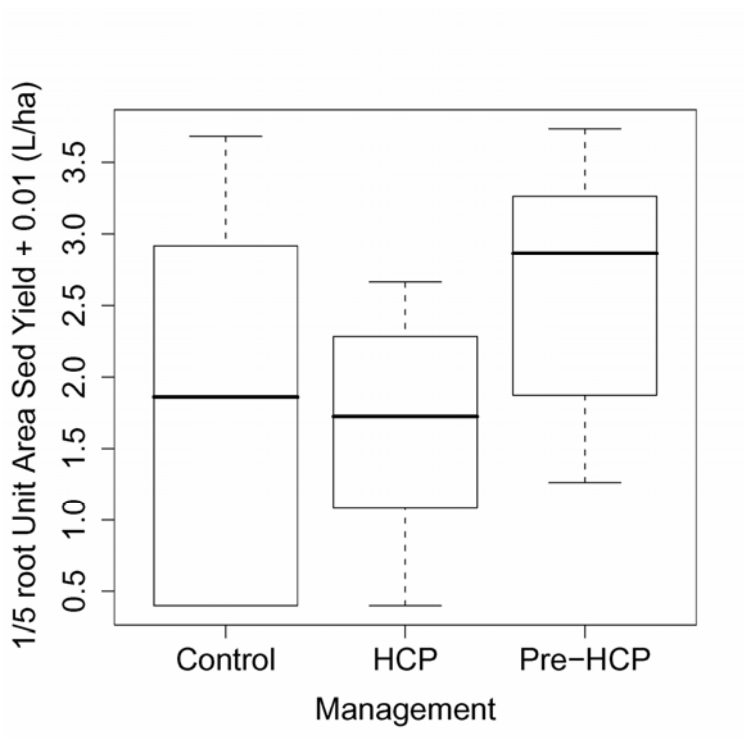


Figure 3—Box plot of the management predictor.

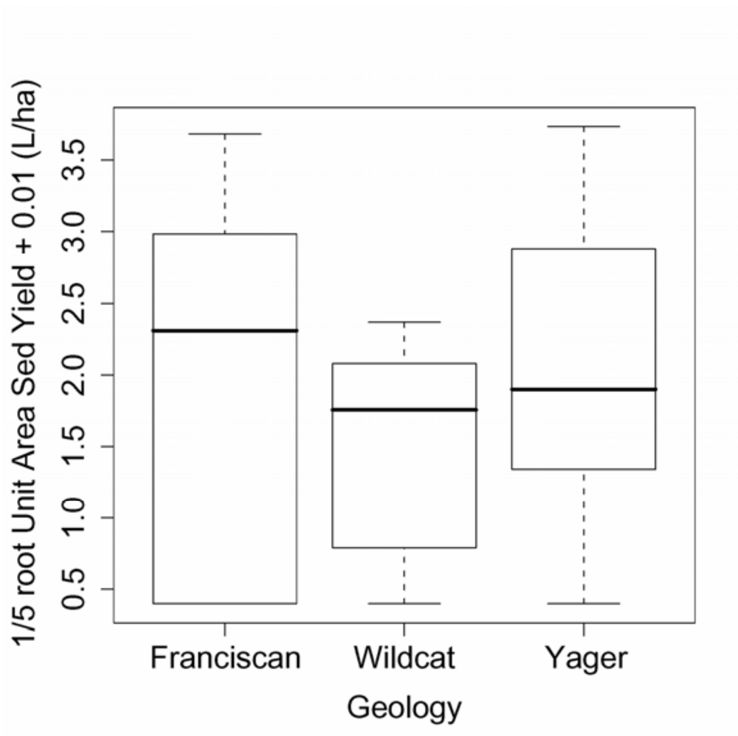


Figure 4—Box plot of the geology predictor.

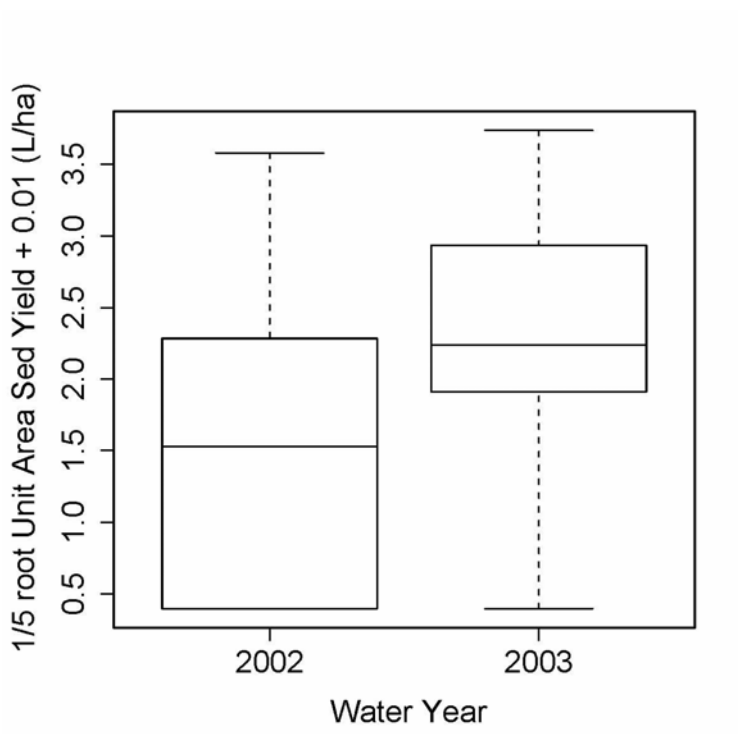


Figure 5—Box plot of the predictor water year.

An analysis of variance model was fit to test the effects of management, geology, and water year on unit-area sediment yield:

```
model <-     aov((Sed.yield+0.01)^0.2
             ~ Mgmt*Geology*Year + Error(Site))
```

The `Error(Site)` term accounted for the temporal psuedoreplication in our observations (MathSoft 1999). Time was the only significant predictor in this model *(table 3)*. Diagnostic plots of the model suggested that sites in Franciscan (`co2`) geology were the most likely to be outliers. Removing this geology from the analysis did not alter the result.

Table 3—*Analysis of variance (including repeated measures) for sediment yield in Class III streams.*

Error: Site	df	Sum Sq	Mean Sq	F value	Pr(>F)
Mgmt	2	7.6654	3.8327	2.6618	0.1025
Geology	2	2.2431	1.1215	0.7789	0.4766
Mgmt:Geology	4	2.1362	0.5340	0.3709	0.8257
Residuals	15	21.5988	1.4399		

Error: Within	df	Sum Sq	Mean Sq	F value	Pr(>F)
Time	1	5.8530	5.8530	14.2147	0.0018
Mgmt:Time	2	0.5163	0.2582	0.6269	0.5476
Geology:Time	2	0.0458	0.0229	0.0556	0.9461
Mgmt:Geology:Time	4	1.4453	0.3613	0.8775	0.5004
Residuals	15	6.1763	0.4118		

Attempts to analyze our data with generalized linear models were hampered by the small sample size and the unbalanced design. We could not fit a complete model of management, geology, and time with all of the interaction terms. Tests of management and geology alone affirmed that they were non-significant predictors ($p = 0.3846$ and $p = 0.8577$, respectively). We fit a simple additive model without any interactions by assuming that the non-significant management:geology interaction would not become significant with the inclusion of time in the model. Testing this model by deletion (Crawley 2002) also suggested that time was the only significant predictor ($p = 0.0425$) of unit-area sediment yield in these Class III streams.

Sediment trapping efficiency likely varied among sediment basins of different size and geometry, as well as at different levels of sedimentation at individual sites. In the first year, 12.5 percent of sites had sedimentation greater than one-third of estimated capacity. In the second year, this was true for 33 percent of the sites. Sedimentation equaled or exceeded estimated storage capacity at three sites in the first year and at two sites in the second year. Consequently, our observations were right-censored, and the measured unit-area sedimentation in some cases is only a lower limit. Therefore, it may be more appropriate to analyze these data using nonparametric statistical tests that would be less sensitive to the structure of our data.

The observed variances in unit-area sediment yield were not constant between the different management categories or geologic formations, so we performed the Kruskal-Wallis One-Way Analysis of Variance on Ranks on each of the two years of data separately. We tested the geology and management classes in separate analyses *(tables 4* and *5).* These nonparametric analyses were consistent with the parametric analyses; no significant differences were found among management or geologic groups.

Table 4—*Kruskal-Wallis One-Way Analysis of Variance on Ranks on transformed data for 2002 observations.*

Group	N	Missing	Median	25 percent	75 percent
Control	8	0	0.789	0.398	2.309
HCP	10	0	1.529	0.398	1.809
pre-HCP	6	0	2.258	1.364	3.190

H = 3.876 with 2 degrees of freedom (p = 0.144).

Group	N	Missing	Median	25 percent	75 percent
Franciscan	7	0	1.638	0.398	2.808
Wildcat	6	0	1.221	0.398	1.628
Yager	11	0	1.705	1.082	2.545

H = 1.233 with 2 degrees of freedom (p = 0.540).

Table 5—*Kruskal-Wallis One-Way Analysis of Variance on Ranks on transformed data for 2003 observations.*

Group	N	Missing	Median	25 percent	75 percent
Control	8	0	2.407	1.611	3.096
HCP	10	0	2.041	1.329	2.368
pre-HCP	6	0	2.964	2.138	3.341

H = 5.067 with 2 degrees of freedom (p = 0.079).

Group	N	Missing	Median	25 percent	75 percent
Franciscan	7	0	2.663	2.130	3.156
Wildcat	6	0	1.979	1.883	2.138
Yager	11	0	2.490	1.495	2.997

H = 3.064 with 2 degrees of freedom (p = 0.216).

Discussion

Our data suggested that among the potential factors controlling sediment yield in the Class III watersheds that we studied—management, geology, and water year—only water year served as a significant predictor. Specifically, there was significantly greater sediment yield during Water Year 2003 than during Water Year 2002.

These two water years had differences in measured precipitation. Using data from the California Data Exchange Cooperative (http://cdec.water.ca.gov) and precipitation stations from Scotia and Eureka Woodley Island, WY 2002 was near-normal. Scotia, located within about 10 miles of the study area, reported about 90 percent of average and Eureka 105 percent of average. Abundant rainfall occurred in November and December 2001 at both stations; at Scotia, rainfall in each month was about 150 percent of average. WY 2003 was wetter than average, with Scotia reporting 137 percent of average and Eureka 142 percent of average. December 2002 was exceptionally wet. Eureka and Scotia were far above average for the month, 364 percent and 316 percent respectively. Scotia had nearly 700 mm of rain (over 27 inches) in that month. Rainfall would correlate with streamflow, sediment transport capacity, and channel erosion potential, as well as potential surface erosion processes.

The sediment yield data did not reveal significant differences among either management or geologic substrate, suggesting that for purposes of estimating minimum sediment yield from these Class III streams, a grand mean of the data set provides the best summary statistic. The statistical power of our analysis was low ($\beta < 0.2$) compared to the typical desired power ($\beta > 0.8$). Given the relatively small sample size and high variance of the data, this in not a surprising outcome. Sample size calculations suggest that between about 25 and 125 samples would be required in each group to attain $\beta > 0.8$ when $\alpha < 0.05$.

The degree to which sediment yields may be larger than observed sedimentation depends on the size distribution of eroded sediment and the trapping efficiency of the basins. Considering the fact that several basins contained a high proportion of fine sand (0.25 to 0.125 mm) *(table 2),* there is cause to believe that the basins were relatively efficient sediment traps.

Size distribution of soils derived from Wildcat and Franciscan parent material in the region were obtained from other studies (PALCO 2003) to estimate annual sediment yield at each of the sediment basins. In the Franciscan, about 23 percent of soil material was coarser than two mm, whereas in the Wildcat, only eight percent of material was coarser than two mm. We assumed that the sedimentation basins captured all material coarser than two mm; the average median grain size in deposits was 1.9 mm. Using the mean volume of sediment deposited in 2002 (0.139 m^3), the mean volume of eroded soil would be 0.139 m^3 / 0.23 = 0.60 m^3 for Franciscan and 0.139 m^3 / 0.08 = 1.70 m^3 for Wildcat. Using an estimate of the original soil density of 830 kg/m^3 (PALCO 2003), and the mean watershed area of 3.3 ha, estimated annual sediment yield for these sites in 2002 would be range from 150 to 440 kg/ha. Alternatively, we estimated sediment yield using the same technique for each site individually. We assumed that each basin captured all sizes coarser than the median size and that the Yager soil size distribution to be the mean of the Franciscan and Wildcat size distributions. Using this approach, we found the mean yield to be about 320 kg/ha. Either approach is likely to overestimate yield because the sedimentation basins are probably more efficient sediment traps than assumed for the calculation, and the estimate of sediment yield would decrease with increasing trap efficiency. Although this technique is imprecise, it produces an estimate of maximum likely sediment yield. The upper range of sediment yield could be constrained considerably by conducting additional studies on basin trapping efficiency.

The estimated range of mean sediment yield of 150 to 440 kg/ha for the Class III watersheds in this study can be compared to sediment yields measured in larger Class II watersheds at Caspar Creek (Lewis 1998). The clearcut drainages studied at Caspar Creek (BAN, CAR, EAG, GIB, and KJE) had mean drainage area of 20 ha, and the mean annual sediment yield post-harvest was 440 kg/ha. This comparison suggests that the Class III channels observed in this study produced sediment at rates not greater than observed for Class II channels in clearcut watersheds at Caspar Creek. The comparison also suggests that the use of sedimentation basins for studies of sediment yield produces results consistent with prior studies, even considering uncertainty in trapping efficiency.

Conclusions

This study was motivated by concerns that erosion processes in Class III channels (low-order headwater stream channels) could be sensitive to forest management activities, and it evaluated the efficacy of conservation practices to minimize disturbance to such channels that could affect downstream water quality. This study confirmed that studies in rugged, previously disturbed and heavily vegetated terrain is challenging. We were not successful in developing the desired distribution of sample sites, and this hampered quantitative evaluation of the data.

We did not find statistically significant differences in sedimentation between sites with differing geologic substrates and management histories. Although HCP sites did tend to have lower sedimentation than pre-HCP sites, high variance in the data and low statistical power (β) prevented us from concluding whether HCP management practices are more effective at preventing erosion and sedimentation. We also found that there was no correlation between measures of channel conditions and ground disturbance near streams and observed sedimentation. This suggested that the erosion processes responsible for observed sedimentation were of a dispersed nature and/or operated at a relatively small scale.

We estimated likely sediment yields based on sedimentation data for these types of channels, and we found that they were of similar magnitude to measured sediment yields for substantially larger clear cut drainages in Caspar Creek. Our estimation technique was likely to overestimate sediment yield, and it is likely that sediment yield from these small Class III watersheds was in fact lower than that in the larger Class II watersheds. These quantitative comparisons suggested that erosion and sedimentation processes in Class III watersheds were not strongly differentiated in magnitude or process from somewhat larger Class II watersheds.

References

Benda, L.; Dunne, T. 1987. **Sediment routing by debris flow.** In: Beschta, R.L.; Blinn, T.; Grant, G.E.; Ice, G.G.; Swanson, F.J., eds. Proceedings of the symposium on erosion and sedimentation in the Pacific Rim; 1987 August 3-7; Corvallis, Oregon. International Association of Hydrological Sciences Publication No. 165. Wallingford, Oxfordshire, UK: International Association of Hydrological Sciences; 213–223.

Benda, L.; Dunne, T. 1997. **Stochastic forcing of sediment routing and storage in channel networks.** Water Resources Research 33(12): 2865-2880.

California Department of Forestry and Fire Protection. 2004. **California Forest Practice Rules.** Title 14, California Code of Regulations; Chapters 4, 4.5 and 10. Sacramento: California Department of Forestry and Fire Protection.

Cafferata, P.; Munn, J. 2002. **Hillslope monitoring program: monitoring results from 1996 through 2001.** Sacramento, CA: Monitoring Study Group, California State Board of Forestry and Fire Protection; 128 p.

Crawley, M.J. 2002. **Statistical computing.** Chichester: Wiley; 772 p.

Dietrich, W.E.; Dunne, T. 1978. **Sediment budget for a small catchment in mountainous terrain.** Zeitschrift für Geomorphologie, Suppl. 29: 191-206.

Dietrich, W.E.; Dunne, T. 1993. **The channel head.** In: Bevin, K.; Kirkby, M., editors. Channel network hydrology. Chichester: John Wiley & Sons; 175-219.

Goldman, S.J.; Jackson, K.; Bursztynsky, T.A. 1986. **Erosion and sediment control handbook.** New York: McGraw-Hill; 443 p.

Henry, N. 1998. **Overview of the Caspar Creek watershed study.** In: Ziemer, R.R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story; 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 1-9.

Lewis, J. 1998. **Evaluating the impacts of logging activities on erosion and sediment transport in the Caspar Creek watersheds.** In: Ziemer, R.R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story; 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 55-69.

Lisle, T.E. 1990. **The Eel River, northwestern California: high sediment yields from a dynamic landscape.** In: Wolman, M.G.; Riggs, H.C., editors. Surface Water Hydrology, v. O-1, The Geology of North America. Boulder, CO: Geological Society of America; 311-314.

MathSoft. 1999. **S-Plus 2000 Guide to Statistics.** Volume 1. Seattle: MathSoft; 722 p.

McLaughlin, R.J., Ellen, S.D.; Blake, Jr., M.C.; Jayko, A.S.; Irwin, W.P.; Aalto, K.R.; Carver, G.A.; Clarke, Jr., S.H. 2000. Barnes, J.B.; Cecil, J.D.; Cyr, K.A., database editors. **Geology of the Cape Mendocino, Eureka, Garberville, and southwestern part of the Hayfork 30 x 60 minute quadrangles and adjacent offshore area, northern California.** U.S. Geological Survey Miscellaneous Field Studies MF 2336. *Available at* http://geopubs.wr.usgs.gov/map-mf/mf2336/ (accessed 2004 May 13; verified 2005 October 25). U.S. Geological Survey.

Montgomery, D.R.; Buffington, J.M. 1997. **Channel-reach morphology in mountain drainage basins.** Geological Society of America Bulletin 109(5): 596-611.

O'Connor, M.D. 1993. **Bedload transport processes in steep tributary streams, Olympic Peninsula, Washington, U.S.A.** Advances in Hydro-Science and Engineering, Vol. 1, Part A. Wang, S.S.Y., ed. Oxford, MS: University of Mississippi; 243-250.

Pacific Lumber Company. 2003. **Freshwater Creek Watershed Analysis Final Report. Appendix E-Stream Channels.** Scotia, CA: Pacific Lumber Company.

Reneau, S.L.; Dietrich, W.E. 1991. **Erosion rates in the southern Oregon Coast Range: evidence for an equilibrium between hillslope erosion and sediment yield.** Earth Surface Processes and Landforms 16: 307-322.

Reid, L.M.; Dunne, T. 1996. **Rapid evaluation of sediment budgets.** Reiskirchen, Germany: Catena; 164 p.

Swanson, F.J.; Fredriksen, R.L.; McCorrison, F.M. 1982. **Material transfer in a western Oregon forested watershed.** In: Edmonds, R.L., editor. Analysis of coniferous forest ecosystems in the Western United States. US/IBP Synthesis Ser 14. Stroudsburg, PA: Hutchinson Ross Publishing; 233-266.

Vannote, R.L.; Minshall, G.W.; Cummins, K.W.; Sedell, J.R.; Cushing, C.E. 1980. **The river continuum concept.** Canadian Journal of Fisheries and Aquatic Sciences 37(1): 130-137.

Ziemer, R.R. 1998. **Flooding and stormflows.** In: Ziemer, R.R., technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story; 1998 May 6; Ukiah, CA. Gen. Tech. Rep. PSW-GTR-168. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 15-24.

Statistical Analysis of Streambed Sediment Grain Size Distributions: Implications for Environmental Management and Regulatory Policy[1]

Brenda Rosser[2] and Matt O'Connor[2]

Abstract

Fish habitat in cold water streams in many northwestern California watersheds has been declared degraded under provisions of the Federal Clean Water Act, contributing to listings of anadromous fish species under the Endangered Species Act. It is believed that past and present land management activities induce erosion that contributes excess sand-size and finer sediment to stream systems, which then causes an increase in the proportion of fine sediment in spawning gravels. The higher proportion of fine sediment can reduce the rate of survival of eggs. Target thresholds for desirable fine sediment concentrations in spawning beds have been identified based on scientific literature and watershed studies. There are few data describing natural or unimpaired sediment size distributions. This is of concern in the region owing to high natural erosion rates.

This study examines data from gravel bed streams collected by McNeil sampling and bulk sediment sampling in northern California (samples typically 25 to 30 kg) and New Zealand (samples typically 50 kg). The McNeil streambed sampling protocol is the preferred method to determine sediment size distributions and stream substrate quality for salmonids in the fisheries literature. Confidence intervals for various percentiles of the grain size distributions were computed from field data using a two-stage sampling approach. Accuracy and precision of data from these sampling programs are considered in relation to the biological/regulatory thresholds as well as the effort required to obtain, process and analyze grain size distributions. Either very large samples, and/or a large number of samples are typically required to obtain data with high precision, suggesting that in many circumstances, it may be difficult to assess whether regulatory thresholds are exceeded.

Key words: fish habitat, sediment sampling, spawning gravel, statistical analysis

Introduction

One of the factors contributing to the decline in salmonid populations in western North America is the impairment of spawning habitat. A primary factor determining spawning habitat availability and quality is determined by the quantity and distribution of suitably sized gravels (Kondolf 2000). One cause of impairment of spawning habitat is sedimentation by sand and fine gravel, from erosion associated with land management. These sediments may be deposited on and mix with

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] O'Connor Environmental, Inc., P.O. Box 794, Healdsburg, CA, (707) 431-2810. email: brendar@oe-i.com and matto@oe-i.com, respectively. website: http://www.oe-i.com

streambed gravels, and can reduce survival of eggs deposited in redds by reducing the flow of oxygenated water and increasing the concentration of metabolic waste. In addition, fine gravel (~ 2 to 8 mm) may impede emergence of salmonid fry from the redds by blocking gravel interstices. The proportion of fine material in the gravel substrate has been linked to the rate of survival of salmonid species (Kondolf and Wolman 1993). Research on the effect of fine sediment (less than about one mm diameter) on incubation of salmonid eggs in redds suggests that spawning gravels are in good condition if fine sediment (<1 mm) comprises less than 12 to 14 percent of spawning bed material (Chapman 1988, Kondolf 2000). Similarly, previous research suggests that emergence of fry will not be significantly reduced when fine sediment less than ~3 mm to 6 mm comprises roughly 30 percent or less of spawning gravels (Kondolf 2000).

Regulatory thresholds have been set for percent fines through the U.S. Environmental Protection Agency's Total Maximum Daily Load's (TMDL) process. Targets are set based on literature review of optimal spawning gravel conditions and limited sampling of existing conditions in the watershed. The USEPA sediment TMDL studies for the Navarro and Garcia watersheds in northern California, set target values for percent fines <0.85 mm at 14 percent, and <6.35 mm at 30 percent, and bed material median particle size (D50) at 37 to 69 mm (min & mean) (USEPA 1998, USEPA 2000).

Another regulatory process, implementation of the Pacific Lumber Company Habitat Conservation Plan (HCP) in northern California, includes a set of targets for sediment conditions pertaining to spawning habitat. The fine sediment target in the HCP is 11 to 16 percent sediment <0.85 mm, and 20 to 25 percent sediment <6.35 mm.

This study compares results from three different particle size sampling programs, each representing a different level of sampling intensity. Confidence limits for various particle size fractions are compared for the three levels of sampling intensity. The sampling programs were conducted independently of each other, at different times and with different purposes in mind, however, the particle size statistics obtained serve to illustrate various levels of accuracy and precision that may be achieved with different levels of sampling effort, as well as the implications of statistical and sampling issues for regulatory programs.

Study Areas

Particle size data were available for three study areas including the Waipaoa River in New Zealand; Soda, Carneros, and Sulphur Creeks in the Napa River watershed, and Green Valley and Salt Hollow Creeks in the Russian River watershed (the North Bay region in northern California); and Freshwater Creek in Humboldt County. All study reaches are gravel-bed streams. The study sites in northern California are known to support salmonid populations, including coho salmon and/or steelhead trout. Green Valley Creek contains one of the few remaining coho salmon populations in the Russian River. Freshwater Creek supports coho and Chinook salmon as well as steelhead and cut-throat trout. The Napa streams are believed to support steelhead. Study reach characteristics are summarized in *table 1.*

Table 1—*Summary of study reach characteristics.*

	Waipaoa River	Napa/Sonoma	Freshwater Creek
Location	East Cape, New Zealand	Northern California	Northern California
Watershed scale	Large – 100 km	Moderate – 2-10 km	Small – 1000 ft
Study objective	Scientific investigation	Spawning gravel study	Aquatic trends monitoring (HCP)
Sampling type	Bulk sediment samples	McNeil samples	Shovel samples
Sampling position	Exposed point bars	Pool tail outs	Riffles
Average subsample size	50 kg	25 kg	5 kg
Number of subsamples	102	22	27
Average D50	4 mm	24 mm	26 mm

Geology of the Waipaoa River, East Cape, New Zealand

The Waipaoa River is a 104 km long gravel-bed river located on the East Cape of New Zealand's North Island. The Waipaoa River is not a fish-bearing stream, however, it was included in this study as an example of an extensive particle size sampling effort—the spatial resolution and extent of the sampling program are unique for a field situation. The Waipaoa River basin is situated within a zone of active deformation associated with the Hikurangi subduction margin (Moore and Mazengarb 1992), which has induced high uplift rates in the ranges at the head of the basin. Sediment supply rates to the river are high; annual suspended sediment yield is 6750 t/km/yr (Hicks and others 2000) and bedload is estimated at one percent of this value (Trafford 1998). The Waipaoa River and its tributaries have been aggrading in response to increased sediment supply rates since the beginning of the twentieth century, when an intense phase of erosion was initiated in the upper watershed following large-scale deforestation, which destabilized the landscape.

Geology of North Bay Region Watersheds, Northern California

Napa Valley and Salt Hollow creek basins are underlain by the Central Belt Franciscan complex. Here, the Franciscan complex includes a mélange of sheared shale and sandstone with some chert, high-grade metamorphic rocks, shattered sandstones and greenstones, and serpentinite. The Franciscan assemblage supports high topography and steep hillslopes, and is notorious for its erosion potential, hence, landslides and other mass movement events are important geomorphic agents in these watersheds. The Franciscan formation is highly fractured and sheared, thus providing a source of fine sediment to the channel when failures occur (Pearce and others 2000). The uppermost headwaters of Carneros Creek, portions of Sulphur Creek and the majority of Soda Creek are underlain by Tertiary Sonoma volcanics, which consists mainly of andesitic and rhyolitic lava flows. Green Valley Creek is located in the coastal range, and is underlain by Coastal Belt Franciscan rocks, which tend to be more stable than Central belt Franciscan rocks. The Napa study areas have drainage basins less than 25 km^2, and Green Valley Creek has a drainage area of 44 km^2. The study site on an unnamed tributary of Salt Hollow Creek has a drainage area of about 1.5 km^2, and is located below on-stream reservoirs.

Geology of Freshwater Creek, Northern California

Freshwater Creek is an 80 km^2 drainage basin located near Eureka, California in Humboldt County. Approximately 77 percent (62 km^2) of the watershed, is owned and managed for timber production by the Pacific Lumber Company (PALCO). The underlying geology present within Freshwater Creek consists primarily of three groups: the Wildcat Group, the Franciscan Central Belt Group, and the Yager Formation (PALCO 2003). The Wildcat Group is composed mostly of mudstone, siltstone, claystone, fine-grained sandstone, and minor conglomerate. The Wildcat formation rocks are inherently erodible and potentially unstable. Gravels in the streambed that are derived from the Wildcat formation are typically very soft and can be broken very easily, and tend to weather quickly into fine particle sizes once in the stream (PALCO 2003). Rates of hillslope erosion and downcutting by major streams within the soft rocks of the Wildcat Formation are geologically rapid (PALCO 2003).

The Franciscan Central Belt is composed of metasedimentary rocks that consist of a matrix of fine sediments with included blocks of harder metamorphic rocks (PALCO 2003). Like the Wildcat Group, this group weathers rapidly to sand, silt, and clay; however, it has a higher fraction of larger rocks that weather more slowly. The Yager Complex consists of dark gray indurated mudstones, shales, graywackes, siltstones, and conglomerates, with interbedded limey siltstones. Rocks from the Yager Formation are much harder and generate larger classes of gravel and cobble (PALCO 2003).

Particle Size Sampling

A two-stage sampling approach was used which involved collecting several subsamples per reach and determining statistics for the pooled sediment sample data (Bunte and Abt 2001). Bunte and Abt (2001) recommend collecting bed material samples that are large enough so that the mass of the largest particle in each subsample is generally less than three percent of the total sample mass. The less than three percent criterion was not achieved for all samples (particularly the Freshwater samples).

In the Waipaoa River, bulk sediment samples were obtained from exposed bars, near the channel centerline (Rosser 1997). The surface and subsurface sediment populations were sampled separately. Sample weights for the Waipaoa River samples were typically >50 kg, and the weight of the largest particle was typically less than two percent of the total sample weight. A total of 104 subsurface sediment samples were obtained at 1 km intervals along the mainstem, which included seven reaches within the river. Each study reach represented a different channel morphology, where the river has responded to different types and rates of sediment supply and sediment transport processes.

All particle size samples from the North Bay streams were obtained using a McNeil sampler, and applying a modified version of the method described by McNeil and Ahnell (1964). Under laboratory conditions, Schuett-Hames and others (1996) found the McNeil sampler to produce samples that most closely represented the composition of test gravel mixtures based on comparison of particles by individual size classes. In the North Bay study reaches, samples were taken from the thalweg in pool tail out positions and were intended to represent potential salmonid spawning sites. Research has shown that female salmonids appear to select spawning sites where flow conditions within the gravel are favorable for the successful incubation of eggs and alevin (Schuett-Hames and Pleus 1996). Favorable subsurface flow conditions exist where surface water infiltrates into the bed material, and is enhanced by a combination of surface water velocity, convex bed profile (such as in pool tail outs and riffle crests) and high gravel permeability (Vaux 1962). Bed material samples included the surface particles as well as subsurface particles. Sample weights from the North Bay region streams were typically 20 to 30 kg; the largest particle within each sample was generally three to 10 percent of the total sample weight.

Bed material samples from Freshwater Creek were obtained using a shovel technique. Data were provided by PALCO. Samples were obtained from the thalweg in pool tail crest positions within the channel. Individual subsample weights from Freshwater Creek were typically five to 15 kg. The particle size statistics were obtained by pooling the statistics from the three subsamples per reach. Schuett-Hames and others (1996) found that shovel based sampling techniques produced samples similar to the McNeil sampler, and that McNeil and shovel samples did not differ significantly in composition. Samples were collected facing upstream, and were taken from within the wetted channel in flowing water. Fine sediment may have been suspended when the bed material was disturbed during sampling. For this reason, the percentage of fine sediment in spawning sites in Freshwater Creek may have been underestimated by this sampling technique. The proportional volume of the smallest size category (<0.106 mm in their experiments) was found by Schuett-Hames and others (1996) to be significantly different between McNeil and shovel samples.

Statistical Analysis

A two stage sampling approach involved collecting several subsamples per study reach, analyzing each subsample separately, then computing statistics using the mean particle size parameters from all pooled subsamples, and assuming a normal distribution of sample means. Sampling precision was estimated by calculating the 95 and 80 percent confidence intervals for the D_{25} and D_{16} percentiles for each reach according to method in Bunte & Abt (2001, p. 303). The mean, standard deviation and standard error were calculated for the D_{16} and D_{25} in each study reach. The confidence intervals represent the range within which the true population mean is expected to occur at the stated level of probability.

Sample bias was evaluated by comparison of the mass of the largest particle in the subsample to the total subsample mass. To avoid bias towards larger particles, Bunte and Abt (2001) recommend that the weight of the largest particle should be less than three percent of the total sample weight. Samples in which the weight of the largest particle exceeded 3 percent of the total sample weight were considered to be biased and should be treated with caution.

Results

Cumulative particle size distribution curves for each of the study reaches are shown in *figure 2*. The distribution curves are composites for each study reach, and were constructed by combining the individual particle size fraction weights from all samples within a reach, and recalculating the percent finer values for each reach. The one mm threshold, representing the particle size that significantly reduces permeability and oxygen flow in redds, can be represented by a D_{16} that is one mm or coarser (in other words, the upper bound of the 11 to 16 percent target for fines in the PL HCP). In *figure 3,* mean D_{16} values that fall above the 1 mm threshold represent spawning gravels where incubation of eggs in redds is not threatened by excess fine sediment. The PL HCP target range for <6.35 mm sediment (particles that may block emergence of alevins) is 25 to 30 percent in the PFC matrix. Therefore the D_{16} and D_{25} can be used to test fine sediment thresholds.

Although the Waipaoa River is not a fish-bearing stream, the same fine sediment target thresholds were applied for purposes of evaluating sediment sampling techniques. The D_{16} threshold value of one mm appears to be met in four of seven reaches at the 95 percent confidence level. The D_{25} threshold of 6.35 mm was not met in any reach. Confidence intervals about the mean D_{16} and D_{25} values were ±0.22 mm and ±0.47 mm for the 95 percent confidence level.

The D_{16} target value of one mm appears to be met by all sampled North Bay (Napa, Sonoma, Mendocino) streams at the 80 percent confidence level. However, at the 95 percent confidence level we were unable to determine if the target value for D_{16} was met. The only stream that met the target for D_{16} at the 95 percent confidence level was Green Valley Creek. The D_{25} target of 6.35 mm was not achieved for Salt Hollow and Green Valley Creeks at the 95 percent confidence level. The target value appears to be met in Soda Creek at the 80 percent confidence level. We were unable to determine if the target value was met for Sulphur or Carneros Creeks at the 95 percent and 80 percent confidence level.

The smaller volume bed material samples collected from Freshwater Creek indicate that the D_{16} target value was met for five of nine reaches at the 80 percent confidence level. We were unable to determine if targets were met at the 95 percent confidence level for all but one reach, where the target was definitely met (reach 32). Mean D_{25} values indicate that HCP targets were met in most reaches in Freshwater Creek, however, considering the 95 percent and 80 percent confidence intervals we were unable to determine if the targets were met or not. The D_{25} target was not met in one (reach 135) at both 95 percent and 80 percent confidence levels.

Discussion

The accuracy of a sediment sample is defined as how closely the sample distribution represents the true distribution of sediment sizes in the channel, and sample precision refers to the size of the deviations from the sample mean to the mean value obtained by repeated sampling (Bunte and Abt 2001). For a specified sampling accuracy and precision, sample size should increase as the variability of the parent population increases (in other words, as the sorting becomes poorer or the standard deviation becomes larger). By consistently sampling the same position within the channel, or habitat unit, the variability of particle sizes within a single stream reach should be minimized. However, the northern California data were collected exclusively from pool tail-out positions (potential spawning sites), and illustrate the high degree of variability within a single habitat unit and stream reach. The data also indicate that either a larger number of replicate samples, or larger individual samples are required to assess if fine sediment thresholds are met or not.

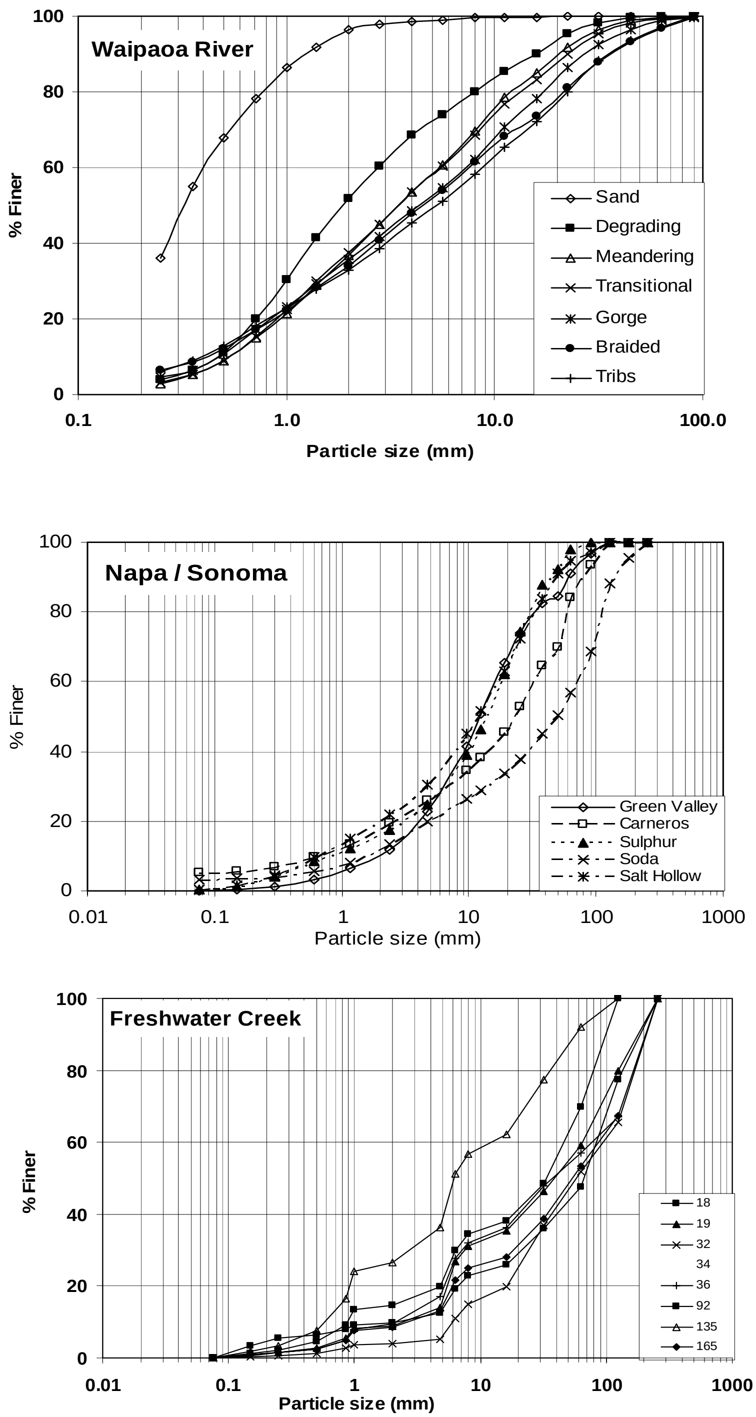


Figure 2—Cumulative particle size distribution curves showing composite samples from the 3 study areas.

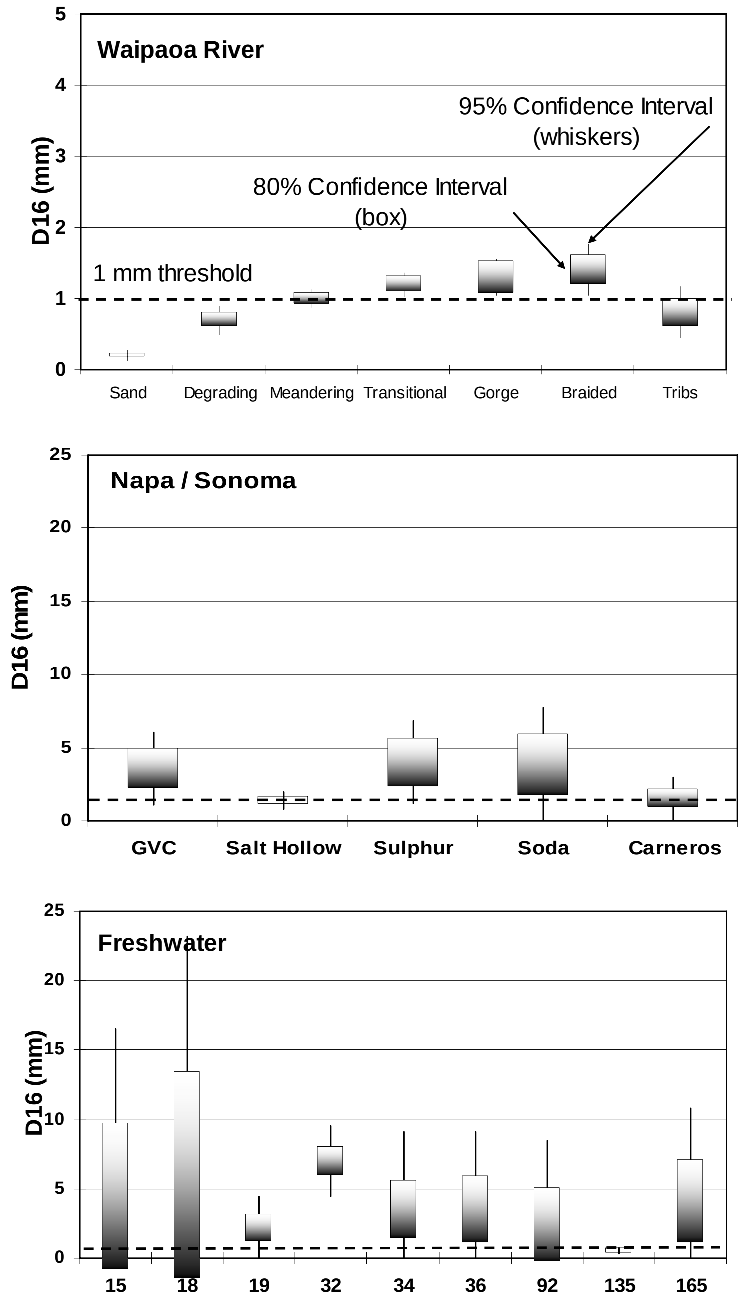


Figure 3—Box and whisker plots showing the 95 percent confidence intervals (whiskers) and the 80 percent confidence intervals (box) around the mean D_{16} percentile values. When the mean value is below the threshold value, the implication is that the "properly functioning conditions" are not met.

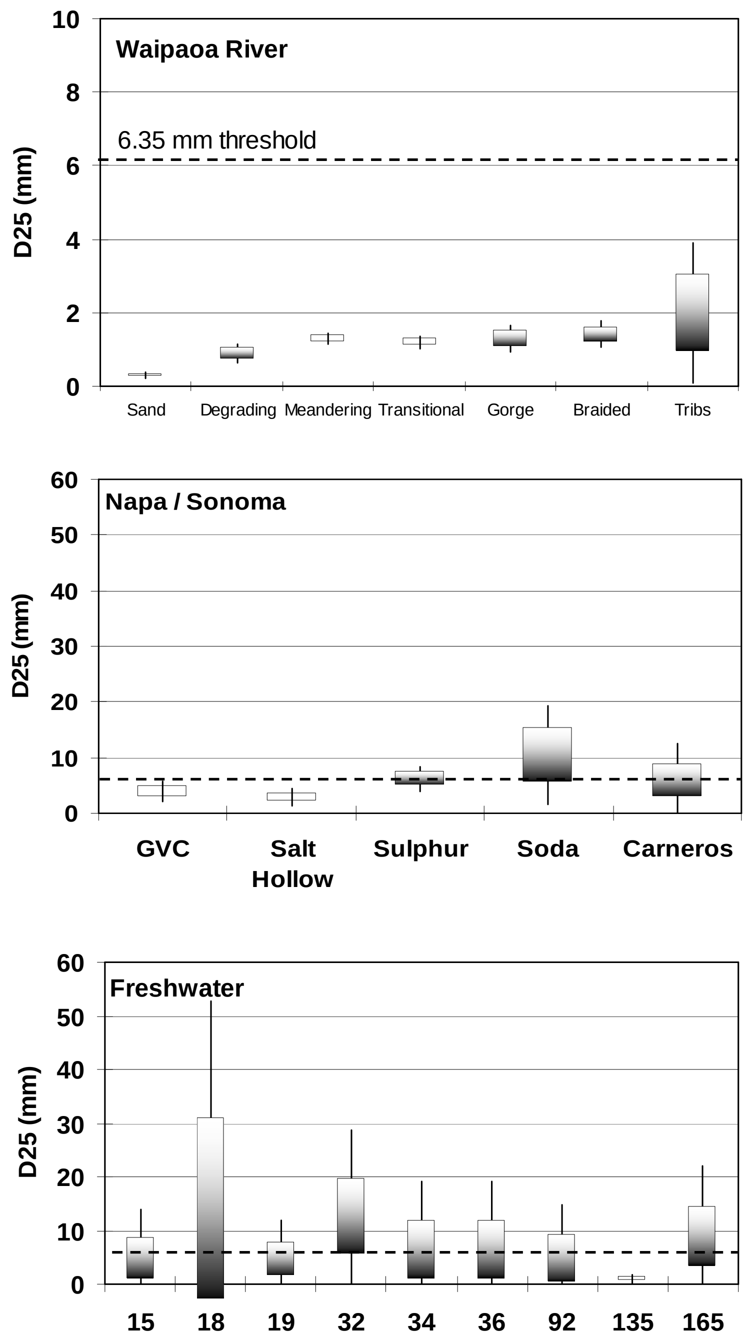


Figure 4—Box and whisker plots showing the 95 percent confidence intervals (whiskers) and the 80 percent confidence intervals (box) around the mean D_{25} percentile values. When the mean value is below the threshold value, the implication is that the "properly functioning conditions" are <u>not</u> met.

The Waipaoa River does not support a salmonid population because of the characteristics of the bed material. There is abundant fine material supplied to the river by large active gullies in the upper basin. The very narrow (±0.22 mm mean value for 95 percent confidence level) confidence intervals for bed material samples from the Waipaoa River likely reflects the large number of large unbiased samples collected, the fine particle size distribution of the bed material, and a sampling scheme that was designed for the physical conditions in the river. If a large number of unbiased samples are collected, the confidence intervals around the mean D_{16} and D_{25} values are small and we can say with greater certainty that fine sediment thresholds have been met or not (in the case of the Waipaoa River, not met). However, this level of sampling intensity is generally not necessary or achievable for most practical situations where spawning gravel composition is to be assessed.The majority of the sampled reaches in the North Bay region are known to support salmonid populations, predominantly steelhead trout except for Green Valley Creek. These streams were sampled using the standard technique for determining spawning gravel composition in the Pacific Northwest, using a McNeil sampler at pool tail-out postions within the channel. Most of the samples stream reaches appeared to meet the D_{16} target value at the 80 percent confidence level. We were unable to determine if the target value was met for Sulphur or Carneros at the 95 percent and 80 percent confidence level. The volume of material collected with a McNeil sampler is often not large enough to exclude sample bias caused by unrepresentative sampling of the larger size classes. A larger number of subsamples is required to achieve greater sample accuracy and precision. However, due to the small areal extent of potential spawning sites, particularly in small streams, it may not be practical to take larger samples.

The large spread of confidence intervals for bed material samples from Freshwater Creek likely reflects the variability and coarse nature of the bed material, the small amount of sediment in the subsamples, and the small number of subsamples used to characterize each reach (n = 3). The uneven shape of the cumulative distribution curves *(fig. 2A)* is indicative of the bias caused by the high proportion of larger grains in the sample. Larger samples are required to reduce the bias of larger particles and to produce smooth representative size distribution curves.

If regulatory targets are to be set with respect to percent fines, the required confidence intervals should also be stated so that sample sizes and the number of replicates required can be calculated. There may be a need to relax standard statistical confidence levels from the standard value of $\alpha = 0.05$ when dealing with natural resource applications to take into account the highly variable nature of riverbed sediments and the level of effort required to collect and analyze samples. The given examples represent fairly intensive sampling efforts, yet do not produce very precise estimates of the particle size percentiles of interest.

Conclusions and Implications

To determine if streambed material meets specified regulatory targets with respect to salmonid spawning habitat quality, bed material sampling efforts should be tailored to the specific study reach under investigation. Sampling technique, including sample size, and number of samples, should reflect the physical characteristics of the study reach. Any particle size sampling program should consider the level of accuracy that will be achieved, and whether the chosen sampling scheme will produce results that will determine if regulatory thresholds are met or not. A critical step is collection of sample data to calculate sample variances used to predict requisite sample mass and number of samples required to achieve an acceptable level of precision. Statistical power should also be considered.

Particle size sampling is a substantial undertaking. Users of particle size data, including regulatory agencies and landowners, should be conscious of the statistical constraints involved with bed material sampling if numeric targets are to be set and assessed.

Acknowledgments

Lester McKee, Sarah Pearce–San Francisco Estuary Institute; Ron Benkert–Sonoma County Water Agency; Ulysses Lolonis and Greg Lolonis–Lolonis Vineyards, Inc; Gretchen Oliver, Kate Sullivan, Jeff Barrett–Pacific Lumber Company; Basil Gomez –Indiana State University; Dave Peacock–Gisborne District Council; Andrea Rosser, Vaughan Griffin, Claire Guthrie–visiting Kiwi field assistants.

The particle size data from the Waipaoa River were collected as part of a MSc thesis (Gomez and others 2001, Rosser 1997), with funding from the Gisborne District Council. Data for Freshwater Creek were provided by the Pacific Lumber Company. Data from Soda, Carneros and Sulphur Creeks were collected as part of a Cal-Fed funded study under contract with the San Francisco Estuary Institute. Data from Green Valley Creek was collected under contract with the Sonoma Water County Agency. Data from Salt Hollow Creek was collected for Lolonis Vineyards, Inc.

References

Bunte, K.; Abt, S.R. 2001. **Sampling surface and subsurface particle-size distribution in wadable gravel- and cobble-bed streams for analyses in sediment transport, hydraulics, and streambed monitoring.** Gen. Tech. Rep. RMRS-GTR-74. Fort Collins, CO: Rocky Mountain Research Station, Forest Service, U.S. Department of Agriculture; 428 p.

Chapman, D.W. 1988. **Critical review of variables used to define effects of fines in redds of large salmonids.** Transactions of the American Fisheries Society 117: 1-21.

Gomez, B.; Rosser, B.; Peacock, D.; Hicks, M.; Palmer, J. 2001. **Downstream fining in a rapidly aggrading gravel bed river**. Water Resources Research 37: 1813-1823.

Hicks, D.M.; Gomez, B.; Trustrum, N.A. 2000. **Erosion thresholds and suspended sediment yields: Waipaoa River basin, New Zealand**. Water Resources Research 36: 1129-1142.

Kondolf, G.M. 2000. **Assessing salmonid spawning gravel quality**. Transactions of the American Fisheries Society 129: 262-281.

Kondolf, G.M.; Wolman, M.G. 1993. **The sizes of salmonid spawning gravels**. Water Resources Research 29: 2275-2285.

McNeil, W.J.; Ahnell, W.H. 1964. **Measurement of gravel composition of salmon stream beds**. Circular No. 120. Seattle, WA: University of Washington, Fisheries Research Institute.

Moore, P.R.; Mazengarb, C. 1992. **Geology and landforms of the Raukumara Peninsula**. In: Soons, J.; Selby, M.J., eds. Landforms of New Zealand. Reading, MA: Addison-Wesley-Longman; 334-343.

Pearce, S.; O'Connor, M.; Grossinger, R.; McKee, L. 2002. **Napa River TMDL baseline study: geomorphic processes and habitat form and function in Soda Creek**, a technical report of the Regional Watershed Program. SFEI Contribution 63. Oakland, CA: San Francisco Estuary Institute.

Rosser, B. 1997. **Downstream fining in the Waipaoa River; an aggrading gravel-bed river, East Coast, New Zealand.** Palmerston North, New Zealand: Massey University. M.S. thesis.

Schuett-Hames, D.; Conrad, B.; Pleus, A.; Smith, D. 1996. **Field comparison of the McNeil sampler with three shovel-based methods used to sample spawning substrate composition in small streams**. Timber Fish and Wildlife Ambient Monitoring Program Report TFW-AM 9-96-005.

Trafford, J.A. 1998. **Bedload transport at Kanakania, Waipaoa River, East Coast, New Zealand**. Auckland, New Zealand: University of Auckland. M.S. thesis.

Valentine, B.E. 1995. **Stream substrate quality for salmonids: guidelines for sampling, processing, and analysis**. Santa Rosa, CA: California Department of Forestry and Fire Protection, Coast Cascade Regional Office.

Vaux, W.G. 1962. **Interchange of stream and intragravel water in a salmon spawning riffle**. Special Science Report, Fisheries No. 405. Washington, DC: USDI Fish and Wildlife Service.

Simulation of Surface Erosion on a Logging Road in the Jackson Demonstration State Forest[1]

Teresa Ish[2] and David Tomberlin[3]

Abstract

In constructing management models for the control of sediment delivery to streams, we have used a simulation model of road surface erosion known as the Watershed Erosion Prediction Project (WEPP) model, developed by the USDA Forest Service. This model predicts discharge, erosion, and sediment delivery at the road segment level, based on a stochastic climate simulator and road segment characteristics. To apply the WEPP model, we collected data on road segment dimensions, design and slope for 57 contiguous road segments totaling 3.5 miles of Road 630 on the Jackson Demonstration State Forest.

Results of the simulation exercise suggest a long-term average annual erosion rate on Road 630 of 20.8 metric tons per kilometer per year, with a standard deviation of seven metric tons per kilometer per year. Overall, we found the WEPP software well designed and flexible enough to meet our needs, as it allows specification of many different road and environmental parameters. One shortcoming for our purposes was that WEPP's stochastic climate generator does not simulate serial correlation. We have begun a field study of road surface erosion rates that will enable evaluation of the predictive ability of the WEPP model in the redwood region.

Key words: erosion simulation, road surface erosion, and WEPP

Introduction

Because erosion can impair road function and contribute to excess sedimentation in fish-bearing streams, road erosion control is a high priority for many landowners. While the relative effectiveness of different erosion control treatments on roads is not yet well known, a few studies have been completed (Burroughs and King 1989, Madej 2001), and others are underway. More fundamentally, data on road erosion are relatively scarce in comparison with the data available on sediment loads in streams. Having information only on in-stream sediment makes the attribution of erosion to a particular type, amount or locality of soil loss at the source difficult. Without these basic data, the risks and benefits of different road erosion treatments cannot be characterized empirically.

Road erosion may take the form of mass wasting (for example, fill failures), fluvial erosion (for example, ditch scour) or surface erosion. Mass wasting is generally thought to be the most important of these in the redwood region, while

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Staff Research Associate, University of California, Santa Cruz, under contract to the National Marine Fisheries Service. Current affiliation: Sustainable Fishery Advocates, P.O. Box 233, Santa Cruz CA 95061. email: ishsurf@yahoo.com

[3] National Marine Fisheries Service, Santa Cruz, CA. email: david.tomberlin@noaa.gov

surface erosion is generally considered the least important. Perhaps because of this belief, during the course of our work on decision models for erosion control we were unable to find any published time series of road surface erosion rates. To develop estimates of road surface erosion, we turned to a simulation model known as the Watershed Erosion Prediction Project (WEPP) model. This paper describes our application of the WEPP model to Road 630 on the Jackson Demonstration State Forest in Mendocino County, summarizes the results obtained, and concludes with an assessment of the model's usefulness for our purpose, which was to develop input data for a dynamic optimization model of erosion management.

Study Site

Road 630 is a partially rocked road located in the Middle Fork Caspar Creek watershed. This watershed covers approximately 1300 acres, with elevation ranging from 100 to 1000 feet and slopes from zero to 100 percent. The climate is Mediterranean, with average annual precipitation between 40 and 70 inches. Second- and third-growth redwood and Douglas-fir are the dominant vegetative community, and streams within the watershed support both coho salmon and steelhead trout. The coastal belt of the Franciscan assemblage, consisting of marine sedimentary and volcanic rocks, underlies the watershed. For most of its length, Road 630 runs mid-slope along the Middle Fork of Caspar Creek, where predominant soil types are the Dehaven-Hotel complex and Irmulco-Tramway complex at lower elevations and Vandamme at higher elevations. For most of its length, the road is crowned with an inside ditch. A recent inventory of erosion potential on the road suggests that both mass wasting (landslides or stream crossing failures) and chronic surface erosion are likely to deliver significant volumes of sediment to nearby Caspar Creek (Pacific Watershed Associates 2001).

Methods

The WEPP Model

The Water Erosion Prediction Project model was developed by the USDA Agricultural Research Service to estimate soil loss on rangeland and cropland (Flanagan and Nearing 1995). It has been adapted by the USDA Forest Service for use on forestlands and forest roads (Elliott and others 1999). The model includes 12 conceptual components that influence soil detachment, among which are climate, irrigation, hydrology, water balance residue, management and composition, tillage impacts on infiltration, and deposition. A full description of the model is available in Flanagan and Livingston (1995).

The unit of analysis for the WEPP Road application is the road segment, defined in hydrologic terms: a segment is the area of road from which surface flow exits at a common point, such as a ditch relief culvert or water bar. Application of the model requires input of road segment characteristics (described in the next section) and specification of model parameters related to climate, soil characteristics and management practices. WEPP provides a wide variety of pre-specified parameter files (for example, climate in Fort Bragg, or Dehaven soil characteristics) and also allows the user to adapt these parameters or create an entirely new parameter profile. WEPP includes a stochastic climate generator known as Cligen, which generates rainfall and storm frequency intended to reflect historical data for a specific study site.

WEPP is available in both DOS and Windows forms. We used the Windows version, which allows the model to be specified within an easy-to-use graphical user interface.

Data Collection

Data for Road 630 were collected on June 24 and June 25, 2003. A total of 57 contiguous segments were identified, totaling 3.5 miles of road. Data collected for each segment included road grade, segment length, segment width, ditch presence, vegetative level in ditch, and critical features such as rutting or standing water. Because of the difficulty of ascertaining how much of the road width would contribute to point-source runoff from the segment, three values were recorded for width: a low value corresponding to the traveled way, a medium value indicating measurement from the outer berm or beginning of loose detritus on the fill-slope side of the road to the center of the ditch (if present) and a high value indicating measurement to the start of forest vegetation on both sides. Where length data for a segment could not be taken with a single measurement, the segment was broken into sub-segments.

Specification of Simulation Parameters

For each road segment, we specified soil type and road configuration (in other words, insloped or not, ditch present or not) for the simulation. Because road construction may have altered the physical properties of the soil types found in this region, we standardized the soil type to 'sandy loam road surface' for all segments during our basic simulations, and reserved the use of segment-specific soil types (Dehaven, Tramway, and Vandamme soil, which are available in the standard WEPP distribution) for the sensitivity analysis. WEPP allowed for the inclusion of sub-segment slope and length data. We selected the Fort Bragg, CA, climate data for our simulation from the climate models provided by WEPP. We used a continuous simulation over a 1000-year time horizon.

Due to the difficulty of matching the road segments to the stylized geometry of the WEPP model, we initially explored two road configurations for Road 630. The first scenario, labeled 'insloped' below, designated each sub-segment either insloped road with vegetated or rocked ditch, or outsloped road with rutting; only six of fifty-seven segments were designated outsloped. The second configuration, labeled 'outsloped' below, designated all segments as outsloped, some rutted and some not. Given that most segments were crowned or of ambiguous configuration due to the road being unmaintained, we averaged the results of simulations for these two configurations to arrive at a summary estimate, labeled 'best estimate' below.

Output Analysis

Model projections for soil loss were reported by WEPP in $kg/m^2/yr$. Due to uncertainty about how much of the road width would contribute to point-source runoff from each segment, we ran the model for each segment with low, medium, and high width estimates. With these estimates, we calculated metric tons of soil lost per kilometer (mt/km) of road per year using the reported WEPP soil loss in kg/m^2 and a representative soil bulk density estimate of 1100 kg/m^3 (Wosika 1981, pp. 47-48).

Results

We generated two basic estimates of long-run rates of surface erosion on Road 630, both based on 1000-year simulations and both calculated as the long-run weighted average[4] soil loss for the entire road. The first simulation, 'insloped,' in which most segments were classified as insloped (n = 51) while those with ruts (n = 6) were classified as outsloped, generated an estimate for the rate of annual surface erosion of.18.8 mt/km/yr ± 7.2 (127 yd^3/yr ± 45). The second simulation, 'outsloped,' in which for comparative purposes all segments (n = 57) were classified as outsloped, generated an estimate of 22.0 mt/km/yr ± 6.8 (148 y^3/yr ± 47). This second simulation yielded a higher estimate because outsloped segments were assumed to require more frequent grading. Further averaging over these two putative road configurations to arrive at our 'best estimate' yielded estimated soil loss of 20.4 mt/km/yr ± 7.0. Some external corroboration is provided by a comparison to erosion estimates from a field survey, which estimated future annual soil loss on Road 630 at 66 to 330 yd^3/yr (10 to 49 mt/km/yr), depending on use (Pacific Watershed Associates 2001). Our predicted losses are well within this range.

On individual segments, soil loss (again, after averaging over estimates by road width) ranged from zero mt/km/yr to 41.8 mt/km/yr.

Sensitivity Analysis

As mentioned, the 'outsloped' simulation yielded somewhat greater soil loss than did the 'insloped' simulation *(fig. 1)*. To examine the results' sensitivity to soil type, we re-ran these two basic simulations, substituting site-specific soil types (Dehaven, Tramway, and Vandamme) for the generic 'sandy loam road surface.' Under the insloped road configuration, subsituting particular soil types resulted in somewhat lower estimated soil loss than under the sandy loam soil type, however this pattern was not apparent under the 'outsloped' road configuration. When the two configurations were averaged, using the 'sandy loam road surface' soil type tended to result in a lower estimated average soil loss than all other soil types and variants on these types, though this effect varied by segment *(table 1)*. Further tests of model sensitivity, for example, altering albedo or critical shear, had little effect on estimated soil loss.

Variation in soil loss for each segment as well as for the entire road was more heavily influenced by the measured width of the road than by changing parameters of soil or management type *(table 1)*. For the segment (number 37) with the largest difference between the low and high widths, soil loss was over 400 percent greater using the high width than the low width. This is in contrast to the difference in soil lost by varying soil or management parameters, where the largest difference was about 50 percent from the base scenario. Using the soil types specific to each segment had a larger effect than management type, but this effect was still small in comparison to the effect of measurement error in width.

[4] The low and high width estimates were weighted by 0.2 each and the medium width estimate was weighted 0.6.

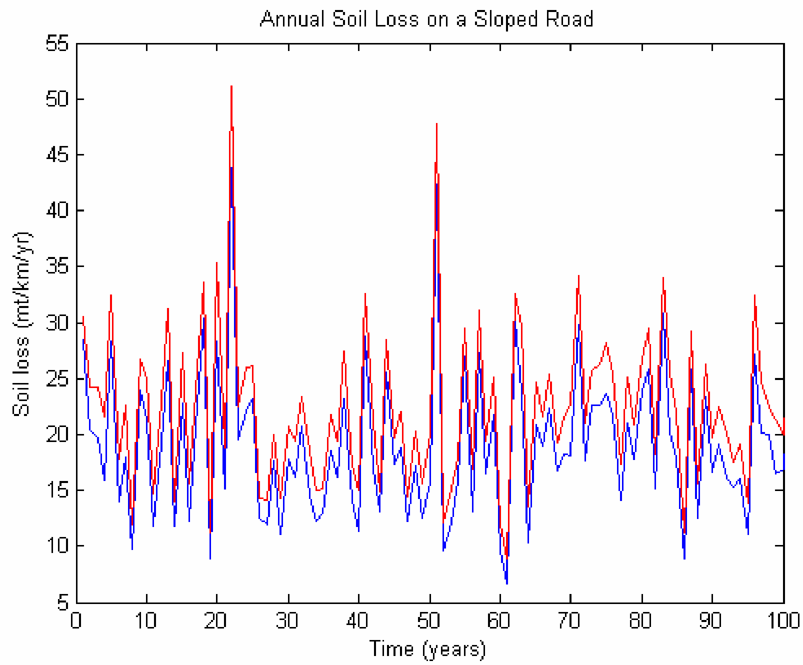


Figure 1—Simulated soil loss, after taking the weighted average of width, for 100 years. Red shows the outsloped road configuration, blue shows the insloped road configuration. Note this figure does not show the averaged 'best estimate' scenario.

Table 1—*Estimated soil loss (in mt/km/yr) on two segments and the entire road at the low, medium, and high width measurements, and with different road configurations and soil types. Segments 2 and 37 represent extreme examples of measurement differences between the low and high widths.*

	Low width	Medium width	High width	Weighted mean	Percent difference from base scenario
Best estimate scenario, uniform sandy loam soil					
Segment 2	7.8	7.8	7.8	7.8	---
Segment 37	13.3	19.5	67.1	27.8	---
Entire road	13.3	20.0	28.8	20.4	---
Insloped scenario, uniform sandy loam soil					
Segment 2	6.5	6.5	6.5	6.5	−17
Segment 37	13.2	19.3	66.6	27.5	−1
Entire road	12.2	18.5	26.6	18.8	−8
Outsloped scenario, uniform sandy loam soil					
Segment 2	9.0	9.0	9.0	9.0	15
Segment 37	13.4	19.6	67.7	28.0	1
Entire road	14.3	21.6	31.0	22.0	8
Best estimate scenario, segment-specific soil type					
Segment 2	9.8	9.8	9.8	9.8	25
Segment 37	10.8	15.8	54.6	22.6	−19
Entire road	20.6	30.6	43.0	31.1	52

Discussion

Our primary goal in using the WEPP model was to generate estimates of the long-run mean and variance of surface erosion rates on Road 630. The model allowed us to accomplish this goal fairly easily, and with a reasonable degree of confidence in the results. The WEPP model incorporates the most important hydrological and geophysical information needed to make basic predictions of surface erosion rates on logging roads. Further, the graphical user interface and the many data files packaged with the WEPP software (on climate, soil, management, and so forth) make it easy to apply the model to a particular road. The user can easily reflect local conditions by altering parameter values. In testing the sensitivity of model projections to a range of different characterizations of soil and road geometry that might reasonably apply to our study site, we found that model projections for the whole road were generally within a 20 percent range of each other. Since our purpose was to generate order-of-magnitude estimates rather than precise forecasts, we were pleased with this stability. In our case, uncertainty regarding the size of each segment's catchment on soil loss/km/yr dominated the effect of parameter specification (in other words, soil characteristics, such as critical shear or albedo, and road configuration).

An important secondary goal for us was to generate a model of the dynamics of road surface erosion, and for this purpose we found the model was less useful, specifically because of the way the Cligen stochastic climate generator is designed. Cligen produces rainfall data so that the final distribution reflects the historical distribution of rainfall data for the region in question, but it does not have the ability to generate precipitation data that incorporate serial correlation. We tested for serial correlation in the precipitation time series for Fort Bragg from 1913 to 2002 and found more support for the hypotheses that precipitation is generated by an autoregressive or moving average process than by a white noise process. Because Cligen does not provide a means to capture serial correlation, the erosion data generated by WEPP cannot reflect the serial correlation we found in the precipitation history. To the extent that precipitation influences interannual variability in road surface erosion, the formulation of the WEPP model prevents simulated erosion data from reflecting the medium-term dynamics of the surface erosion. Land managers or researchers interested in WEPP-based Monte Carlo simulation should also be aware that the WEPP's current configuration requires resetting the Cligen climate model with each simulation, because the erosion model itself is deterministic.

Our interest in generating a time series of road surface erosion rates arose from the need for this data as input to a dynamic optimization model of erosion control. For this purpose, we needed the long-run mean and variance estimates provided by WEPP, and supplemented WEPP results with a proxy for inter-annual erosion dynamics derived from analysis of historical precipitation data in the study area. While not ideal, this approach seems like a reasonable blend of simulation and time series analysis.

The biggest question, of course, is whether the WEPP model projections for Road 630 are in some sense reliable, and whether the model could usefully be applied to other roads in the redwood region. At this time, we have no basis for judging this. We have begun a project with the California Department of Forestry to collect field data on erosion rates over time. Once this project has produced several years of data, it will be possible to begin empirical testing of the WEPP model's ability to forecast erosion rates in the redwood region.

Acknowledgments

We thank Bill Elliott (USDA Forest Service) for generously sharing his experience and expertise with the WEPP model. For help with the field component of this study, we thank Brian Barrett, Bill Baxter, and Brian Bishop from the California Department of Forestry and Fire Protection and Melissa Chaney and Wes Silverthorne of the National Marine Fisheries Service.

References

Burroughs, E.R., Jr.; King, J.G. 1989. **Reduction of soil erosion on forest roads.** Gen. Tech. Rep. INT-264. Ogden, UT: Intermountain Research Station, Forest Service, U.S. Department of Agriculture; 21 p.

Elliot, W.J.; Hall, D.E.; Graves, S.R. 1999. **Predicting sedimentation from forest roads**. Journal of Forestry 97(8): 23-29.

Flanagan, D.C.; Livingston, S.J., eds. 1995**. USDA-water erosion prediction project: user summary**. NSERL Report No. 11. West Lafayette, IN: USDA-ARS National Soil Erosion Research Laboratory.

Madej, M.A. 2001. **Erosion and sediment delivery following removal of forest roads**. Earth Surface Processes and Landforms 26: 175-190.

Pacific Watershed Associates. 2001. **Summary report, 2001 S.B. 271 Caspar Creek demonstration road assessment and decommissioning plan for Jackson Demonstration State Forest Road 630**. Prepared for California Department of Forestry and Fire Protection and California Department of Fish and Game. Arcata, CA: Pacific Watershed Associates.

Wosika, E.P. 1981. **Hydrologic properties of one major and two minor soil series of the coast ranges of Northern California**. Arcata, CA: Humboldt State Univ.; 150 p. M.S. thesis. *Available at* http://www.fs.fed.us/psw/rsl/projects/water/WosikaMS.pdf

The Relationship Between Wildlife Damage Feeding Behavior and Stand Management in Coastal Redwoods (*Sequoia sempervirens*)[1]

Gregory A. Giusti[2]

A number of vertebrate species occur in the coastal redwood type. None of which are unique to that forest type. However, certain populations of mammals respond favorably to silvicultural and other management manipulations resulting in direct conflict with redwood regeneration efforts. Species such as American black bear *(Ursus americanus)*, dusky-footed woodrat *(Neotoma fuscipes)* and California vole *(Microtus californicus)* have all been shown to exhibit feeding behaviors on various stages of redwood growth. This paper characterizes damage and feeding behaviors for each species and examines the habitat relationships responsible for inciting this behavior.

Data is presented to demonstrate the positive correlation between vertebrate feeding behavior and stand age and structure.

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Unviersity of California, Cooperative Extension, 579 Low Gap Road, Ukiah, CA 95482, (707) 463-4495. email: gagiusti@ucdavis.edu

Disease Ecology of *Phytophthora ramorum* in Redwood Forests in the California Coast Ranges[1]

P.E. Maloney,[2] S.L. Lynch, S.F. Kane, C.E. Jensen, and D.M. Rizzo

Phytopthora ramorum, commonly known as sudden oak death (SOD), infects a whole suite of hosts, ranging from oaks, to conifers, to herbs, causing lethal branch or stem cankers, as well as non-lethal foliar and twig infections. In the US, the disease is found on host species (in other words, tanoaks, oaks, redwood, and bay) from the Big Sur coastline in Monterey County, California to Curry County, Oregon, with most *P. ramorum* confirmations and collections being within 30 km of the Pacific coastline. Given its wide host range *P. ramorum* is associated with a number of forest types throughout coastal California and Oregon. Included in these forest types are coast redwood forests with redwood and many associated species in this forest type as hosts of *P. ramorum*. Symptoms and signs on hosts in coast redwood forests vary from host to host. On redwood *P. ramorum* causes needle lesions, cankers on small branches, and tip dieback of sprouts. A total of 120 plots in four locations (30 plots/location) have been established in redwood forests along the California coast ranges to study the epidemiology and ecology of *Phytophthora ramorum* in this forest ecosystem. Our plots represent a north-south gradient of *P. ramorum* populations from Sonoma to Monterey Counties. Our study sites in Sonoma and Marin counties fall into the central redwood forest subregion while our study sites in Santa Cruz and Monterey counties fall into the southern redwood forest subregion. All four locations share similar climate with cool and wet fall, winter, and early spring; conditions favorable for *P. ramorum* which appears to sporulate most actively during December to May. Disease incidence in these four redwood forest locations range from 19 to 32 percent, with three to seven percent of redwoods infected with *P. ramorum*. However, no mortality has been observed to be associated with this disease. Many redwood infected individuals are seedlings, saplings, or sprouts. In two highly infested *P. ramorum* stands, two large redwood individuals (= 45 m tall) were sampled. Samples were taken from the upper, mid, and lower canopy of these trees, in addition bark was also collected. No *P. ramorum* was recovered from any of the needle or bark material from these large individuals. In a canopy contribution experiment we find that redwood becomes infected, will sporulate while the infection is new, but disease generally does not progress much past a small needle or branch lesion. Of course cumulative, year to year infections in *P. ramorum* infested forests may have an effect on redwood individuals and populations. High incidence of *P. ramorum* and mortality of many of the associated species in these forests may lead to structural and compositional changes that will subsequently effect forest dynamics and other ecosystem processes in coastal California redwood forests.

[1] An expanded version of this paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Department of Plant Pathology, University of California, One Shields Ave., Davis, CA 95616, (530) 754-9894. email: tbntm@telis.net

A Multiple Logistic Regression Model for Predicting the Development of *Phytophthora ramorum* symptoms in Tanoak (*Lithocarpus densiflorus*)[1]

Mark Spencer[2] and Kevin O'Hara[3]

Abstract

Phytophthora ramorum attacks tanoak *(Lithocarpus densiflorus)* in California and Oregon. We present a stand-level study examining the presence of disease symptoms in individual stems. Working with data from four plots in redwood *(Sequoia sempervirens)*/tanoak forests in Marin County, and three plots in Mendocino County, California, we evaluated a number of stem variables such as dbh, height, crown class, foliar condition and the presence of insect, fungal infection or disease symptoms for significance in multiple logistic regression analysis. We built a maximal model of variables and simplified this down following Akaike's Information Criterion. Our final model includes only variables significant at p = 0.01. We compare disease presence and mortality results in Marin stands with results collected in Mendocino County.

Key words: mortality, multiple logistic regression, Phytophthora ramorum, *tanoak*

Introduction

Phytophthora ramorum has emerged as a major source of tanoak *(Lithocarpus densiflorus)* mortality in the redwood/tanoak forests of central California. In this forest system other species hosts of *P. ramorum* include madrone *(Arbutus menziesii)*, California bay laurel *(Umbellularia californica)*, bigleaf maple *(Acer macrophyllum)*, huckleberry *(Vaccinium ovatum)* and redwood *(Sequoia sempervirens)* (Rizzo and others 2002a). While the pathogen seems to cause rapid mortality in madrone and tanoak, infection appears to be limited to leaves in these other hosts (Rizzo and Garbelotto 2003).

Unlike coast live oak which is limited to stem infections, tanoak are also susceptible to foliar *P. ramorum* infection (Rizzo and Garbelotto 2003, Rizzo and others 2002b). The ability of tanoak to potentially support sporulation might diminish the role of other foliar host species as vectors in redwood/tanoak stands. To date, published reports on the dynamics of the disease in redwood/tanoak stands are limited.

This study builds a multiple logistic regression model using individual stem

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Academic Coordinator, Univ. of California, Berkeley, College of Natural Resources, 137 Mulford Hall, Berkeley, CA 94720. email: spencer@nature.berkeley.edu

[3] Professor of Forestry, Univ. of California, Berkeley, Department of Environmental Science, Policy and Management, 137 Mulford Hall, Berkeley, CA 94720. email: ohara@nature.berkeley.edu

attributes. Variables initially considered for inclusion in the model are listed in *table 1*. We developed our binary response variable using the results from our last stand survey in June 2003 including only trees that showed no *P. ramorum* symptoms in our July 2001 survey. The model that we have built enables us to assess the influence of individual tree attributes on *P. ramorum* infection probabilities for individual tanoak stems. We utilize the model to develop suggestions for management action.

Table 1—*List of stem attributes collected and included in the maximal model.*

Species
height, diameter, crown class
percent live foliage, foliar condition
presence/absence of bark and ambrosia beetles
stem growing individually or attached below breast height
presence of infection on neighboring stem attached below breast height

Methods

The study area is located on Marin Municipal Water District lands on the East Slope of Bolinas Ridge above Kent Lake reservoir (38.0^o N, 122.7^o W). The study plots are at elevations between 110 and 270 m. Soils of the area include Dipsea and Barnabe very gravelly loams (Kashiwagi 1985). The forest canopy is stratified with redwood and Douglas-fir *(Pseudotsuga menziesii)* in the upper stratum and tanoak, madrone, and California bay occupying the layer beneath. Annual precipitation averages between 750 and 1250 mm. Average temperature is 12 ^{o}C (Kashiwagi 1985).

In the summer of 2001 we established the four Marin plots containing an average of 410 stems greater than 4cm dbh. These plots range in size from 50 m by 60 m (0.32 ha) to 50 m by 80 m (0.4 ha). The density of tanoak stems in these stands ranged between 840 and 1020 stems/ha. We established the three Jackson plots 0.26, 0.29 and 0.30 ha, respectively, in February of 2002. Tanoak stem densities in these plots ranged between 890 and 1090 stems/ha. Within each plot we used a Laser-Ace 300 and Haglof distance measuring device to map all stems greater than 4 cm dbh with a relative precision of ±10 cm. In addition to recording species, diameter, height, and crown class we visually estimated foliar condition and documented the presence/absence and age of bark and ambrosia beetle infestation, presence/absence of fungi infestation including *Hypoxylon thouarsianum*, presence/absence and location of bleeding areas, presence/absence of stem cankers beneath bark wounds and the presence/absence of apparent foliar and twig infections consistent with *P. ramorum* infection. These plots were revisited and each tree assessed four times over the three year period of the study.

To better understand the dynamics of tanoak mortality in the absence of *P. ramorum* we established and mapped three plots in *P. ramorum* free stands at Jackson Demonstration State Forest. The three plots are located in the Little North Fork of the Big River watershed (39.35° N, 123.65° W at 180 m) and (39.35° N, 123.71° W at 240 m) and the Upper Parlin Creek (39.40°N, 123.66°W at 240 m) in Mendocino County. The soils of the basins are well-drained clay-loams, one to two m in depth and derived from Franciscan greywacke sandstone and weathered, coarse-grained shale of the Cretaceous Age. Ninety percent of the annual precipitation of 1200 mm falls from October through April (Rittiman and Thorson 1987).

To test the accuracy of our field diagnosis and assess our ability to differentiate between bleeding areas related to *P. ramorum* cankers and those due to other causes we randomly selected 36 stems from our population of stems with bleeding areas. We submitted these samples for PCR and culture analysis. Of the 36 samples, our field assessment classified 19 as *P. ramorum* infected and 17 as uninfected with bleeding areas that were due to causes unrelated to *P. ramorum*. The results of the PCR analyses confirmed our field diagnoses with 100 percent consistency.

We developed a multiple logistic regression model to assess the relationship between stem attributes and the probability of individual stems developing *P. ramorum* symptoms. We coded these stems with "0" or "1" depending on whether they developed a *P. ramorum* related bleeding stem canker or foliar symptoms between the initial, 2001, and final, 2003, stand surveys. For each stem the percent live foliage, foliar condition and presence/absence of bark variables date from the prior year's survey. Thus if a tree became symptomatic in 2002 its relevant vegetation and invertebrate data from the 2001 survey were entered in the model. For trees that remained symptom free throughout the course of the study, data from the survey immediately prior to the final June 2003 survey were entered in the model. We arcsine transformed foliar data. Prior to including predictor variables in the maximal model we examined them for correlation and used stepwise deletions following the Akaike Information Criterion to arrive at an initial minimal adequate model (Akaike 1973). We then tested for non-linear and interaction effects and compared the resulting models with the initial minimum adequate model to determine whether these non-linear or interaction explanatory variables led to a significant increase in the explanatory power of the model. We performed all statistical analyses using SPLUS 6.2 statistical software.

Results

In the three Mendocino plots the average density of tanoak stems was 1040 stems/ha. In our initial survey in February 2002 we found an average of 131 dead stems/ha (11 percent of stems). No additional stem mortality occurred in those plots between the initial survey in January of 2002 and final survey in August of 2002. In that time period no stems exhibited symptoms of *P. ramorum* infection. By comparison in the Marin plots mortality climbed from an average 151 to 283 tanoak stems/ha between July 2001 and June 2003, an average increase of 15 to 29 percent of tanoak stems over the course of the study. Symptom incidence rose from an average 112 to 183 symptomatic tanoak stems/ha, an average increase of 12 to 20 percent of tanoak stems. In *table 2* we provide a summary of stem counts, disease incidence and stem mortality by plot.

We list our multiple logistic regression model parameter estimates for the binary disease outcome in *table 3*. The presence or sign of bark beetle activity on the stem in the previous year contributes more to the probability of becoming symptomatic than the other variables in this model. Of the 24 trees that became symptomatic between July of 2002 and June of 2003, four had insect activity in 2002. Of the 29 trees that became symptomatic between February of 2001 and July of 2002, seven had signs of insect activity in 2001. Signs of insect activity in tanoaks are rare (McPherson and others 2002). Of the 888 stems included in the model only 112 showed signs of insect activity throughout the study. The percent live foliage of the previous year was also highly significant.

Table 2—*A summary of plot sizes, stem counts, disease incidence, and stem mortality by plot and sampling date.*

Plot				Sampling date					
Marin county				**July 2001**	**Jan 2002**	**Feb 2002**	**July 2002**	**Aug 2002**	**June 2003**
11	0.42	ha.	dead tanoak stems	30		61	65		78
	425	stems	percent tanoak stems that are dead	9%		18%	19%		22%
	348	tanoak stems	density of dead tanoak per ha.	72		147	157		188
	839	tanoak stems/ha.	symptomatic tanoak stems	48		66	80		90
			percent tanoak stems that are symptomatic	14%		19%	23%		26%
			density of symptomatic tanoak stems per ha.	116		159	193		217
12	0.36	ha.	dead tanoak stems	34		77	86		100
	397	stems	percent tanoak stems that are dead	10%		23%	25%		30%
	338	tanoak stems	density of dead tanoak per ha.	95		215	241		280
	945	tanoak stems/ha.	symptomatic tanoak stems	33		47	52		57
			percent tanoak stems that are symptomatic	10%		14%	15%		17%
			density of symptomatic tanoak stems per ha.	92		131	145		159
13	0.33	ha.	dead tanoak stems	56		89	91		107
	379	stems	percent tanoak stems that are dead	17%		28%	28%		33%
	321	tanoak stems	density of dead tanoak per ha.	172		274	280		329
	988	tanoak stems/ha.	symptomatic tanoak stems	40		58	65		68
			percent tanoak stems that are symptomatic	12%		18%	20%		21%
			density of symptomatic tanoak stems per ha.	123		178	200		209
14	0.36	ha.	dead tanoak stems	95		108	115		121
	439	stems	percent tanoak stems that are dead	26%		30%	32%		33%
	365	tanoak stems	density of dead tanoak per ha.	266		302	322		338
	1021	tanoak stems/ha.	symptomatic tanoak stems	42		44	47		53
			percent tanoak stems that are symptomatic	12%		12%	13%		15%
			density of symptomatic tanoak stems per ha.	117		123	131		148
Mendocino county									
21	0.26	ha.	dead tanoak stems		26			26	
	450	stems	percent tanoak stems that are dead		10%			10%	
	260	tanoak stems	density of dead tanoak stems per ha.		100			100	
	1000	tanoak stems/ha.	symptomatic tanoak stems		0			0	
22	0.29	ha.	dead tanoak stems		27			27	
	380	stems	percent tanoak stems that are dead		10%			10%	
	285	tanoak stems	density of dead tanoak stems per ha.		93			93	
	983	tanoak stems/ha.	symptomatic tanoak stems		0			0	
23	0.3	ha.	dead tanoak stems		58			58	
	506	stems	percent tanoak stems that are dead		18%			18%	
	328	tanoak stems	density of dead tanoak stems per ha.		193			193	
	1093	tanoak stems/ha.	symptomatic tanoak stems		0			0	

Table 3—*Multiple logistic regression model for binary disease outcome—the probability of individual stems developing* P. ramorum *symptoms.*

Variable	Value	Std. Error	F value	Prob (F)
Presence or sign of bark beetle activity on the stem in the previous year	0.87	0.22	16	<0.0001
Percent live foliage the previous year	−1.8	0.38	51.5	<0.0001

Null deviance of the model: 641 on 887 degrees of freedom

Residual deviance: 574 on 880 degrees of freedom

Discussion

The multiple logistic regression model enables us to evaluate the effect of stem attributes on the probability that an individual stem would develop stem or foliar symptoms through the period of the study. We examine the role of the two variables in the model and reflect on the variables that were not significant. We also discuss the potential role of the variables not measured in this study.

The two variables in this model may be indicators of undetected infection rather than predictors of symptom development. Both the "presence or sign of bark beetle activity in the previous sampling period" variable and the "percent live foliage the previous year" were highly significant. Unfortunately the design of this study does not enable us to determine the source of their significance.

As previously noted, beetle activity in tanoaks is rare. The association between beetle activity in the year preceding the appearance of symptoms indicates a potential role for beetles in the spread of the disease. It is also possible that *P. ramorum* infected tanoaks may become susceptible to beetle infestation prior to exhibiting visible stem canker bleeding. McPherson and others (2002) note that bark beetles are rare on living Coast live oak *(Quercus agrifolia)* in the absence of *P. ramorum* related bleeding stem cankers. Their study suggests a weak association between bleeding stem cankers in tanoak and the subsequent appearance of beetle infestation.

While our study does not provide proof of this assertion we suggest that "decrease in percent live foliage in the preceding year" variable in the model is more likely to be an artifact of undetected infection rather than a predictor of either visible foliage or bleeding stem canker symptom development. Flagging branch tips and scattered leaf death often precede the appearance of bleeding cankers (McPherson and others 2002). Thus the significance of decreased percent foliage observed in the year previous to the actual detection of *P. ramorum* symptoms has two likely causes. First, while we were able to detect a decrease in foliar cover we might have missed signs of foliar infection at crown height. We may have also missed signs of infection on abscised leaves at the base of all stems despite our efforts. Second in the initial phases of infection *P. ramorum* related stem cankers may sufficiently obstruct xylem flow to reduce foliar cover prior to causing the development of bleeding characteristics.

The dbh of a tree was not a significant predictor for the development of symptoms under our model nor were the height of a stem or its location in the canopy. Stems that grew individually or clumped were not significantly different in their probability of infection. Uninfected stems growing attached below breast height to *P. ramorum* infected stems were also no more likely to become infected. While we did no testing for the genetic relationships between our stems these results indicate that such testing may not be necessary.

It is clear that multiple factors contribute to the probability of tanoak stems becoming symptomatic. Our study compares mortality rates between Marin and Mendocino County. Mortality remained unchanged for the first two years of the study in the un-symptomatic Mendocino stands. By comparison mortality increased in all four Marin County plots. We find it important to acknowledge that our Marin plots represent a small portion of the possible area for testing. We acknowledge that we tested a limited number of tree characteristics (dbh, crown class) and did not measure a number of potentially important variables including seasonal water stress, stem water potential, age, root mass, leaf area, and genetics. The most important contribution of our model is the indication it provides of the stem factors that do not affect the probability of a stem becoming symptomatic. The significance of beetle presence in the sampling period prior to detecting symptoms suggests that the matter should be researched more thoroughly.

References

Akaike, H. 1973. **A new look at the statistical model identification.** IEEEE Trans. Automat. Control; 716-723.

Kashiwagi, J.H. 1985. **Soil Survey of Marin County.** Soil Conservation Service, U.S. Department of Agriculture, Washington, D.C.

McPherson, B.A.; Wood, D.; Storer, A.; Kelly, M.; Standiford, R. 2002. **Sudden Oak Death: Disease trends in Marin County Plots after One Year.** In: Standiford, R.B.; McCreary, D.; Purcell, K.L., technical coordinators. Proceedings of the fifth symposium on oak woodlands: oaks in California's changing landscape; 2001 October 22-25; San Diego, CA. Gen. Tech. Rep. PSW-GTR-184. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 751-764.

Rittiman, C.A.; Thorson, T. 1987. **Soil Survey of Mendocino County, California, Western Part.** Natural Resources Conservation Service, U.S. Department of Agriculture.

Rizzo, D.; Garbelotto, M. 2003. **Sudden oak death: endangering California and Oregon forest ecosystems.** Frontiers in Ecology and the Environment 1: 197-204.

Rizzo, D.; Garbelotto, M.; Davidson, J.; Slaughter, G.; Koike, S. 2002a. ***Phytophthora ramorum* and Sudden Oak Death: Host Relationships.** In: Standiford, R.B.; McCreary, D.; Purcell, K.L., technical coordinators. Proceedings of the fifth symposium on oak woodlands: oaks in California's changing landscape; 2001 October 22-25; San Diego, CA. Gen. Tech. Rep. PSW-GTR-184. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 733-740.

Stand Dynamics of Coast Redwood/Tanoak Forests Following Tanoak Decline[1]

Kristen M. Waring[2] and Kevin L. O'Hara[3]

Abstract

Current threats to North American forests increasingly include exotic tree pathogens that cause extensive mortality. In California, tanoak *(Lithocarpus densiflorus)* mortality has increased rapidly since 1995, due to *Phytophthora ramorum*, believed to be an introduced pathogen. Tanoak frequently grows as a major component of redwood forests along the northern California coast. This research examined the response of residual trees (primarily neighboring tanoak and redwood) to the decline of tanoak clumps. Redwood/tanoak forests currently infected with the pathogen and utilized in this study were located within the Marin Municipal Watershed District, Marin County, while uninfected sites with comparable forest age and species composition were located within Jackson State Demonstration Forest, Mendocino County. Stand reconstruction methodology was utilized to detect baseline growth patterns and changes related to the tanoak decline. Similar patterns of stand development appeared on both study sites, with codominant tanoak and suppressed redwoods occupying a middle canopy stratum below codominant redwoods. The lower strata, suppressed redwood may benefit the most from tanoak decline as they gain access to increased growing space.

Key words: exotic pathogen, forest pathogen, Lithocarpus densiflorus*,* Sequoia sempervirens, *stand development*

Introduction

Worldwide introductions of organisms beyond their natural range have increased tremendously in the past 100 years with increased globalization. Introduced organisms frequently do not establish and maintain populations. The environment is favorable for a handful of organisms however, due in part to changing landscapes and lack of co-evolution. Forest ecosystems in North America are no exception to these trends and continue to face new and emerging exotic pests.

Contributing to extensive mortality and decline of tree species throughout the United States are exotic insects and pathogens. A few of the most widely cited pathogens include chestnut blight (*Cryphonectria parasitica*), white pine blister rust (*Cronartium ribicola*), Dutch elm disease (*Ophiostoma ulmi* and *O. novo-ulmi*) and pine pitch canker (*Fusarium circinatum*) (Liebhold 1995). Chestnut blight, in particular, was primarily responsible for relegating American chestnut (*Castanea dentata*) to a mostly sapling size tree in the understory of eastern hardwood forests. In the 1990s, *Phytophthora ramorum* (a water mold) emerged as a new threat to

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Department of Environmental Science, Policy and Management, 137 Mulford Hall MC 3114, University of California, Berkeley, CA 94720-3114. email: kristen.waring@nature.berkeley.edu
[3] Department of Environmental Science, Policy and Management, 137 Mulford Hall MC 3114, University of California, Berkeley, CA 94720-3114. email: ohara@nature.berkeley.edu

forest trees in coastal California ecosystems (Rizzo and Garbelotto 2003). While many species of trees, shrubs, and herbs are infected with *P. ramorum*, tanoak *(Lithocarpus densiflorus)* has been hardest hit, suffering widespread mortality within infected ecosystems (Rizzo and Garbelotto 2003). Tanoak has long been considered a "weed species" by some foresters along the California coast, primarily because it tends to inhibit redwood *(Sequoia sempervirens)* growth following timber harvesting. However, it is a mast producing species that provides complexity to forest stands beneath the conifer strata. Tanoak serves to enhance wildlife habitat, prevent erosion and increase biodiversity within the forest stand.

Stand reconstruction provides a retrospective analysis of stand development patterns through time (Agrawal 1995, Oliver and Stephens 1977). Limitations to the use of stand reconstruction methods include small sample sizes (due to intensive sampling procedures) and lack of information regarding past mortality. Restoration of forest stands following invasion of exotic species is warranted in many ecosystems, and is becoming a growing concern in coastal California following *P. ramorum* invasion (Swiecki 2002). However, patterns of development in redwood/tanoak stands have not been described. Understanding these patterns is crucial background knowledge for directing future stand conditions and restoration activities. Without knowledge of historic stand development patterns and an understanding of how those patterns may be changing, land managers face a daunting task of meeting management objectives, such as restoration, without the necessary tools and quantitative basis for the development of such tools. Through use of reconstruction, the pest arrival time into the ecosystem can be estimated and the resulting changes through time can be compared to similar stands that are yet to be invaded by the pest. Our objectives were to describe development patterns in redwood/tanoak stands and show quantitatively how those patterns are changing in response to tanoak decline.

Methods

Research plots were located on Marin Municipal Watershed District land (MMWD), Marin County, California, and Jackson Demonstration State Forest (JDSF), Mendocino County, California. These sites are characterized by a Mediterranean climate, experiencing between 900 to 1100 mm average rainfall each year, the majority of which occurs between November and March. MMWD has been experiencing mortality of tanoak due to *P. ramorum* since the mid-1990s. Presence of *P. ramorum* within the study stands has recently been confirmed by DNA analysis (Spencer and O'Hara 2006). To date, *P. ramorum* has not been found within JDSF, and it was used as a control site for this research. A protocol was developed for use by all parties involved in this research to avoid spread of the pathogen between sites. All samples removed from each site were autoclaved upon arrival at the lab.

At each site, stands with similar management histories and species composition were selected. Target species composition for this research within the conifer component was at least 75 percent coast redwood, with Douglas-fir *(Pseudotsuga menziesii)* and occasional grand fir *(Abies grandis)* comprising the remainder of the conifers. Tanoak comprises the vast majority of the hardwood component within this forest type, with minor components of madrone *(Arbutus menziesii)* and bay *(Umbellaria californica)* (Zinke 1988). Four stands were sampled in the MMWD, and three stands were sampled in JDSF. Three circular plots ranging in size from $1/100^{th}$ ha to 1/10 ha were located in each stand, and all trees at least 1.37 m (breast height) tall were included in the plot inventory. Location of plots in MMWD was based on location of dead tanoak stems, with plots centered around groups of tanoak mortality. Plots in JDSF were of similar density and species composition as MMWD plots, centered around live tanoak. Between six and 10 sample trees were selected from each plot for more intensive sampling. Data collected from sample trees included total height, age and growth.

To complete age and growth analysis on the sample trees, two separate methods were undertaken at each site. At MMWD, non-destructive sampling was used, and each sample tree was cored at the base, breast height, and three meters up the bole. Cores were removed to the lab, where they were glued and sanded for growth and age analysis. At JDSF, destructive sampling took place during 2002, and involved felling the sample trees on six of the nine plots. After each tree was felled, bole and live crown length were recorded, in addition to five-year height growth. Cross sections were removed from the base, breast height, and at intervals along the tree bole. Interval distance varied by species and size, as there were restrictions on log length for utilization following sampling. Where the tree could not be utilized for timber, cross sections were removed every 3 meters. The cross sections were then taken to the lab, where they were planed and sanded. After preparation, each core and cross section was aged. All samples were sanded up to 600 grit and manually aged using a dissecting microscope.

Results

Results shown here compare one stand in MMWD with one stand in JDSF. Species composition of each stand met the original objectives, with at least 75 percent redwood in the conifer stratum and a majority of tanoak in the hardwood stratum. Species composition was similar between sites, particularly among trees greater then 10 cm DBH. Redwood plot density per hectare was 933 at MMWD and 567 at JDSF; for trees larger than 10 cm DBH, plot density per hectare was 333 at MMWD and 400 at JDSF. Plot density per hectare of tanoak at MMWD was 1700, compared to 833 at JDSF. Among tanoak trees larger than 10cm DBH, plot density per hectare was 400 at MMWD and 533 at JDSF. There were no large dead tanoak trees on plots at JDSF, compared with a plot density of 300 per hectare at MMWD.

The JDSF stand exhibited a higher site index (43 m, base 100) than MMWD (37 m, base 100) (Lindquist and Palley 1963). *Figure 1* shows the pattern of height development over time for three individual trees from one plot in the JDSF stand. A comparative plot from MMWD is shown in *figure 2*. The site at JDSF was logged approximately 80 years ago, which corresponds with initiation of the trees shown. Likewise, the MMWD stand was logged approximately 90 years ago, which also corresponds to the age of the main canopy trees.

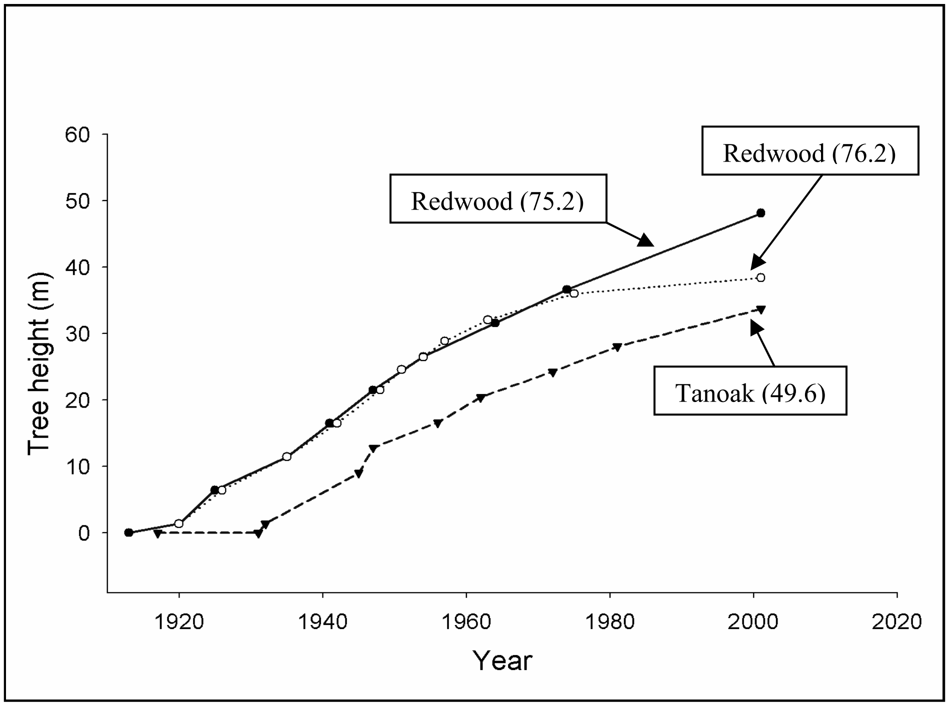


Figure 1—Stand development pattern for three trees at Jackson Demonstration State Forest, Mendocino County, CA. Number in parentheses is diameter at breast height (cm).

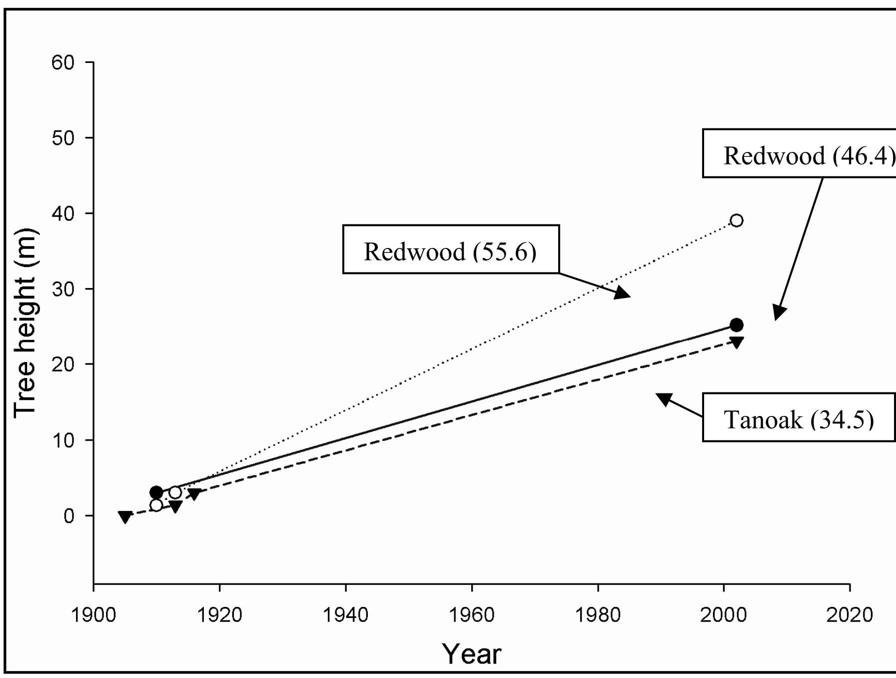


Figure 2—Development pattern of three trees at Marin Municipal Water District, Marin County, CA. Number in parentheses is diameter at breast height (cm).

Discussion

Similar patterns of species composition and stand development were found on both the JDSF and MMWD study sites. Even though these two sites are separated geographically and have slightly different site indices, it appears that they are comparable and the use of JDSF as a control in this research is valid. As *figure 1* shows, data from JDSF are more comprehensive; each point on the graph represents a data point from along the tree bole. Conversely, in *figure 2*, the only data points available are quite low on the tree bole and at the treetop. Data from mid-bole is lacking and obscures patterns of stand development occurring in this range. Quantitative analysis of stand dynamics will allow managers to understand ongoing changes in development and serve as predictive tools for restoration. Additionally pre-infection data from JDSF is invaluable if *P. ramorum* arrives at JDSF in the future.

Stratification within these redwood/tanoak stands began shortly after stand initiation. Both tanoak and redwood trees likely originated from sprouts following harvesting in the 1910s. While it appears in *figure 1* that tanoak cannot compete with redwood in height growth, this data is contrary to general tanoak growth patterns in which it competes heavily with conifers following stand initiation (Tappeiner and others 1990). *Figures 1* and *2* show similar patterns of height growth through time for competing redwoods until differentiation causes one redwood to fall behind. Other redwoods likely fell behind through time and eventually died leaving no current evidence of their existence. These patterns of development serve to create additional vertical complexity within the stand. These forests are often composed of three to four canopy strata. Remnant (residual) redwood forms an emergent stratum above the second-growth stand. The next two strata are shown in *figures 1* and *2*; the uppermost being formed by redwood, with tanoak forming a third, lower strata. The middle stratum frequently contains suppressed redwood trees that probably began falling behind shortly after stand initiation. A fourth strata may be found in the understory, composed of small tanoak and redwood sprouts and seedlings.

Figure 3 shows a hypothetical pathway for stand development following the loss of a large tanoak. The current height differentiation patterns will likely impact future development, with the tallest redwood least likely to benefit and the trees that are differentiating below the most dominant redwoods most likely to benefit from tanoak mortality. Furthermore, suppressed or intermediate trees may increase their height growth as resources become available. However, these trees will continue to compete with larger, overtopping redwoods and the outcome of this competition remains to be seen. This work represents the first stage of an ongoing project that will not only quantify stand development and residual tree response but also provide managers with guidelines for management and restoration in these forest systems.

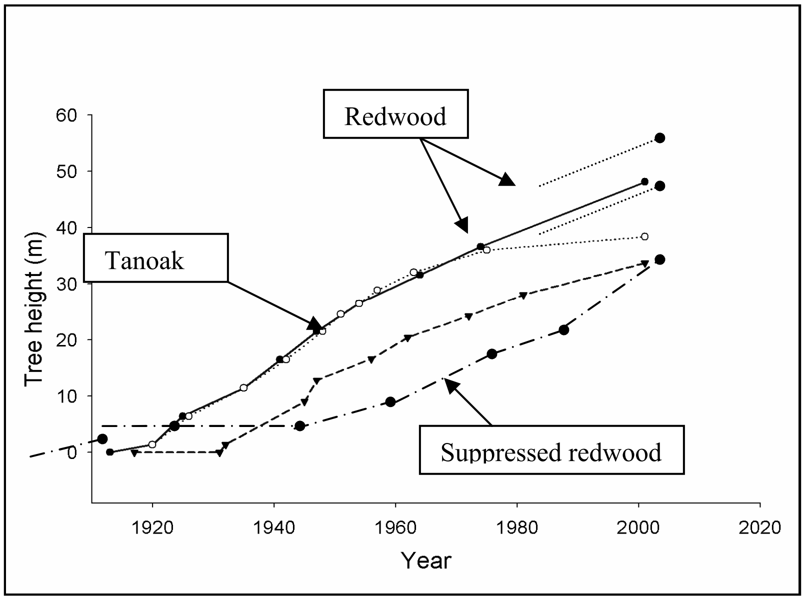


Figure 3—Hypothetical development of three redwood trees following loss of a tanoak.

References

Agrawal, A. 1995. **Use of dendrochronological methods to estimate an ecological impact date of the chestnut blight**. Virginia Journal of Science 46(1): 41-47.

Liebhold, A.M.; MacDonald, W.L.; Bergdahl, D.; Mastro, V.C. 1995**. Invasion by exotic forest pests: a threat to forest ecosystems**. Forest Science Monograph 30 41(2): 1-49.

Lindquist, J.L.; Palley, M.N. 1963. **Empirical yield tables for young-growth redwoo**d. Bulletin 796. California Agriculture Experiment Station; 47 p.

Oliver, C.D.; Stephens, E.P. 1977. **Reconstruction of a mixed-species forest in central New England**. Ecology 58: 562-572.

Rizzo, D.M.; Garbelotto, M. 2003. **Sudden oak death: endangering California and Oregon forest ecosystems**. Frontiers in Ecology 1(5): 197-205.

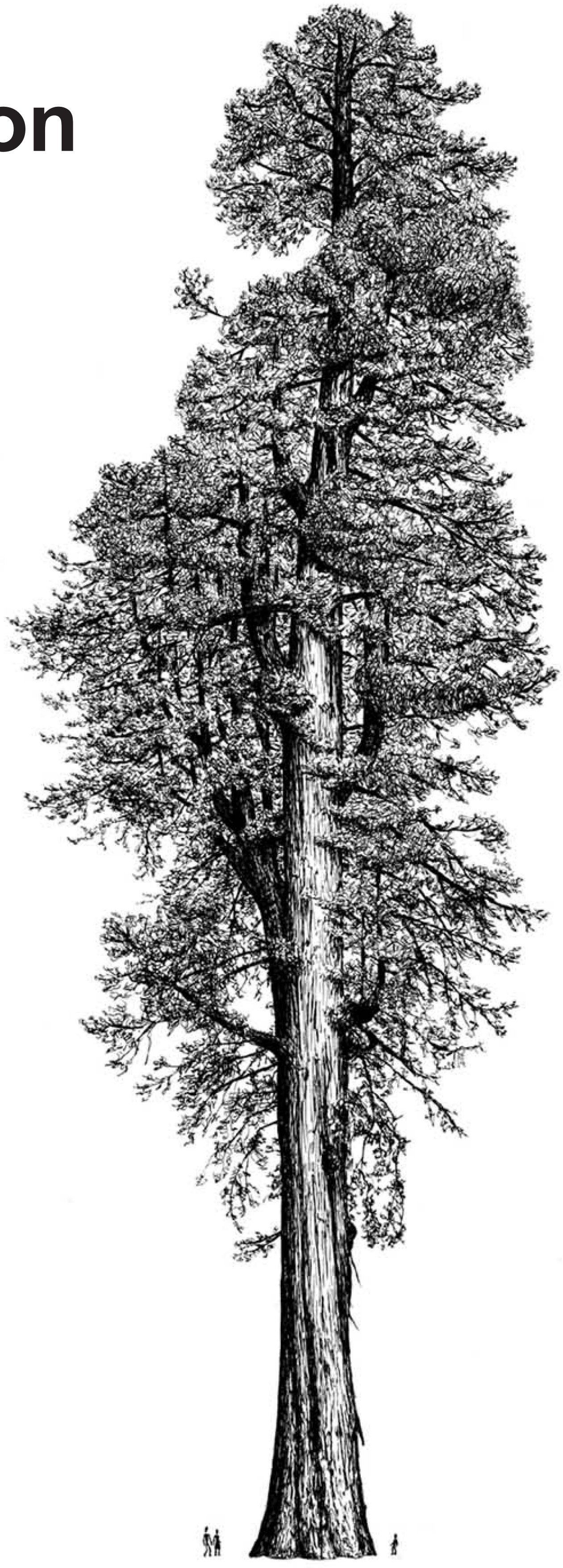
Spencer, M.; O'Hara, K.L. 2006. **A Multiple Logistic Regression Model for Predicting the Development of *Phytophthora ramorum* symptoms in Tanoak (*Lithocapus densiflorus*)**. In: Standiford, R.B.; Giusti, G.A.; Valachovic, Y.; Zielinski, W.J., technical coordinators. Proceedings of the Redwood Region Forest Science Symposium; 2004 March 15-17; Rohnert Park, CA Gen. Tech. Rep. PSW-GTR-194. Albany, CA: Pacific Southwest Research Station. USDA Forest Service; [this volume].

Swiecki, T. 2002. **Restoration of oak woodlands impacted by sudden oak death**. California Oak Mortality Task Force Executive Committee; 9 p.

Tappeiner, J.C., II; McDonald, P.M.; Roy, D.F. 1990. ***Lithocarpus densiflorus* (Hook&Arn.) Rehd.: Tanoak**. In: Burns, R.M.; Honkala, B.H., technical coordinators. Silvics of North America. Volume II. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture; 22 p.

Zinke, P.J. 1988. **The redwood forest and associated north coast forests**. In: Barbour, M.G.; Major, J., eds. Terrestrial vegetation of California. Exp. ed. Sacramento, CA: California Plant Society Press; 679-699.

Poster Session



Envisioning Ways Jackson Forest Could Demonstrate How to Revitalize the Region's Depleted Biological Heritage and Timber Production Capacity[1]

Kathy Bailey[2]

Jackson Demonstration State Forest is a 48,652-acre forest near Ft. Bragg managed by the California Department of Forestry and Fire Protection (CDF). It is, by far, the largest publicly-owned forest in the redwood region between San Francisco and Humboldt County offering unique conservation, recreation, and forest management demonstration and research opportunities. Approximately 10,000 acres have not been logged since the initial harvest entry 80 to 120 years ago. An additional total of 459 acres of old growth redwood and Douglas fir are isolated in 11 groves. These older forest components are unusual assets in a region that has been heavily logged.

A recent court decision set aside Jackson's new management plan and required revision of its environmental impact report (EIR). Further, the court ruled that the California Board of Forestry is the lead agency rather than CDF, indicating the Board is responsible for management decisions that are to be implemented by CDF.

The poster will use maps and satellite imagery to visualize opportunities to re-orient Jackson Forest toward research and demonstrations that help re-vitalize both the region's environment and its timber production capacity by addressing restoration of biologically and economically depleted stands to productivity.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] PO Box 256, Philo, CA 95466; (707) 895-3716. email: kb@pacific.net

Inner Gorge in Redwood Forests[1]

Julie A. Bawcom[2]

Coalescing landslide scars along eroding streams where the base level lowered, or a stream undercuts the toe of a deep-seated landslide form an inner gorge. Over-steep and undercut stream banks are susceptible to landslide failure by shallow debris sliding.

Inner gorge identification is important when planning timber harvest activities near watercourses. Aerial photo mapping of inner gorge in forested terrain is difficult due to tree cover. Anthropogenic alteration of the stream channels result in further complication. An inner gorge model of continuous symmetrical steep stream slopes is rarely found. Early logging that changed or significantly modified channel morphology produced inner gorge-like characteristics or obliterated inner gorge features altering many streams in the redwood region. Field mapping stream channels is the best method in determining the presence and historical alteration of an inner gorge.

Two field mapping terms are introduced, "Highly Modified Channel," and "Discontinuous Inner Gorge," which will aid in understanding and documenting the stability of slopes along redwood forest streams.

A highly modified channel (HMC) is defined as a channel changed by past in-stream logging that presently continues to erode exposing bare stream banks. Stream flow can intermittently be forced subsurface under buried logs. The stream channel is often "captured" within the old road fill and does not have the ability to undercut adjacent slopes. These modified channels are filled with legacy sediment from a previous era of logging practices.

A discontinuous inner gorge (DIG) occurs along a stream channel with intermittent sections exhibiting an inner gorge. It can be represented by an inactive inner gorge, or is often developed on only one side of the channel. This may be due to old railroad grades that break up the steep slope or to natural geomorphic irregularities. The one-sided inner gorge can be represented as a line with barbs pointing to the active side of the channel.

In both these specific geomorphic types the origin of shallow landsliding often originates upslope via slope creep, shallow soil slips or debris slides above the channels' influence. Management and stability considerations are different for a natural or continuous inner gorge.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] California Geological Survey, 17501 N. Hwy 101, Willits, CA 95490, (707) 456-1814. email: julie.bawcom@fire.ca.gov

Research at Jackson Demonstration State Forest—Building Partnerships for a Better Understanding of the Forest Environment[1]

William Baxter[2]

Jackson Demonstration State Forest (JDSF) has conducted and facilitated research in the redwood region for over 50 years. JDSF's mission of research and demonstrations helps to increase our understanding of redwood forest ecology and improve our forest management methods. Examples of research projects are presented to gain a better understanding of the diversity of JDSF research and the network of partnerships representing universities, public agencies, and wildland management professionals.

A wide variety of research is conducted on the 48,650 acre JDSF with almost endless possibilities for researchers. The predominantly redwood and Douglas-fir forest encompasses approximately 90 miles of streams with fish habitat and a mixture of forest types, age classes and management methods. JDSF is the largest state forest in California conducting research and demonstrations of forest management and it provides a unique opportunity to investigate the interaction of forest management with forest ecology in a public setting that is also used for recreation. There is ample opportunity to study the ranges of conditions and treatments including unit or landscape level treatments, and sufficient area for replications and control area.

Forest research often takes many years to offer reliable conclusions. Many of the historical research projects at JDSF, such as the Caspar Watershed Project initiated in 1962, provide baseline data that may be used as a foundation for future research. Demonstrations and experiments are used to initiate improvements in management methods and also to test the effects of existing standards. JDSF is also used to study regulatory standards prior to implementation to increase their effectiveness and reliability. This helps policy makers determine the balance between scientific knowledge, landowner rights and desires, and legal constraints.

JDSF is an ideal location for tours with universities and colleges, resource professionals and the public. Approximately 26 percent of the presentations at the Redwood Region Forest Science Symposium contain research associated with JDSF. Some examples of the types of research associated with JDSF are:

Watersheds:

- Caspar Creek Watershed Study – 150 research papers prepared since initiation in 1962.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] 802 North Main Street, Fort Bragg, CA 95437, (707) 964-5674. email: william.baxter@fire.ca.gov or brian.barrett@fire.ca.gov

- Sediment storage and transport on the Noyo River.
- Microclimate in riparian zones.

Forest Ecology:

- Fire history in coast redwood.
- Genetic study of clonal growth in coast redwood.
- Sudden Oak Death – Stand dynamics and spatial patterns of SOD symptoms.

Silviculture:

- Pre-commercial stocking control of coast redwood, 17 years of response.
- Commercial thinning growth and yield, 29 years of response.
- Variable retention modeling of management regimes in coast redwood.

Fisheries and Wildlife:

- Salmonid trends in Caspar Creek for 30 years.
- Large woody debris placement in Parlin, Caspar, and Hare Creek.
- Wildlife use of legacy trees in managed forests.

Erosion and physical processes:

- Landslide inventory of even-aged management.
- Erosion rates over millennial and decadal scales.
- Significance of suspended organic sediments.

Restoration and Monitoring:

- Road decommissioning: Demonstration of different methods.
- Exotic weed control – Participation in the International Broom Initiative.
- Road surface erosion measurements of course and fine sediment.

Research and demonstrations on JDSF improve our understanding of the forest environment and increase our ability to make informed management decisions. With an ever-increasing demand for the multitude of uses for forestland, information from research is more critical than ever. There is a history of success on JDSF that creates a foundation for the future. In-kind operational support is available through technical assistance and through housing at the Forest Learning Center with a goal of building partnerships for a better understanding of the forest environment.

Growth and Survival of Redwood and Douglas-Fir Seedlings Planted Under Different Overstory Removal Regimes[1]

William Bigg[2]

A twenty acre stand of mature second growth redwood was marked for selective thinning and clear cutting after an intensive cruise based on basal area. Four treatments: an uncut control, a clearcut, and 66 percent and 33 percent basal area thinning were done. In each treatment, four plots were planted with redwood seedlings and three with Douglas-fir seedlings. The growth and survival of these seedlings has been checked each year since planting in early 1997.

No Douglas-fir seedlings survived the first summer in the uncut area. After six years there was 79 percent, 68 percent and 47 percent survival in the clearcut, 66 percent removal and 33 percent removal areas respectively. After six years the average height of Douglas-fir seedlings was 231 cm, 150 cm and 120 cm in the clearcut, 66 percent removal and 33percent removal areas.

Thirty eight percent of the redwood seedlings survived to six years in the uncut area with survival continuing to decline over time. There was 78 percent, 97 percent and 83 percent survival in the clearcut, 66 percent removal and 33 percent removal areas respectively. Heights of seedlings in the uncut, clearcut, 66 percent removal and 33 percent removal areas were 84.7 cm, 100.1 cm, 112.5 cm and 146.8 cm after six years.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Department of Forestry and Watershed Management, Arcata, CA 95521, (707) 826-4220. email: wlb1@humboldt.edu

Habitat Characteristics and Spatial Extent of Burrow Systems of Point Arena Mountain Beavers on Managed Timberlands[1]

Sarah C. Billig[2] and Robert B. Douglas[3]

The Point Arena mountain beaver (PAMB) *(Aplodontia rufa nigra)* is one of seven subspecies of mountain beaver and is restricted in range to a small coastal area in northern California 62 square kilometers in size. Its restricted range and the lack of information regarding the population prompted the United States Fish and Wildlife Service (USFWS) to list the PAMB as endangered in 1991. Mountain beavers are generally found in cool microclimates with good drainage (Beier 1989, Pfeiffer 1953) and in areas with a higher proportion of small downed wood and soft soil (Hacker and Coblentz 1993). Because most information on mountain beaver ecology comes from studies on other subspecies, less is known about the Point Arena subspecies. PAMB live in underground burrow systems in areas with dense perennial vegetation and have been associated with three major habitats on forestlands: fresh water seep, alder/herbaceous ground cover, and conifers/sword fern (USFWS 1998). A majority of managed timberlands within the Point Arena mountain beaver assessment area are known to contain these habitat types. Our reasons for initiating a Point Arena mountain beaver habitat and spatial extent study were to examine burrow system habitat with respect to availability, develop hypotheses for future testing and modeling, and determine spatial extent of existing Point Arena mountain beaver burrows. Eventually, we hope to define potential habitat for Point Arena mountain beaver within forested systems with more specifics for inclusion in a Habitat Conservation Plan (HCP). We will use the spatial extent of burrow systems as a baseline for measurement within the HCP, and will re-measure spatial extent of each burrow system throughout the term of the HCP (with measurements taken every five years) to determine whether Point Arena mountain beaver burrow systems are changing in size.

To make a better assessment of stand-level characteristics associated with PAMB burrow systems, we sampled 22 habitat characteristics within known burrow systems ($n = 7$) on Mendocino Redwood Company lands. Plot measurements were made within PAMB burrow systems and around a random point 100 meters away from the edge of burrow systems. Proportion of stinging nettle ($P = 0.08$), sword fern ($P = 0.07$), and sorrel ($P = 0.08$) was greater in burrow systems than random plots. A greater proportion of these plants may be more indicative of potential habitat than other herbaceous plants. Though total number of trees was not different between burrow and random sites, there were fewer Douglas-fir ($P = 0.02$) and grand-fir ($P =$

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Wildlife Biologist, Mendocino Redwood Company, P.O. Box 390, Calpella, California 95418. email: sbillig@mendoco.com

[3] Wildlife Biologist, Mendocino Redwood Company, P.O. Box 489, Fort Bragg, California, 95437. email: rdouglas@mendoco.com

0.06) at burrow sites than random sites, however. Canopy cover (P = 0.20 center and P = 0.47 boundary) was not different between burrow and random sites, which may be due to an overestimation of alder canopy cover (using a spherical densiometer). We suspect burrow systems were generally closer to water than random sites and our inability to detect a difference (P = 0.138) may have been due to small sample size. Our results suggest the proportion of specific herbaceous species such as sword fern and stinging nettle may be an important factor explaining the location of Point Arena mountain beaver burrow systems. In addition, we characterized the spatial extent of each burrow system. Area of burrow systems ranged from 226.7 m^2 to 2,319.9 m^2 and had a, mean area of 685.8 m^2 (SE = 200.1). Burrow systems had a clumped distribution throughout the landscape and all were less than one-acre in size. Given estimates of density and home range sizes from other subspecies (Lovejoy and Black 1979, Martin 1971), each identified burrow system may only be occupied by 1 to 2 individuals, with each cluster of burrow systems constituting a potential “population.”

In the future, we will continue to collect spatial extent information to determine if known burrow systems are increasing or decreasing in size as vegetation around the burrow systems change, and survey potential PAMB habitat in and around timber harvest plans to monitor burrow system distribution over time. This information will be used in conservation of PAMB in managed timberlands and may assist the USFWS in future status reviews of the species.

Table 1—*Perimeter (m) and area (m^2 and acres) of PAMB burrows measured on MRC lands in Mendocino County, 2003.*

Burrow identification	Perimeter (m)	Area (m^2)	Area (acres)
Lower Alder Creek 1	133.9	360.1	0.09
Mills Creek 1	107.0	271.5	0.07
Mills Creek 2	79.6	280.0	0.07
Mallo Pass 1	123.6	385.4	0.10
Mallo Pass 2	343.8	1332.6	0.33
Mallo Pass 4	220.5	1227.2	0.30
Owl Creek 1	155.5	391.6	0.10
Owl Creek 2	305.8	2320.0	0.57
Owl Creek 3	121.2	378.9	0.09
Owl Creek 4	98.3	226.7	0.06
Owl Creek 5	119.4	370.7	0.09

References

Beier, P. 1989. **Use of habitat by mountain beaver in the Sierra Nevada.** Journal of Wildlife Management 53(3): 649-654.

Hacker, A. and B. Coblentz. 1993. Habitat **selection by mountain beavers recolonizing Oregon coast range clearcuts.** Journal of Wildlife Management 57(4): 847-853.

Lovejoy, B. and H. Black. 1979. **Population analysis of the mountain beaver, *Aplodontia rufa pacifica* in Western Oregon**. Northwest Science 53(2): 82-89.

Martin, P. 1971. **Movements and activities of the mountain beaver *(Aplodontia rufa)*.** Journal of Mammalogy 52(4): 717-723.

Pfeiffer, E. 1953. **Animals trapped in mountain beaver *(Aplodontia rufa)* runways, and the mountain beaver in captivity**. Journal of Mammalogy 34(3):396.

Riparian Flora Observed at Riparian Revegetation Projects in North Coastal California[1]

R. Katz,[2] M. Lennox,[2] D. Lewis,[2] R. Jackson,[3] J. Harper,[4] B. Allen-Diaz,[5] S. Larson,[2] and K. Tate[6]

The flora observed at revegetation sites is a management concern for many landowners and agency efforts involved in analyzing stream function, riparian restoration, native plant conservation, and natural resource management in California. There is the potential for competition from non-native species to displace individuals and populations of native riparian species. We have conducted a cross-sectional survey of 70 existing riparian revegetation projects in Marin, Sonoma and Mendocino Counties to document the resulting composition of flora. The project is a collaborative effort between the University of California Cooperative Extension, resource agencies, consultants, private landowners, and watershed groups, working in coastal California. Sites ranged from four to 39 years in project age and received treatments of exclusionary fencing and active planting or fencing alone. The poster will report and compare the presence of native and non-native plant species. In addition, we will share observations of species succession, as well as plot measurements of dominant species cover. This documentation is the first step in using the project database to inform effective design, installation, and maintenance of riparian revegetation projects.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] University of California Cooperative Extension, 133 Aviation Blvd. Suite 109, Santa Rosa, CA 95403, (707) 565-2621. email: mlennox@ucdavis.edu
[3] University of Wisconsin, Madison.
[4] University of California Cooperative Extension, Ukiah.
[5] University of California, Berkeley.
[6] University of California, Davis.

A Literature Review to Examine the Potential of Silviculture to Enhance the Formation of Old-Forest Characteristics in Coast Redwood Stands[1]

Christa M. Dagley[2] and Kevin L. O'Hara[2]

Restoration of old forests is an emerging forest management priority in the Pacific Northwest. A literature review was sponsored by Save-the-Redwoods League to identify and examine the potential of silviculture to enhance and accelerate the formation of old forest characteristics in coast redwood stands. This review focused on four questions: 1) What is the range of old forest characteristics for coast redwood? Can we quantify these characteristics and identify geographic differences in different populations? 2) How did old redwood forests develop? What are the roles of shrub and hardwood species, and fire? Do conifers typically grow at low densities throughout their development, or is there evidence of periods of intense competition and suppression? 3) What are the opportunities for silviculture to restore and maintain old forest characteristics? Potential management activities include regeneration, vegetation control, early or mid-rotation spacing treatments, pruning, fuel reduction, creation of wildlife habitat features, and prescribed fire. 4) There may be differences between socially preferred old forest characteristics and those characteristics supported by scientific data. Are there treatments that will be conducive to maintaining old forest health and integrity while meeting societal expectations? Key findings from past research and a list of research priorities are presented.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] University of California, 151 Hilgard Hall, MC3110, Berkeley, CA 94720-3110. email: cdagley@nature.berkeley.edu

Channel Incision and Suspended Sediment Delivery at Caspar Creek, Mendocino County, California[1]

Nicholas J. Dewey,[2] Thomas E. Lisle, and Leslie M. Reid

Tributary and headwater valleys in the Caspar Creek watershed, in coastal Mendocino County, California, show signs of incision along much of their lengths. Headcuts are numerous in each drainage. An episode of incision followed initial-entry logging which took place between 1860 and 1906. Another episode of incision cut into skid-trails created for second-entry logging in the 1970s. Gullies resulting from both of these episodes of incision are sensitive to hydrologic fluctuations and feature active headcuts, deepening plungepools, and unstable banks, which continue to contribute sediment to the Caspar Creek channel network.

Surveys indicate that bank retreat, plunge pool deepening, and headcut retreat all contributed sediment to the channels between 2000 and 2003. During the study period, bankwall retreat appears to be a more significant source of sediment than headwall retreat.

Stream gage records show that some channels consistently deliver higher levels of suspended sediment than others. On an annual to decadal time-scale, rates of suspended sediment delivery per unit area of catchment correlate better with gully length and exposed bank area, than with the volume of sediment delivered by landslide events, with total catchment area, or with peak storm flow per unit area.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Humboldt State University, Geology Department, Arcata, CA 95521 and USDA Forest Service, PSW Research Station, 1700 Bayview Ave., Arcata, CA 95521. email: nickjdewey@yahoo.com

Landscape and Site-Level Habitat Characteristics Surrounding Accipiter Nests on Managed Timberlands in the Central Coast Redwood Region[1]

Robert B. Douglas,[2] John Nickerson,[3] A. Scott Webb,[4] and Sarah C. Billig[5]

Accipiters such as the Cooper's Hawk *(Accipiter cooperii)* and Sharp-shinned Hawk *(A. striatus)* commonly nest in managed timberlands in the redwood region. A few published accounts describe accipiter nest sites in the western U. S. (Asay 1987; Moore and Henny 1983, 1984; Reynolds and others 1982; Siders and Kennedy 1996), however none exist for managed timberlands in northwestern California. Additionally, these species are listed by the California Department of Fish and Game as a Species of Special Concern primarily because of a lack of demographic information and continued habitat loss. Logging has been identified as one of several threats to accipiters and is a common activity in the redwood region that alters forest structure and plant species composition, and hence, may influence accipiter nest-site selection.

Since accipiter nest-site selection is not well understood on managed timberlands within the redwood region, resource managers often have little information to use for making management decisions. In 2001, Mendocino Redwood Company (MRC) initiated a project to identify accipiter nest sites throughout its forestlands using broadcast surveys, stand searches, and incidental sightings. In an effort to better understand accipiter nest-site selection, a project was designed to examine forest attributes surrounding nest trees at several scales. Here, we report preliminary results for habitat characteristics surrounding Cooper's Hawk (n = 7) and Sharp-shinned Hawk (n = 1) nests at stand and landscape scales. At the stand scale, we measured vegetative characteristics around nest trees and random trees (located 125 m away) within 0.05, 0.01, and 0.2 hectare radius plots; and at the landscape scale, we calculated percent area of specific forest structure classes and vegetation types surrounding nest trees within 7, 29, 51, 203, and 458 hectare circles.

Site- and landscape-level results suggest that tanoak and large conifers are important elements at nest sites. All nests were located in tanoak trees in the upper size classes (>39. 37 cm dbh; *table 1*) and nest sites had higher mean hardwood basal

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 17-17, 2004, Rohnert Park, California

[2] Wildlife Biologist, Mendocino Redwood Company, P.O. Box 489, Fort Bragg, CA 95437. email: rdouglas@mendoco.com

[3] GIS/Inventory Manager, Mendocino Redwood Company, P.O. Box 390, Calpella, CA 95418. email: jnickerson@mendoco.com

[4] GIS Analyst, Mendocino Redwood Company, P.O. Box 390, Calpella, CA 95418. email: swebb@mendoco.com

[5] Wildlife Biologist, Mendocino Redwood Company, P.O. Box 390, Calpella, CA 95418. email: sbillig@mendoco.com

area and higher mean hardwood tree density (1.23±0.09 m^2/plot, 22.14±3.24 trees/plot), primarily in small-to-medium sized tanoak, than random sites (0.93±0.19 m^2/plot, 14.57±3.88 trees/plot, respectively). Although mean conifer basal area and mean conifer tree density were lower at nest sites (0.96±0.54 m^2/plot, 4.00±1.60 trees/plot) than random sites (1.13±0.41 m^2/plot, 7.71±1.74 trees/plot), most nest sites contained a few large conifers that contributed to most of the conifer basal area. Accipiter nest sites contained relatively high densities of small and medium hardwoods and a low density of conifer in all size classes. Landscape analysis also showed that the mean percent area of mixed hardwood/conifer vegetation type was highest at the smallest spatial scale and declined with increasing area around nest sites, suggesting that accipiters may be selecting this habitat type at the nest-stand scale.

Historical records indicate that accipiters have nested in other tree species besides tanoak on MRC lands; however, the disproportionate discovery of nests in tanoak (and stands dominated by tanoak) since the inception of this study suggests that birds are cuing on tree and/or stand characteristics conducive for nesting. Moore and Henny (1983) found that Cooper's hawks in Oregon typically selected Douglas-fir, as opposed to six other species of conifer, to build nests because this species often contained mistletoe brooms which provide a solid substrate for nest building. Wiggers and Kritz (1991) also documented Cooper's hawks nesting primarily in deformed trees below canopy in Missouri. While accipiters on MRC timberlands may be selecting trees suitable for constructing nests, they may also be selecting stands with tanoak and a few emergent conifers because these stands may provide increased access to prey and/or have higher prey availability, as well as provide protection from potential predators.

Moreover, since these results are based on a small sample size, our interpretation of the data should be regarded as working hypotheses subject to change as more information is collected. We are continuing to survey for accipiters in timber harvest plans, provide protective buffers around nests, and measure nest-site characteristics. If our results do indeed represent a phenomenon common in the redwood region, then some level of hardwood and emergent conifer retention may be an important element in conserving accipiters on commercial timberlands in Mendocino County.

Table 1—*Summary statistics for eight accipiter nests found in tanoak on MRC lands in Mendocino County.*

Nest Tree	Mean	SE
Diameter at Breast Height (cm)	49.31	2.72
Tree Height (m)	22.42	1.99
Nest Height (m)	16.25	2.09
Height to Crown Base (m)	9.42	1.54
Elevation (m)	350.12	65.38
Distance to Watercourse (m)	136.68	29.27
Canopy Cover (percent)	93.55	0.94
Position on Slope (percent)	55.28	5.91

References

Asay, C. 1987. **Habitat and productivity of Cooper's Hawks nesting in California.** California Fish and Game 73(2): 80-87.

Moore, K.R.; Henny, C.J. 1983. **Nest site characteristics of three coexisting Accipiter hawks in northeastern Oregon.** Journal of Raptor Research 17(3): 65-76.

Moore, K.R.; Henny, C.J. 1984. **Age-specific productivity and nest site characteristics of Cooper's Hawks (*Accipiter cooperii*).** Northwest Science 58: 290-299.

Reynolds, R.T.; Meslow, E.C.; Wight, H.M. 1982. **Nesting habitat of coexisting accipiters in Oregon.** Journal of Wildlife Management 46: 124-138.

Siders, M.S.; Kennedy, P.L. 1996. **Forest structural characteristics of accipiter nesting habitat: is there an allometric relationship?** Condor 98(1): 123-132.

Wigger, E.P.; Kritz, K.J. 1991. **Comparison of nesting habitat of coexisting Sharp-shinned and Cooper's hawks in Missouri.** Wilson Bulletin 103(4): 568-577.

Restoring Riparian Conditions Along Valley Floors Affected by Multiple Coarse-Grained Flood Deposits: An Approach from Bull Creek, Humboldt Redwoods State Park[1]

Rocco Fiori,[2] Ruth Goodfield,[3] and Patrick Vaughan[4]

Sedimentation from the 1955 and 1964 floods aggraded portions of the Bull Creek valley by several meters and widened the channel bed as much as 100 percent. Past efforts to restore riparian conditions, based largely on vegetative characteristics alone, have had limited success. Limiting factors include poor water holding capacity within expansive coarse deposits (d50 >10 mm), seasonal rainfall, continued flooding, high solar exposure and channel migration and sedimentation related to the legacy of poor land use in the upper watershed before acquisition by the park in the mid-1960s.

California State Parks and Cooperators are now beginning to restore riparian areas using a process based approach. These efforts follow watershed improvement projects, begun in 1997, that have greatly reduced the density of hydrologically linked roads in the upper watershed and are intended to decrease sediment production, attenuate flood peaks and reduce the zone of annually mobilized bed sediment.

In several opportunistic locations within affected floodplains and channel margins we have created planting islands. In these areas we have mechanically shifted the coarse deposits to a finer texture and increased the organic content to a depth approaching the summer low flow elevation. Preliminary results suggest that soil moisture and over summer survival is significantly greater for seedlings planted in islands compared to untreated deposits. By strategically locating planting islands, where fine sediments are accreting and other geomorphic indicators suggest conditions are favorable to longer-term riparian vegetation, naturally occurring riparian areas can be expanded and linked with other islands.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Engineering Geologist, California State Parks. email: rfiori@parks.ca.gov
[3] Watershed Restorationist, Eel River Improvement Group. email: rgoodfield@sabre.net
[4] Engineering Geologist, California State Parks. email: pvaug@parks.ca.gov

Determining the Distribution of Three Amphibian "Species of Concern"[1]

Matthew O. Goldsworthy[2]

Currently the distribution of tailed frogs, red-legged frogs and southern torrent salamanders in Mendocino County is largely unknown. Baseline data on the distribution of these species was collected in 2003. Approximately 56,000 acres or 25 percent of Mendocino Redwood Company's (MRC) ownership was surveyed in 2003. The remainder of the MRC ownership will be surveyed during the next three years and monitored throughout the next 80 years.

Approximately 33 percent of the Calwater planning watersheds surveyed were determined to support red-legged frog reproduction. Fifteen documented breeding sites were located throughout five planning watersheds. The majority of breeding sites found contained little canopy cover (<40 percent) and were located within floodplains (57 percent). Many of the documented breeding sites were manmade (42 percent).

Tailed frog surveys were conducted at 148 sites, of which 24 sites yielded detections (16 percent). The species was detected within approximately 40 percent of the planning watersheds surveyed. The majority (96 percent) of detections were from watercourses which did not have southerly aspect.

Southern torrent salamander distribution surveys were conducted after the first few rains of 2003. There were 108 sites surveyed and eight sites (seven percent of sites) yielded detections of the species. The distribution of southern torrent salamanders does not appear to be as widespread as it is in Humboldt County.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Aquatic Biologist, Mendocino Redwood Company, PO Box 489, Fort Bragg, CA. 95437, (707) 962-2909. email: mgoldsworthy@mendoco.com

The Effect of Overstory Canopy Alterations on Air Temperature in a Managed Redwood Forest[1]

Elizabeth Wilson Hadley and William Bigg[2]

This study was conducted to determine if there is a relationship between air temperature and overstory canopy, if there is an effect on the air temperature at the center of the buffer strip with a 50-foot reduction in width, and with an overstory removal to bring the canopy down to 85 percent.

Following a control period, the canopy surrounding three circular study plots were cut first to create a 200-foot buffer from the center, second to bring the buffer width down to 150 feet, and third, to bring the overstory canopy to a level of 85 percent closure.

The daily mean, minimum, and maximum temperature difference between 52 sampling points and the center untouched 50-foot area of the buffer zone was found following every logging event. No significant changes in the air temperature at the center were found as a result of any of the harvests ($p < 0.001$). There was a strong relationship between the maximum daily air temperature differences and overstory canopy as measured by the solar pathfinder ($R2 = 0.66$).

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Department of Forestry and Watershed Management, Arcata, CA 95521, (707) 826-4220. email: wlb1@humboldt.edu

A Comparison of 10 Techniques Used to Estimate Canopy Interception[1]

Todd A. Hamilton and William Bigg[2]

An eighty-year coast redwood *(Sequoia sempervirens)* and Douglas-fir *(Psuedotsuga menziesii var. menziesii)* forest type was used to compare ten techniques of measuring canopy interception. Measurements for all techniques were taken in a series of four treatments: 1st) before operations, 2nd) after surrounding area was clearcut to retain 200-foot radial circles, 3rd) after surrounding area was clearcut to retain 150-foot radial circles, and 4th) after the 150-foot radial circles were thinned to retain 85 percent canopy interception.

Thinning 33 percent of the basal area changed canopy closure from 91 percent to 86 percent and canopy cover from 96 percent to 85 percent. Vertical sighting tube showed the greatest change in canopy interception after thinning, whereas gap fraction (5'175˚) showed the least change in canopy interception. The strongest correlation ($r^2 = 0.96$) was between hemispherical photographs with different view angles (15'75˚ and 15'175˚). Average change in canopy interception increased 12.7 percent from view angles of 5º to 175º (slope = 0.07). Stem structure remained about the same though the average number of trees per acre went from 202 to 150.

It is recommended that techniques of canopy cover are used in stands with interception less than 65 percent and techniques of canopy closure are used in stands with interception greater than 65 percent.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Department of Forestry and Watershed Management, Arcata, CA 95521, (707) 826-4220. email: wlb1@humboldt.edu

Redwood and Douglas-Fir Stumpage Price Trends in Coastal California[1]

Richard B. Standiford[2]

The North Coast is California's largest timber harvesting area. Harvest in this area, stretching from Sonoma to Del Norte Counties, ranged from 1.4 billion board feet in 1978 to 520 million board feet in 2002. This represents approximately 30 percent of the state's timber harvest. Old growth percent has decreased from 70 percent in the mid-1970s to less than 10 percent currently.

Real prices for young growth redwood and Douglas-fir in the North Coast have shown an increasing trend since 1978. Annual real price increases over the past 14 years have averaged 5.3 percent for redwood, and 4.1 percent for Douglas-fir. Despite these trends, there have been tremendous annual price fluctuations, reflecting volatility and uncertainty for landowners. Over the past 14 years, changes from year to year varied from −40 percent to +74 percent.

The unique niche for redwood products and high consumer acceptance is expected to continue the strong price trends for this species. The Douglas-fir prices are expected to be more of a commodity, tied in closely with pine and Douglas-fir from other regions. This information will be useful in modeling anticipated affects of supply changes on product prices.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Center for Forestry, University of California, Berkeley, CA 94720, (510) 643-5428. email: standifo@nature.berkeley.edu

Large Woody Debris and Pool Dynamics in the Caspar Creek Experimental Watershed, Northern California[1]

Sue Hilton[2] and Leslie Reid[2]

Although large woody debris (LWD) is now widely recognized as an important contributor to channel habitat, LWD dynamics are still poorly understood. This poster describes interim results of a study of inputs, breakage, transport, and decay of LWD in the mainstem channels in the two Caspar Creek Experimental Watersheds. LWD volumes and characteristics differ in the two reaches. Here we discuss possible causes for the differences, how the differences affect pools in the reaches, and what might happen to LWD in these reaches over time.

The two Caspar Creek experimental watersheds supported approximately 100-year-old second-growth redwood *(Sequioa sempervirens)* forest in 1968, when road building began for the first Caspar creek experiment. From 1971 to 1973, approximately 65 percent of the timber volume in the entire South Fork Watershed was removed in a series of selection cuts, with logs tractor-yarded to stream-adjacent roads. Twenty years later, 50 percent of the North Fork was harvested in a series of small clearcuts. Logs were cable yarded from ridgetop roads, and 100' selectively logged buffers were left along both sides of the mainstem.

LWD was inventoried in an 1800 m reach in the North Fork mainstem in 1986, 1994, and 1996, and in an equivalent reach in the South Fork in 1994 and 1996 (Keppeler 1996, O'Connor and Ziemer 1989, Surfleet 1996). In 1998, all recent LWD pieces >0.2 m diameter and 2 m long and all old LWD pieces >0.5 m^3 were tagged and measured in the two reaches. Those logs were resurveyed in 1999, and all pieces larger than the minimum new piece size were tagged at that time. Both reaches were remapped and remeasured in 2002 and 2004.

Since 1998, the volume of LWD in the North Fork study reach has remained more than twice that in the South Fork reach. We identified three potential causes for this difference. First, much of the existing wood in the South Fork channel was removed during the 1970s logging, and that LWD may not yet have been replenished. Second, there was significant blowdown along the North Fork in buffer strips left during the 1990s logging, and much of that wood entered the channel. Inputs into the South Fork channel during the same period were much lower. Third, since the 1970s logging, stands adjacent to the South Fork channel have not been capable of producing as much LWD as those along the North Fork. In 2004, trees within 100' of the South Fork channel were smaller and shorter than North Fork trees, and a higher proportion of the trees were species that are relatively resistant to blowdown

---

[1] This poster was presented at Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California

[2] Hydrologist and Research Geologist, USDA Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, 1700 Bayview, Arcata, CA 95521. email: shilton@fs.fed.us and lreid@fs.fed.us, respectively.

(redwood) or fairly short-lived in the channel (alder).

We compared LWD volumes in the 800 m downstream subreach of each study reach to pool volumes in a 500 m section of that subreach (Lisle and Hilton 1999). Total pool volumes in the two reaches are similar, but pool LWD relationships differ. Most of the pool volume in the North Fork reach is in pools associated with second-growth LWD, and that proportion, as well as the total volume, increased from 1994 to 1996 in response to increased LWD from buffer strip blowdown *(fig. 1)*. In the South Fork, almost half of the pool volume is associated with residual old-growth pieces, and about 10 percent is in non-wood pools. Changes in second-growth associated pool volume appear to be related to changes in total LWD.

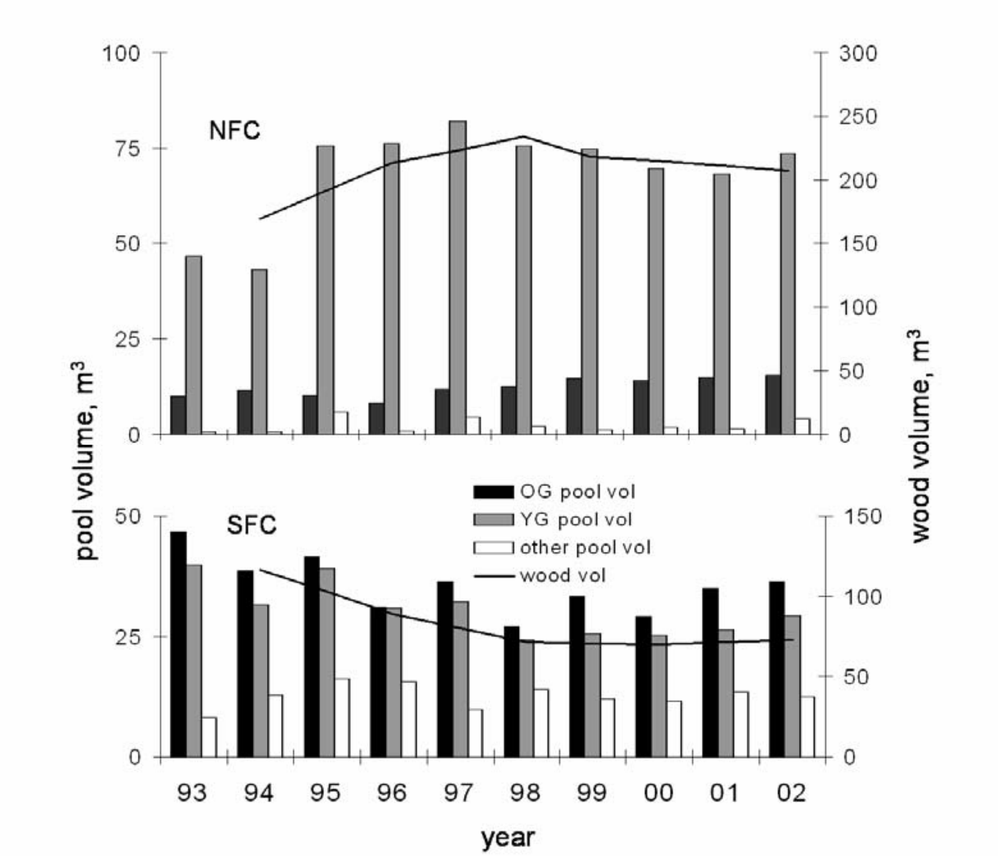


Figure 1—LWD Volume in the downstream 800 m of each study reach compared to pool volume in the downstream 500 m by type of pool each year, 1993 to 2002.

In the South Fork, 30 percent of all pieces and 16 percent of the volume was gone, moved, or broken from 1998 to 2004, although the total volume changed by less than five percent. In the North Fork, 24 percent of the pieces and 17 percent of the volume had changed, while the total volume increased, due primarily to input from snags. Monitoring continues at both tributaries. Data will be used to create a yearly wood budget for the channels, and will be used in combination with stand growth and wood input models to project the future of LWD in these reaches.

References

Keppeler, E. 1996. Caspar Creek Large Organic Debris revisited. Unpublished report supplied by author.

Lisle, Thomas E., Hilton, Sue 1999. **Fine bed material in pools of natural gravel bed channels**. Water Resources Research 35(4): 1291-1304.

O'Connor M.D., Ziemer, R.R. 1989 **Coarse woody debris in a second-growth Sequoia Sempervirens forest stream**. In: Proceedings of the California riparian systems conference 1988 September 22-24; Davis, CA. General Technical Report PSW-GTR-110; Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 165-171.

Surfleet, C.G., Ziemer, R.R. 1996. **Effects of forest harvesting on large organic debris in coastal streams**. In: Proceedings of the conference on coast redwood ecology and management, 1996 June 18-20; Arcata CA. Arcata, CA: Humboldt State University; 134-136.

Adapting Silvicultural Practices to Respond to Changing Societal Demands for Forest Resource Management[1]

Stephen R. Horner[2]

Today's California forest managers are being asked to feed society's increasing appetite for forest products while providing exceptional protection for all non-commodity forest resources. Truly sustainable forestry—which requires balancing environmental protection, socio-economic factors and financial considerations of the landowner—must include progressive silvicultural options that improve forest growth and make for efficient timber harvest.

The results from a growth and yield study of a 20-year old pure redwood plantation on a highly productive site in the Redwood Region that has experienced 50+ percent increases in stand growth as a result of intensive management are used as a basis for predicting commodity production within a watershed using a high-yield/habitat protection matrix approach. The results suggest that both high growth rates of stands of commercial forest tree species and protection of terrestrial and aquatic plant and animal species can be achieved, but economic viability of the firm practicing sustainable forestry—and hence the economic success of a harvest-dependent community—is highly dependent upon operational flexibility. Given that commodity yield and habitat and species protection can be achieved, careful harvest unit planning is recommended but must be supported by operations windows that maximize harvest during dry seasons and reconfigured buffer zone boundaries that balance resource protection and efficient harvest operation.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Scotia Pacific Company LLC, P.O. Box 712, Scotia, CA, 95565, (707) 764-4128. email: horner@scopac.com

Riparian Vegetation Recovery Following Road Decommissioning[1]

Emily King[2]

A Humboldt State University Graduate Thesis is in progress in Redwood National and State Parks to study the regrowth of riparian vegetation following road decommissioning. In the last 25 years, the decommissioning of roads in the park has been quite intensive, both in number and complexity. Large areas are disturbed as the landscape is recontoured. Large woody debris is used to stabilize slopes, but little is done to restore vegetation after the decommissioning is complete. The methods of decommissioning roads have been changed over the years resulting in different vegetational stages and speeds of recovery. Long term effects of the disturbance are not well understood and little vegetation monitoring has been done in the decommissioned reaches.

This study will look at perennial stream crossings on decommissioned roads from different years to determine the riparian vegetation regeneration. It will also look at paired sites on the same stream reaches that were decommissioned using different methods to see if there is a difference in the composition of vegetation. Controls for each site will be located in adjacent, undisturbed areas. One goal of this study is to determine which methods of road decommissioning result in riparian vegetation regrowth that is most like the control areas.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] College of Natural Resources, Department of Forestry and Watershed Management, Humboldt State University, 2244 First Street, McKinleyville, CA 95521, (707) 839-5629. email: eking@wso.williams.edu

Are Suspended Sediment Yields a Function of Land Use in the Elk River Watershed, Humboldt County?[1]

Peter Manka and C. Hobart Perry[2]

The majority of watersheds in the redwood region on the North Coast of California are listed by the USEPA as impaired by excessive sediment. High volumes of sediment interfere with the migration and spawning of threatened and endangered salmonids. Excessive sediment in streams may also contaminate drinking water supplies, cause channel aggradation, or change flood frequency and extent. Turbidity threshold sampling is estimating the annual suspended sediment yields of three tributaries of the Elk River in Humboldt County, northern California. These streams have similar drainage areas (one to two square miles), similar geologies, and significant differences in land management. Little South Fork Elk River drains the largely undisturbed, old-growth forest of the Headwaters Forest Reserve. Most of the Corrigan Creek watershed is a mid-successional forest approximately 60 years in age. The South Branch of the North Fork of Elk River underwent extensive even-aged management approximately 13 years ago. We are exploring the differences in total suspended sediment yields and periodicity of suspended sediment movement. These differences provide insight into "background" levels of suspended sediment and the impacts of land management on suspended sediment production. The data may also provide information on the mechanisms and rates of recovery from sediment impairment.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Department of Forestry and Watershed Management, Humboldt State University, Arcata, CA 95521, (707) 826-5622. email: chp1@humboldt.edu

Effect of 70 Years of Recreational Car Camping on Vigor of Old-Growth Coast Redwood and Douglas-Fir[1]

Steven R. Martin,[2] John D. Stuart,[3] Portia Halbert,[4] and Mark A. Rizzardi[5]

Introduction

Recreationists have been car camping at Blooms Creek campground in Big Basin Redwoods state park annually for 70 years. Park managers are interested in better understanding the effects that such long-term recreational use may have on the health and vigor of the forest overstory.

The Problem

Trampling and vehicle use are major causes of impacts to soils in wildland recreation areas, including soil compaction, increased soil density, reduced macroporosity and aeration, changes in soil structure and stability, reduced litter and humus layers, reduced infiltration rates, increased runoff and erosion, changes in soil temperature regimes, a reduction in soil microorganisms, and changes in soil chemistry and available nutrients; these impacts are usually assumed to adversely affect plant vigor.

Concern over the effects of long-term recreational trampling on the vigor of mature redwoods has existed since the early days of the redwood state parks, but the few investigations into those perceived impacts have been inconclusive. This investigation seeks to measure more directly and quantitatively the vigor of mature redwoods and Douglas-fir in a campground that has withstood more than 70 years of recreational trampling.

Methods

The study was conducted in Big Basin Redwoods State Park. Study sites were located in the Blooms Creek campground and along the relatively untrammeled Opal Creek. The Blooms Creek campground was opened in the 1930s and consists of 48

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Professor, Dept. of Environmental and Natural Resource Sciences, Humboldt State University, Arcata, CA 95521, (707) 826-5637. email: srm1@humboldt.edu

[3] Professor, Dept. of Forestry and Watershed Management, Humboldt State University, Arcata, CA, 95521, (707) 826-3823. email: jds2@humboldt.edu

[4] Resource Management Specialist, Big Basin Redwoods State Park, 600 Ocean Street, Santa Cruz, CA 95060.

[5] Associate Professor, Department of Mathematics, Humboldt Sate University, Arcata, CA.

drive-in campsites and four walk-in sites. The Opal Creek site has a narrow, lightly used trail running through it with no evidence of off-trail use, and served as the control site.

Study sites were located in alluvial redwood forests with redwood and Douglas-fir as the dominant or co-dominant species. In each of the two study sites, we sampled all of the redwood and Douglas-fir trees that were emergent or dominant crown class. This resulted in sample sizes of 35 redwood and 22 Douglas-fir trees sampled in the campground, and 19 redwood and 12 Douglas-fir trees in the control site.

For each sampled tree we measured height and crown length, circumference of the tree, sapwood thickness and bark thickness. We used these measurements to calculate live crown percent, diameter, radius inside the bark, total basal area at breast height, heartwood basal area, and sapwood basal area. We then calculated crown length to sapwood basal area (CL/SBA) as an index measure of crown density, our chosen indicator of tree vigor.

Results

A Mann-Whitney test for equality of medians was performed to compare redwoods in the campground and control sites, and also to compare Douglas-firs in both sites. For redwoods, there was no significant difference (at $\alpha = .05$) in height, diameter, crown length, live crown percent, sapwood basal area, or the CL/SBA index measure of crown density between the campground and control study sites. For Douglas-fir, the only significant differences between the campground and control sites were for length of live crown and live crown percent, with Douglas-firs in the control site possessing a longer live crown and a larger live crown percent; there was no significant difference for crown density.

To further test for a campground effect controlling for tree height and diameter, separate linear regression models were constructed for each tree species. There was no statistically significant campground effect for the redwoods ($P = 0.79$) and Douglas-firs ($P = 0.94$) after controlling for tree height and diameter.

(Model: $\log(\text{CSI}) = \beta 0 + \beta 1 \log(\text{Diameter}) + \beta 2 \log(\text{Height}) + \beta 3\ \text{Campground}$)

Conclusions

Despite intuitive concerns expressed by academics and resource managers alike regarding the detrimental effects of recreational trampling on the health and vigor of mature trees in recreational areas, our study of coast redwoods and Douglas-firs in a California state park recreational campground used annually for more than 70 years found no significant difference in crown sparseness between overstory redwoods and Douglas-firs in the campground with those in an untrampled control plot.

Canopy Closure and Soil Moisture in a Second-Growth Redwood Forest[1]

Justin Mercer and William Bigg[2]

This study examined the effects of second-growth redwood canopy on growing-season soil moisture conditions for redwood seedlings. Two sites were utilized to measure soil moisture content over the duration of the growing-season at varying levels of canopy closure. The first, a transect, beginning in a clear-cut and extending into an uncut second-growth redwood stand, was used to compare soil moisture depletion across the forest edge. The second employed a meadow to generate the same comparisons in a forest gap. Canopy measurements were derived from hemispherical photographs; with soil moisture data collected from the upper 20 cm of the topsoil and measured as gravimetric water.

The results of the study indicate a strong correlation between the extent of measured canopy and soil moisture conditions. Increasing exposure to sunlight correlated to lower levels of soil moisture throughout the growing season, significant differences in the rate of depletion, reduced minimum water balances, and shorter potential growing seasons.

Differences in soil moisture conditions were subtle amongst plots in the gap and the uncut forest, with more extreme differences evident for the clear-cut plots. Conditions in the clear-cut differed significantly from every other plot, including the gap center, with soil moisture depletion rates allowing a considerably shorter potential growing season.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] Department of Forestry and Watershed Management, Arcata, CA 95521, (707) 826-4220. email: wlb1@humboldt.edu

Riparian Zone Management and Analysis of Flood Hazard in Urban and Rural Areas[1]

Matthew D. O'Connor[2]

Riparian vegetation and woody debris in stream channels has long been recognized by engineers and landowners to contribute to flood hazard. Excessive vegetation and debris may cause local flooding, bank erosion, and channel avulsion. These disturbance processes are part of the natural pattern of disturbance that creates diversity in aquatic and riparian habitat. In urbanized and rural residential areas, however, these disturbance processes are a threat to property.

Historically, both private and public flood control efforts included removal of vegetation and woody debris from channels. State and Federal environmental regulations now limit degradation of aquatic habitat incidental to flood control efforts. Many streams in the region have thickly vegetated riparian zones where riparian vegetation is encroaching upon the channel. Debris jams also form where woody material is present. Flow resistance in these areas is high, increasing flood hazards.

An analysis of flood hydraulics and flow resistance in one northern California stream demonstrates the potential effect of riparian vegetation and woody debris on flood hazards. This case study demonstrates a method of analysis of flood hazard, paying special attention to quantification of the flow resistance associated with vegetation.

Field surveys of live stems and woody debris in the Elk River were used to determine the flow resistance of woody material in a low gradient coastal alluvial river channel. This stream is prone to flooding and has relatively dense stands of alder along the banks. Mean channel slope is about 0.001, mean bankfull width is about 50 ft (15 m), and mean bankfull depth is 11 ft (3.4 m). The channel is sand bedded, with some fine gravel, and has some deep pools associated with LWD.

The approach developed by Shields and Gippel (1995) was used as the basis for developing quantitative estimates of flow resistance. Two types of data were collected. Over a distance of about 2000 ft (580 m) partitioned into two reaches, detailed measurements of all stems (live or dead) in the bankfull channel >0.1 ft (3 cm) diameter were measured to determine the area of each LWD piece or live stem perpendicular to flow. These data allow a direct computation of flow resistance. Over a distance of 18,000 ft (about five km), the diameters of live and dead woody stems were measured along the channel centerline. This more extensive, less detailed survey was intended to estimate reach-scale variation of flow resistance of woody material and to provide perspective on spatial variation of channel conveyance and flood hazards as a function of the abundance of woody material in the channel. In

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] O'Connor Environmental, Inc., P.O. Box 794, Healdsburg, CA 95448, (707) 431-2810. email: matto@oe-i.com

both cases, the character of woody material was also observed. This included whether the material was live or dead, and standing or downed.

Flow resistance (Manning's n) of woody material in two reaches measured in detail was 0.034 for the reach with relatively few pieces of woody material crossing the channel and 0.046 in the reach with more obstructions in the channel. Center line surveys over a larger area suggest that woody material roughness values found in the reaches are likely much higher than those measured in the two detailed study reaches.

Estimated flow resistance of woody material in the Elk River study reach varied from about n = 0.03 to about 0.07. These n values are relatively high, and would represent the total flow resistance in many gravel bed streams in the region. Flow measurements and gaging records in Reach 6 suggest flow resistance at bankfull flow to be in the range n = 0.08 – 0.14. Resistance from woody material appears to represent about roughly 30 to 50 percent of the estimated total.

In a field study similar to that of Shields and Gippel (1995), Manga and Kirchener (2000) found that woody debris at their site provided about 50 percent of total flow resistance despite the fact that wood covered only about two percent of the channel surface area. Additional sources of flow resistance in the Elk River study reach would include the stream bed and banks, and bends in the river. At Shields and Gippel's study site on the Obion River in Tennessee, measured flow resistance attributed to the stream bed was about n = 0.042. The Obion River has very similar geometry and sediment size distributions to the study reach on the Elk River. Using the Obion River bed resistance of about n = 0.04 to represent bed resistance in the Elk River study reach would suggest n values in the range of about 0.07 to 0.11. Obion River reaches were straight. Some reaches on the Elk River are straight, but several reaches contain sharp bends that would likely add substantially to total flow resistance.

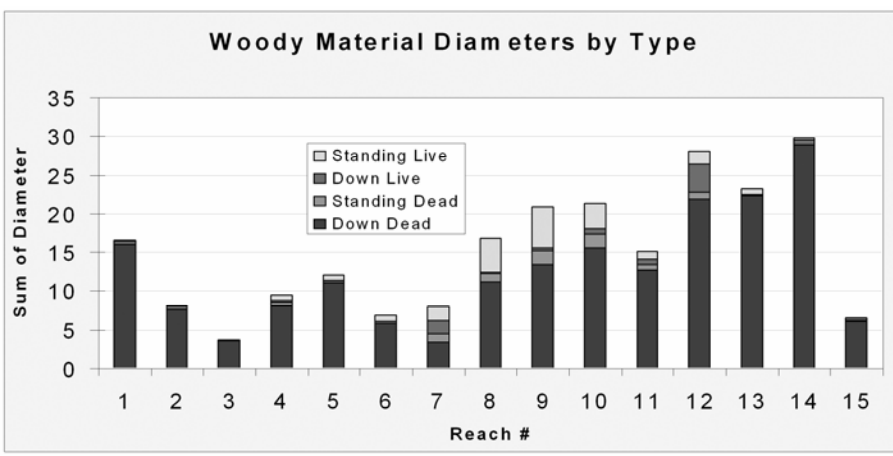


Figure 1—The character of woody stems expressed as the sum of stem diameters in units of feet is shown for consecutive 1,000 ft reaches in Elk River, Humboldt County, California. Down dead woody debris is the dominant type, however, live woody stems are significant in many reaches, particularly 7 through 10. Live stems are primarily found in dense stands of willow and alder on the channel banks.

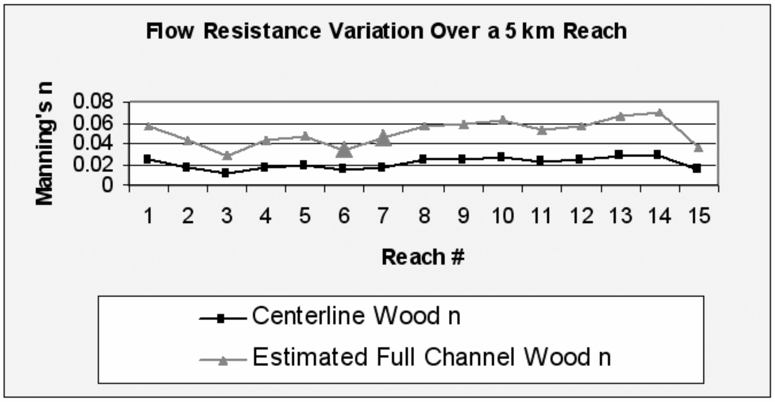


Figure 2—Estimated flow resistance expressed as Manning's n (a roughness coefficient commonly used in hydraulic calculations) is shown for consecutive 1,000 ft reaches in Elk River, Humboldt County, California. Two estimates are provided. The lower line in the graph is derived from field data for woody stems measured throughout the reach along the channel centerline only; it represents a minimum estimate of resistance due to woody stems. The second estimate (the upper line in the graph) is an extrapolation derived from the relationship between observations along the channel center line and full-channel observations in reaches 6 and 7; this estimate better represents the likely magnitude of flow resistance associated with woody debris throughout the study reach.

References

Manga, M.; Kirchener, J.W. 2000. **Stress partitioning in streams by large woody debris.** Water Resources Research. 36(8): 2373-2379.

Shields, F.D.; Gippel, C.J. 1995. **Prediction of effects of woody debris removal on flow resistance.** Journal of Hydraulic Engineering 121(4): 341-372.

A Tree-Marking Procedure for Variable-Density Thinning—Applications to Old-Forest Redwood Restoration[1]

Kevin L. O'Hara[2] and Christa M. Dagley[2]

Key words: restoration, Sequoia sempervirens, *silviculture, stand density, stocking control, variable-density thinning*

Variable-density thinning is an operation intended to enhance short- and long-term stand structural diversity. By thinning to variable densities within a single stand, the resultant structure includes areas at wide spacings, unthinned areas, and areas with intermediate spacings. For some management objectives—such as for promoting wildlife habitat or old forest characteristics—this stand heterogeneity is desirable. Consistent implementation of variable-density thinning operations is difficult because variability is difficult to quantify in an operational setting: tree markers or thinners will have to monitor their activities so their marking or thinning is repeatable and consistent over time and space. To assist with variable-density thinning treatments, we developed a procedure for marking young stands (<20 years). This procedure assumes a target density is known: in this case the target is 50 trees/ac (124 trees/ha) plus an assumed mortality rate of 50 percent (resultant target density = 75 trees/ac (185 trees/ha)). Markers work in cells equal to 1/N acre where N = target density. A random number from zero to three is chosen that provides the number of residual trees for that cell. The resultant density should approach 75 trees/ac. This procedure was tested for variable-density thinning treatments in Del Norte County with target densities of 75 and 150 trees/ac (185 to 371 trees/ha).

Post-thinning results generally indicated the procedure resulted in densities close to targets *(table 1)*. The "high density" treatment at Cougar Ridge was low but this was largely due to the low numbers of trees in the blocks randomly selected for this treatment. Otherwise the procedure appears to be useful for achieving targets with variable-density thinning. However, the method is operationally difficult to implement because of the need to recognize cell sizes that vary with target density and generate a random number in the field. For research purposes, this method appears to be a promising method of achieving target densities and obtaining a variable-density structure. These treatments and the controls will be monitored to describe their future development.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] ESPM, University of California, Berkeley, CA 94720-3110, (510) 642-2127. email: ohara@nature.berkeley.edu

Table 1—*Post-thinning densities from variable-density thinning at Mill Creek in Del Norte County. TPA = trees per acre.*

Treatment site	Treatment	Target TPA	Actual TPA
Childs Hill	Control	N/A	657
	High density	150	166
	Low density	75	78
Cougar Ridge	Control	N/A	682
	High density	150	113
	Low density	75	67
Moratorium	Control	N/A	1357
	High density	150	140
	Low density	75	84

The California Geological Survey and the Review of Timber Harvest Plans in Redwood Forests[1]

Mark G. Smelser[2]

The California Geological Survey (CGS) has been assessing geologic issues associated with timber harvesting in the north coast redwood region since the implementation of the Z'Berg-Nejedly Forest Practice Act in 1975, and is currently a representative of the Interdisciplinary Review Team as defined in the Forest Practice Rules. CGS' Forest and Watershed Geology (FWG) Program provides technical information and advice about landslides, erosion, sedimentation, and other geologic hazards to the California Department of Forestry and Fire Protection (CDF), other state agencies, industries, and the public where proposed activities may affect public safety, soil productivity, water quality, and fish habitat. Within the FWG program, licensed engineering geologists provide independent technical review of proposed Timber Harvest Plans (THPs), Non-Industrial Timberland Management Plans (NTMPs), and other regional-scale land management projects submitted to CDF under the Forest Practice Rules. The geologic evaluation of a THP by CGS follows a systematic approach conducted in accordance with standards of professional practice and scientific accountability. The evaluation includes a desk review of the submitted THP or NTMP, review of pertinent geologic maps and reports, review of historic aerial photographs, participation in the pre-harvest field inspection as staff are available, and if geologic concerns are noted a written report is prepared that often includes specific recommendations for the plan submitter to address.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] California Geological Survey, 2120 Campton Road, Suite E, Eureka, CA, 95033, (707) 441-5743. email: msmelser@consrv.ca.gov

A Context for Cumulative Watershed Effects in Redwood Forests[1]

Thomas E. Spittler[2]

Coastal northern California redwood forests are controlled by the complex interaction of geology, hydrology/climatology, and biology, each of which is in constant flux. Earthquakes, uplift and subsidence, sea level change, storms, droughts, floods, fires, and the growth, death, and evolution of species affect the region. Simple deterministic models cannot integrate these dynamic watershed components.

Comparing what is present today with conceptual “natural” conditions is speculation that cannot be supported. Vegetation management by Native California societies for the past 3,000 to 10,000 years produced a lower average biomass, more open forests and greater streamflow than exists where fires are suppressed. Plant and animal communities evolved under this managed fire regime. Because of the dynamic complexity of redwood forest watersheds and the lack of documentation on Native Californian management, models may not be capable of identifying desired conditions at a site-specific level.

Science has the tools to detect changes and rates of changes in the individual components that define a watershed. Integrating observations on changes to see if and how fast areas are trending toward the complex diversity of conditions anticipated to result from modeling may be one approach to understanding cumulative watershed effects in redwood forest watersheds.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] California Geological Survey, 135 Ridgway, Santa Rosa, California, 94502, (707) 576-2949. email: tom.spittler@fire.ca.gov

Adaptive Management Monitoring of Spotted Owls[1]

Mike Stephens,[2] Larry Irwin,[3] Dennis Rock,[3] and Suzanne Rock[4]

Extensive public and private forests occur in early to mid-successional stages from northern California through Washington. Many landowners and agencies are expected to manipulate many such forests over the next few decades to reduce fuel loads or increase growth via intermediate silvicultural treatments such as thinning or partial harvesting. We initiated an extensive, cooperative project to monitor responses of both the northern and California subspecies of spotted owls to applications of such less intensive forestry practices. Owls are fitted with 7 to 8 g back-pack radio transmitters and signals are recorded using handheld, directional Yagi antennae and portable receivers. The study employs a repeated, or multiple study-area approach, for which data will be combined via meta-analyses. The project combines both repeated observational experiments and manipulative experiments. The primary objectives allow comparisons among owl foraging use of forest stands with and without previous silvicultural treatments, and before-versus-after silvicultural treatments. Products involve resource selection function models, which can be used as decision-support tools for predicting owl responses to various silvicultural treatments in managed forests.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] National Council for Air and Stream Improvement, P.O. Box 751, Fort Bragg, CA 95437, (707) 937-2548. email: strix@mcn.org
[3] National Council for Air and Stream Improvement, P.O. Box 458, Corvallis, OR 97339.
[4] Northwest Economic Associates, 12009 N.E. 99th St. Suite1410, Vancouver, WA 98682-2434.

The Effects of Harvest History on the Lichens and Bryophytes of the Arcata Community and Jacoby Creek Forests[1]

Sunny Bennett[2]

Historical and modern logging, including single tree selection (removal of 20 to 50 percent of individual stems within a unit) and patch cuts (removal of all stems in a unit, clearcut), have resulted in a mosaic of different aged stands within the Arcata Community and Jacoby Creek Forests. Continuous monitoring plots have been established as part of an ecologically sensitive, long-term management plan. I conducted surveys to identify the lichens and bryophytes present in the plots and to determine any effects of harvest history on species richness and abundance.

One hundred and fifty species were identified, including two rare lichens and one rare moss. Average abundance for most species was low (<1 percent cover) due to low frequency of occurrence. When grouped by harvest history, single tree selection plots had the highest mean number of species, and patch cut plots had the lowest mean number of species. Historically logged plots had the highest number of unique species (36), while single tree selection and patch cut plots had equal number of unique species (7).

Single tree selection is probably a better method of timber harvest than patch cutting to promote species diversity in the Arcata Community and Jacoby Creek Forests.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] City of Arcata, 736 F Street, Arcata, CA 95521, (707) 834-5104. email: sunny@asis.com

A Tale of 10 Snags[1]

David L. Suddjian[2] and Thomas Sutfin[3]

During a 1995 50-acre timber harvest conducted on the Soquel Demonstration State Forest (SDSF) in Santa Cruz County, snags were created from ten large standing Douglas-fir trees to provide increased nest, roost and foraging sites for birds. The trees were 34 to 50 inches in diameter at breast height. A bird study was conducted prior to the 1995 timber harvest to asses its effects on breeding bird populations, with subsequent bird surveys conducted in 1996, 1998, and 2001.

By 2001, all ten snags showed evidence of use by cavity-nesting birds, including Pygmy Nuthatch, Pileated Woodpecker (a newcomer to the Soquel Creek watershed), Western Screech-Owl, Northern Pygmy-Owl and Northern Saw-whet Owl. Bird population changes also were noted. For example, by 2001, Acorn Woodpeckers, absent on the 1995 pre-harvest surveys, occurred at 67 percent of the survey stations; Hairy Woodpeckers occurred at 40 percent more survey stations; and Northern Flickers went from nearly absent in 1995 to being present at 50 percent of the stations. Although the creation of the snags alone probably did not change the bird population in the SDSF, the abundance of cavities, active nests, and foraging evidence in the snags by 2001, suggests the snag management program played a big role.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] 801 Monterey Avenue, Capitola, CA 95010, (831) 476-9062. email: dsuddjian@aol.com

[3] California Department of Forestry and Fire Protection, 4750 Soquel-San Jose Road, Soquel, CA 95073, (831) 475-8643. email: thomas.sutfin@fire.ca.gov

Effects of Forest Management in the Caspar Creek Experimental Watersheds[1]

Jack Lewis,[2] Elizabeth Keppeler,[2] and Tom Lisle[2]

Caspar Creek Experimental Watersheds were established in 1962 as a cooperative effort between the California Department of Forestry and Fire Protection and the USDA Forest Service Pacific Southwest Research Station to research the effects of forest management on streamflow, sedimentation, and erosion in the rainfall-dominated, forested watersheds of north coastal California. The project has evolved from a simple paired watershed study into one of the most comprehensive and detailed investigations of its kind. In 1962, weirs were installed for measuring streamflow and sediment loads on the North and South Forks. From 1971 to 1973, 50 percent of the timber volume in the South Fork was selectively cut and tractor yarded, and the untreated North Fork was retained as a control. In 1986, thirteen new gaging stations were installed in the North Fork Basin and three unlogged tributaries served as controls when 48 percent of the North Fork basin was clearcut and cable yarded between 1989 and 1991. Ten new gaging sites in the South Fork will be used to assess impacts of selection harvest and road rehabilitation on tractor-logged terrain. The scope of research in the watershed has expanded beyond hydrological studies to include geomorphological, ecological, silvicultural, and biological investigations.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] U.S. Forest Service, Pacific Southwest Research Station, 1700 Bayview Drive, Arcata, CA 95521, (707) 825-2929. email: jlewis01@fs.fed.us

Evaluation of Low-Altitude Vertical Aerial Videography as a Method for Identifying and Estimating Abundance of Residual Trees[1]

Linda M. Miller,[2] Scott D. Osborn,[2] and David J. Lancaster[2]

Low-altitude color aerial video was acquired within the northern section of the Redwood Region in Humboldt and Del Norte Counties, northwestern California. Four interpreters viewed a sample of video and identified residual trees within one-third hectare circular plots. Each sample plot was ground-truthed and residuals were identified and mapped.

Error matrices presented indicated that identification of residuals was not highly accurate for individual trees, nor consistent among interpreters. However, for three of four interpreters, linear regressions of number of interpreter-identified residuals per plot versus number of field-identified residuals per plot had significant slopes ($p < 0.005$). Coefficients of determination were 0.23, 0.22, and 0.41 for the three interpreters. Interpreters were not very successful at identifying old-growth legacy trees in video, and clonal rings of redwood trees were often mistaken for residuals in video due to large crown diameter.

It was concluded that low-altitude color aerial videography may not be accurate enough for identification of individual residuals, but could be used effectively to estimate abundance of residuals in an area of interest, for example, a watershed. Double-sampling and training of interpreters based on lessons learned in this study could improve prediction intervals of future studies. Identification of legacy trees in aerial video needs further investigation.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] California Department of Fish and Game, 619 Second Street, Eureka, CA 95501, 707-441-2091. email: lmiller@dfg.ca.gov

Pathogenicity and Distribution of Native and Nonnative *Phytophthora* Species on *Sequoia sempervirens*[1]

Camille E. Jensen[2] and David M. Rizzo[2]

The pathogen *Phytophthora ramorum* is known for causing widespread mortality on coast live oak *(Quercus agrifolia)* and tanoak *(Lithocarpus densiflorus*) in California's coastal forests. However, it is not clear how this exotic pathogen will affect coastal redwoods *(Sequoia sempervirens)*. Additionally, two possibly native species of *Phytophthora* (*P. nemorosa* and *P. pseudosyringae*) may play a role in these redwoods systems. We are examining the potential pathogenicity and distribution of these three species on redwoods. In 2003, 54 plots were established throughout the geographic range of redwoods. Symptomatic tissue of redwood and bay laurel *(Umbellularia californica)* trees in the plots were sampled and tested for Phytophthora species using cultural and molecular techniques. Preliminary results show understory foliage of redwoods to be common substrate for *P. ramorum* in forests with high inoculum levels based on sampling from bay laurel leaves, but no associated redwood mortality has been observed. Both *P. ramorum* and *P. nemorosa* have been isolated from symptomatic tissue of coast redwoods, but have not been cultured from bark. *P. pseudosyringae* has not been isolated from coast redwoods. Future research is focused on 1) disease progression of each of the *Phytophthora* species on redwood, and 2) the interaction of these pathogens on redwood.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Department of Plant Pathology, University of California, Davis, One Shields Ave., Davis, CA 95616, (530)-754-9894. email: jensencamille@gmail.com

Silvicultural Treatments to Control Stump Sprout Density in Coast Redwoods[1]

Christopher R. Keyes[2] and Peter J. Matzka[2]

Unmanaged stump sprouts of redwood *(Sequoia sempervirens)* concentrate stems in a small area, contributing to an aggregated spatial distribution of trees that potentially diminishes stand productivity and wood qualities, contributes to tree instability, mandates stand thinning at an early age, and indirectly contributes to the occurrence of cambium feeding damage caused by black bears. Past studies in redwood and other species have shown that sprouts are directly influenced by stump size, age, and height, and that sprout density and vigor can be affected by partial sprout removal, thermal wounding, shading, exposure to hormones, and bark removal. This study has been established for the purpose of identifying practical and efficient techniques for the operational control of immediate post-harvest stump sprouting capacity (basal bud management) and early sprout density management. Differences in the sprouting response of redwood stumps to treatments designed to debilitate the capacity of stumps to produce sprouts—including varying stump heights, mechanical stump scarification, bud incineration, and mechanical sprout removal—are being quantified. Equipment tested for feasibility in this study includes a portable high-temperature torch for bud and sprout incineration, motorized cutting tools for sprout removal, and the innovative use of chainsaws in the modification of stump morphology.

---

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] Department of Forestry and Watershed Management, Humboldt State University, Arcata, California 95521.

Habitat Restoration, Landowner Outreach, and Enhancement of Russian River Coho Populations in Northern California[1]

Paul Olin,[2] David Lewis, Janet Moore, Sarah Nossaman, Bob Coey, Brett Wilson, and Derek Acomb

The Russian River and tributaries in Northern California historically provided habitat for sustainable populations of anadromous fish including coho, chinook and steelhead trout. Activities in the watershed, including gravel mining, construction of dams, agricultural expansion and urban development have degraded habitat such that all these fish populations are in decline, and they are listed as threatened or endangered under federal and state law.

To reverse these declining population trends, a ten-year effort by the California Department of Fish and Game has assessed over 800 miles of stream habitat throughout the watershed and identified priority restoration needs. Since 1997, University of California Sea Grant and Cooperative Extension outreach programs have created a knowledgeable cadre of riparian landowners committed to habitat protection and restoration to promote recovery of salmon. These programs have been responsible for completion of habitat assessments and more than 55 priority restoration projects throughout the watershed. A recently initiated wild captive broodstock program is an integral component of these efforts, and offspring from these fish will be used to re-establish coho in streams providing good habitat. A comprehensive release and monitoring program is being developed to allow for evaluation of this restoration and enhancement program.

---

[1] This paper was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.

[2] University of California Cooperative Extension, 133 Aviation Boulevard, Suite 109, Santa Rosa CA, 95403-2894, (707) 565-2621. email: pgolin@ucdavis.edu

Conservation Value Assessment of the California North Coastal Basin by Using Special Elements and Focal Species[1]

Doug Smith,[2] Curtice Jacoby,[2] Chris Trudel,[2] Robert Brothers[2]

A conservation model was developed which identifies conservation priorities for the California North Coastal Basin. This process is a conservation value assessment based on conservation biology principles using a computer based GIS to analyze and map applicable information. It is based on assessing special elements, modeling focal species habitat, representing all ecotypes, and creating a community network. Special areas containing significant ecological elements, suitable habitat for focal species, and secure habitat for large carnivore and ungulates are identified. These areas are combined to create a landscape design that identifies conservation priorities. Core conservation areas and stepping stones imbedded in landscape linkages connecting and buffering them are identified. Actions to protect these areas will be taken through collaborative projects with key conservation organizations consisting of promoting best management practices, restoration, conservation easements, and fee title purchase. LEGACY—The Landscape Connection is a 501(c)(3) non-profit organization dedicated to the maintenance and restoration of the ecological integrity of northwest California, using GIS as our primary tool. Although we work with many advocacy groups, we do not engage in advocacy ourselves.

[1] This poster was presented at the Redwood Science Symposium: What does the future hold? March 15-17, 2004, Rohnert Park, California.
[2] LEGACY-The Landscape Connection, P.O. Box 59, Arcata, CA 95518. email: jacoby@legacy-tlc.org and website: www.legacy-tlc.org