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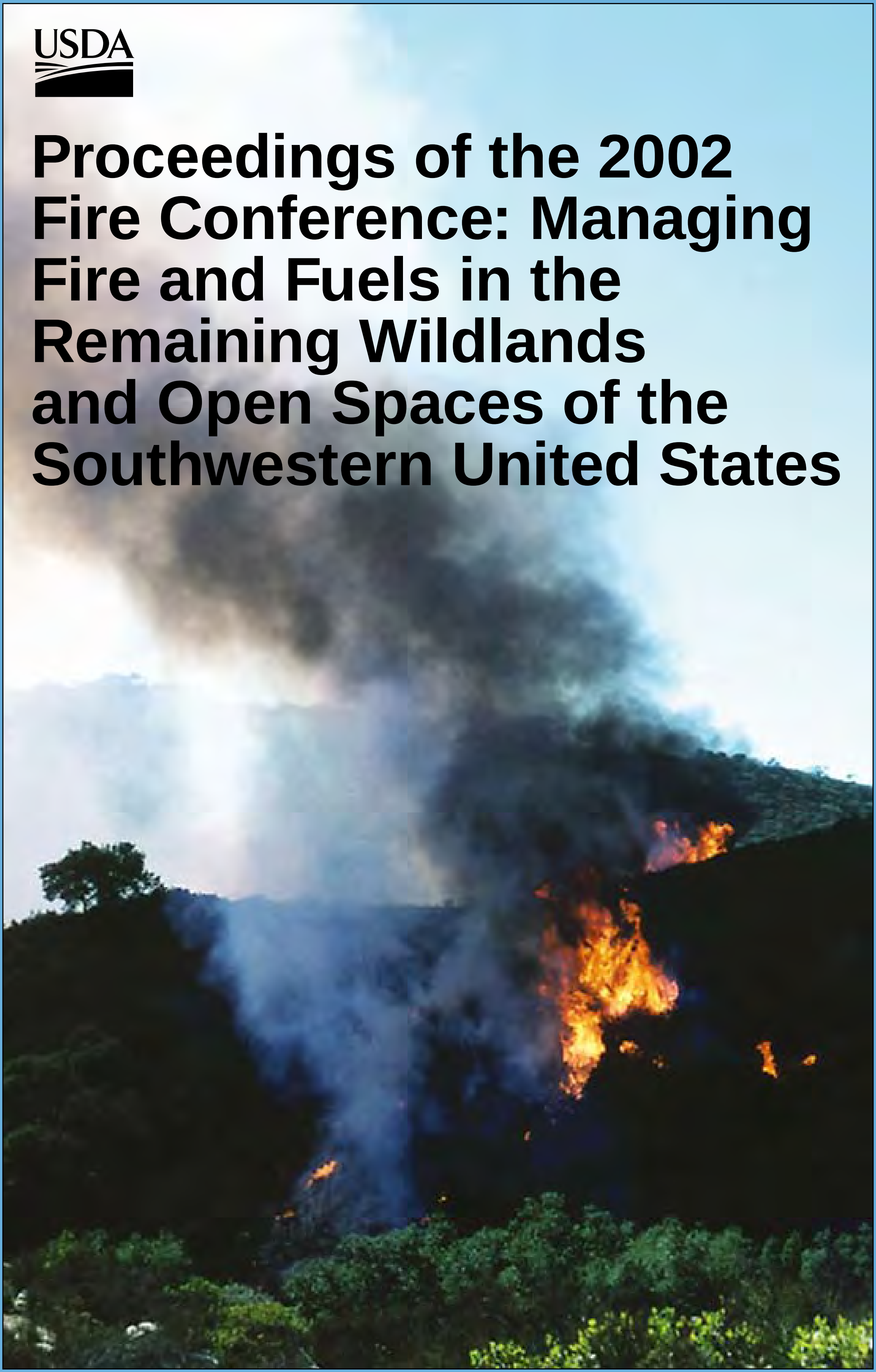
Proceedings of the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States



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Proceedings of the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States

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San Diego, California

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U.S. Department of Agriculture
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Abstract

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Many issues confront scientists, land managers, policymakers, and the public who deal with or are affected by management of fire and fuels across the southwestern United States (Utah, Colorado, Arizona, Nevada, New Mexico, and California). The 2002 Fire Conference was convened to tackle these concerns. It began with a plenary session addressing the central problems of fire and fuels management in the Southwest. Concurrent sessions with over 100 oral presentations covered a wide range of topics, including fire ecology, fire behavior, fire history, fire prevention, fire education, restoration and rehabilitation, air quality, wildlife-fire interactions, fire planning, watershed responses to fire, invasive species responses to fire, National Environmental Policy Act and other regulations, and vegetation-fire interactions. More than 50 posters displayed in an afternoon session rounded out the program. The 39 papers and 17 extended abstracts included in this volume serve as a reference for the management of fire and fuels concerns in the southwestern United States.

Keywords: fire behavior, fire ecology, vegetation treatment, watershed response, wildfire, wildlife response.

Acknowledgments

Many people and institutions contributed to the success of this conference. It is difficult to list them all, but the following persons and groups deserve special recognition. First, thanks go to the primary conference sponsors, the Association for Fire Ecology and the Western Section of The Wildlife Society. Additional financial support was provided by the California Department of Parks and Recreation; the Joint Fire Science Program; U.S. Department of Agriculture, Forest Service, Washington office; USDA Forest Service, Pacific Southwest Research Station, Riverside Fire Laboratory; United States Geological Survey; U.S. Fish and Wildlife Service; and the U.S. Department of Interior, National Park Service. The conference was ably organized by program committee chairs Matthew Brooks, Catherine Hibbard, and Kevin Shaffer. They were aided by committee members Suraj Ahuja, Jan Beyers, Sarah Converse, Cynthia Graves, Wayne Harrison, William Laudenslayer, Marcia Narog, Mary O'Dea, Barbara Rocco, Carrie Shaw, Scott Stephens, Neil Sugihara, and Robin Wills. Christie Neill served as a session chair, along with many program committee members, and Pat Lineback chaired a planning workshop. A special thank you goes to all the volunteers who helped with technical and logistical support and other conference duties; there were too many to list here. I also extend my gratitude to the many peer reviewers of the contributed paper manuscripts for this volume and to Jan Beyers, who did last-minute technical editing. Without everyone's help, the highly successful conference and this General Technical Report would not have been possible.

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Contents

Oral Sessions

Session A: Fire Prevention and Education

Session B: Fuel and Vegetation Treatment and Management

Session C: Watershed Response and Rehabilitation

Session J: Fire History and Restoration

Session K: Fire Behavior

Session L: Fire Ecology III

Poster Session–Extended Abstracts

Equipping Tomorrow's Fire Managers[1]

Christopher A. Dicus[2]

Abstract
Fire managers are challenged with an ever-increasing array of both responsibilities and critics. As in the past, fire managers must master the elements of fire behavior and ecology using the latest technologies. In addition, today's managers must be equipped with the skills necessary to understand and liaise with a burgeoning group of vocal stakeholders while also facing the complications of a changing landscape, particularly an increasing wildland-urban interface. These challenges have been embraced in the Fire and Fuels Management program of study at Cal Poly State University, San Luis Obispo. There, classes are offered in fire suppression, ecology, and management. Other required courses address the historical role of fire in society and its subsequent effects on current policies, the evolution of fire technologies, and the management of the wildland-urban interface. Throughout their tenure in the program, students are perpetually immersed in an atmosphere in which they must develop innovative and realistic solutions to real-world problems. This "learn by doing" philosophy is fostered by course assignments, a mandatory internship and senior project, and in various research opportunities. This paper discusses the successes and lessons learned at Cal Poly and examines the future of equipping tomorrow's fire managers.

Introduction

Wildland fire management is an increasingly high-profile profession. The dominance of wildfire in summer-month headlines is testament to the intense fascination our society places on the fire phenomenon. Because fire evokes such strong emotions of passion and fear in American culture, those charged with fighting these fires are celebrated as heroes who protect the masses and the environment from imminent doom. However, the public has simultaneously begun to question fire managers as to why the extent, damage, and cost of suppressing wildfires seemingly continue to grow on an annual basis. While this is a complex question with a host of answers working together in concert, the public and policymakers are clamoring for changes in land management that reduce the burgeoning negative effects of wildfire.

The fire management profession is becoming increasingly challenging. While managers are now commonly called on to reduce the risk of wildfire through pre-fire management activities, they are forced to work within the constraints of multiple barriers including threatened and endangered species, archaeological sites, and smoke concerns. Further, land managers are faced with an increasing level of public antagonism from factions covering a spectrum of political ideologies and agendas.

Thus, fire managers are challenged with an ever-increasing array of both responsibilities and critics. Effective fire managers cannot focus solely on suppression tactics. Instead, they must also successfully integrate the elements of fire behavior with fire ecology, using the latest technologies, to implement pre-fire

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[1]An earlier version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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treatments. In addition, they must be equipped with the skills necessary to understand and liaise with a burgeoning group of vocal stakeholders while also facing the complications of a changing landscape, particularly an increasing wildland-urban interface.

Unfortunately, a formal education in fire and fuels management is somewhat rare at present. Several exemplary graduate programs that emphasize wildland fire exist today, but many are concerned primarily with specific ecological problems and graduates are not necessarily educated in management techniques and solutions. Also, these programs can only provide a limited number of graduates in a field that desperately needs qualified fire managers. Even fewer educational opportunities to adequately equip undergraduate students in fire and fuels management exist.

The complex challenges of fire management have been embraced in the undergraduate Wildland Fire & Fuels Management concentration in the Forestry & Natural Resources (FNR) major at Cal Poly State University, San Luis Obispo. Throughout their tenure in the program, students are perpetually immersed in an atmosphere in which they must provide innovative and realistic solutions to real-world problems. This "learn by doing" philosophy is fostered in six courses dedicated to various aspects of fire management, mandatory internships and senior projects, and in various research opportunities. The manuscript describes the evolution of the Wildland Fire and Fuels Management concentration at Cal Poly, discusses the skills needed by today's fire managers and how they are addressed in the curriculum, and examines the future needs at equipping tomorrow's fire managers.

Brief History of Program

Dr. Tim Plumb arrived at Cal Poly in 1981 and was the earliest faculty member with an interest in wildland fire in the Natural Resources Management major. His position emphasized the management of chaparral and oak woodlands, but the pervasiveness of fire in these ecosystems naturally led him to explore fire extensively in his courses and research.

Partially as a result of increasing student interest in fire ecology, the Natural Resources Management Department created the Watershed, Chaparral, and Fire Management concentration of the Natural Resources Management major in 1986. Initially, this concentration emphasized chaparral management, but student interest began to diverge into separate fire and watershed factions. In 1993, the Natural Resources Management major was replaced by the Forestry and Natural Resources (FNR) major, which was accredited by the Society of American Foresters. The Watershed, Chaparral, and Fire Management concentration remained within the new major.

After Dr. Plumb's retirement in 1997, the faculty at Cal Poly recognized the continually increasing student interest in wildland fire and hired Dr. Scott Stephens based, in part, on his extensive background in fire ecology and management. Dr. Stephens remained at Cal Poly for two years until he returned to his alma mater, U.C. Berkeley, to assume the faculty position in fire management there.

In 2000, the existing Watershed, Chaparral, and Fire Management concentration was divided into the Wildland Fire and Fuels Management concentration and the Watershed Hydrology concentration, both of which are in place today. Student demand for the new fire concentration was immediate and immense, and continues to grow at a fierce rate. Dr. Chris Dicus, with a background in fire ecology and silviculture, arrived at Cal Poly in 2001 to head the new concentration and presently remains in that capacity.

Curriculum

The curriculum of the Wildland Fire & Fuels Management concentration goes far to equip students with the skills needed for effective fire management. Due to today's need for successful pre-fire management, it is critical that wildland fire managers be strongly grounded in forestry and vegetation management. Thus, the success of graduates in the program is based partly on the concentration being imbedded within the Forestry and Natural Resources major, which emphasizes an ecosystem-level approach to resource management. In addition to a mandatory internship and senior project, all fire and fuels management students must successfully complete courses in fire control, fire ecology, fire management, and wildland-urban interface fire management (*table 1*). Restrictive electives in the concentration can be fulfilled by such courses as Fire and Society, Technology of Wildland Fire Management, an Emergency Medical Technician course, or other advisor-approved courses.

Table 1—*Required courses for Wildland Fire and Fuels Management concentration within the Forestry and Natural Resources major at Cal Poly State University, San Luis Obispo.*

Course number	Course title	Units[1]
FNR 204	Wildland Fire Control	3
FNR 307	Fire Ecology	3
FNR 339	Internship	6
FNR 340	Wildland Fire Management	3
FNR 412	Senior Project	4
FNR 455	Wildland-Urban Interface Fire Protection	3
Restrictive electives[2]		10

[1]Total of 192 units to graduate in FNR major, 92 of which are Major courses
[2]May include Fire & Society, Technology of Wildland Fire Management, college-level EMT course, or other advisor-approved courses

At a minimum, fire managers must be equipped with an understanding of suppression concepts. Thus, students begin their concentration courses in Wildland Fire Control (FNR 204), which initiates them into basic skills and tactics needed in fire suppression. Students learn basic fire concepts such as combustion physics and fire behavior as it is affected by fuel, topography, and weather, and then proceed to suppression tactics, safety, incident command, and mechanized equipment. Laboratory exercises demonstrate concepts learned in the classroom in a field setting and allow students to "get their hands dirty." Because the course successfully fulfills requirements of the Basic-32 federal Firefighter Training course (I-100: Introduction to the Incident Command System, S-110: Wildland Fire Suppression Orientation, S-130: Firefighter Basic Training, and S-190: Introduction to Wildland Fire Behavior), it is immensely popular with majors throughout the University and has been used as a useful recruiting tool for the major and concentration. In an effort to best serve students, additional requirements to the class that would fulfill the California

Department of Forestry and Fire Protection’s (CDF) 80-hour Firefighter certification are currently being explored.

Fire managers must be equipped with an understanding of how fire, or its absence, will affect various ecosystems. Because of the pervasiveness of fire in the western United States and the course’s applicability to all concentrations in forestry, all students in the FNR major are required to complete Fire Ecology (FNR 307). This course investigates how fire affects vegetation, soil, water, air, fauna, and other ecosystem components as well as how it interacts with other disturbance types. Field laboratories reinforce concepts learned in the classroom and explore such topics as fuel loading and potential effects on fire behavior, dendrochronology, soil hydrophobicity, effects on wildlife, and others.

Fire managers should be equipped with advanced knowledge of fire behavior and be able to utilize the latest technologies available. Students learn these and other skills in another required concentration course, Wildland Fire Management (FNR 340). There, students explore factors that affect fire behavior and model fire using the BehavePlus and FARSITE computer simulation models. Students also learn the intricacies of prescribed fire to obtain specific resource objectives while integrating techniques to manage smoke and other areas of concern. Further, students learn some of the socio-political philosophies and constraints that they will likely face in their career and various approaches to address them.

In today’s world, fire managers must be equipped with skills to manage fire where human development has encroached upon wildlands. The capstone class in the fire and fuels concentration is therefore Wildland–Urban Interface Fire Protection (FNR 455). This course examines the social, economic, political, and technological issues affecting fire management in urbanized landscapes where fire continues its ecological role. It further explores fire risk analysis, needs assessment, legislative codes, standards and policies, liability issues, evacuation, and incident response planning. Field laboratories demonstrate the reoccurring and problematic characteristics of interface communities, where fire-prone structures are commonly built around heavy fuels and with inadequate infrastructure. As a final project in the course, students create a fire management plan for a local community that includes GIS mapping of fuels, infrastructure, weather, potential fire behavior, and ways to ameliorate existing conditions. The sophistication of these reports has recently led CDF to seek Cal Poly assistance in creating fire plans for communities throughout San Luis Obispo County.

Fire managers should also be equipped with an understanding of how fire has impacted man socially, as this directly correlates with values and subsequent policies. Thus, students may elect to take Fire & Society (FNR 308), which explores how fire has influenced human development and the use of anthropogenic fire by various Native American cultures. This course further explores the history of fire use and of fire policy in the United States. Laboratories provide invited speakers who discuss social-driven topics such as the mind of the arsonist, the psyche of wildland firefighters, and symbolism of fire in various religions. Because this course can be used for General Education credit in the University and is an extremely high-demand class, it has been used successfully to recruit students to the FNR major.

Fire managers must be equipped with an understanding of state-of-the-art technology that is employed in all phases of management. Thus, students may elect to take Technology of Wildland Fire Management (FNR 312), which examines the use of models and technology to solve complex land management problems. It further explores the assumptions and limitations of fire behavior and suppression models. This is another General Education course that generates a great deal of interest and is also used to recruit students into the FNR major.

Other requirements in the program further equip students with skills needed for effective fire management. For example, students may elect to take an accredited, college-level EMT course. While not directly related to fire management, it facilitates gainful employment for students upon completion of their degree. Students in the concentration are further required to complete an Internship in Forest and Natural Resources (FNR 339) that in some way pertains to fire management; employment on a hand crew is permitted, but students are also encouraged to seek positions that will afford them experience in some of the more sophisticated aspects of fire management. Students are also required to complete a Forest and Natural Resources Senior Assessment Project (FNR 412), which explores some topic related to fire ecology or management. Recent projects have investigated such topics as fire history, fire planning, fire effects, and training exercises for interface fire events. Further, students are also encouraged to participate in the research activities of faculty members, especially if they are considering pursuing a graduate degree.

Future Vision

One area of concern for students is placement after graduation. At present, many entry-level positions do not require the skills gained at Cal Poly, but can be filled by individuals without a college education. Federal agencies are currently facing a multitude of mid-career employees in fire management who do not have the necessary requirements for advancement into higher-level professional job series. In the short term, Cal Poly is modifying existing courses into intensive short-courses, which provides University credit to advance mid-career fire managers into GS-401 professional series. However, this problem should be addressed by agencies in the long-term by the creation of entry-level positions that require the skills that are gained in the Wildland Fire & Fuels Management concentration.

Due to a continuing demand from land management agencies, communication skills are paramount to a successful career in fire and fuels management. To that end, the curriculum will continue to develop opportunities to enhance both written and spoken communication proficiency. For example, class projects are regularly presented to federal, state, and municipal officials in order to enhance oral communication skills as well as receive constructive criticism and comments from working professionals.

Graduates of the concentration are equipped with the skills needed to be competitive in most facets of wildland fire and fuels management. However, because of the University's location on the central coast of California, where both development and wildland fires continue to increase, it is likely that the program will continue to emphasize management in the wildland-urban interface. Students graduating in the Fire & Fuels Management concentration are educated to become active and effective managers who have a mindset beyond suppression activities. While suppression is a vital facet of wildland fire management, Cal Poly graduates are trained to become the future planners and policymakers in the wildland fire arena.

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The Role of Nongovernmental Organizations in Fire Education, Fuels Reduction, and Forest Restoration: A Call for Collaboration[1]

Timothy Ingalsbee,[2] Daniel Henry,[3] Oshana Catranides,[4] and Todd Schulke[5]

Abstract
Successfully educating homeowners and communities about wildland fire ecology and management, reducing hazardous fuels, and restoring fire-adapted forest ecosystems will place enormous demands on the budgets, resources, and staff of federal agencies for several decades to come. This work can be aided by collaboration with non-governmental organizations (NGOs) that are capable of mobilizing grants, donations, local practitioner expertise and indigenous knowledge, as well as volunteer labor from the private sector and local communities. NGOs can also utilize federal funding and resources to work on private residential properties, where federal agencies face many cultural and institutional challenges. This paper features some innovative programs in community-based fire education and fuels reduction currently being conducted by NGOs and argues for the vital role of NGOs in collaborative fire management planning and management as a means of furthering the long-term goals of the National Fire Plan and Federal Wildland Fire Policy.

Introduction

Conservationist non-governmental organizations (NGOs) provide a natural constituency supporting long-term ecosystem protection and ecological restoration; however, federal agencies are wary of working with NGOs, who have long been among the agencies' most ardent critics, often appealing and litigating forest management projects that involve commodity timber extraction. Bush Administration officials and certain members of Congress have recently claimed that appeals and lawsuits by NGOs have unduly delayed or thwarted fuels reduction projects designed to prevent catastrophic wildfires. Countering charges of "environmentalist obstructionism," conservationists have responded that they only challenge projects that propose commercial logging of large-diameter trees, but generally do not oppose projects using non-commercial small-diameter tree cutting or prescribed burning. Fortunately, the National Fire Plan (NFP) provides the political incentive and budgetary means for erstwhile adversaries in conservation organizations and government agencies to become newfound allies collaborating on fire-related projects and programs. This paper provides some brief examples of innovative fire-related programs being conducted by conservationist NGOs in collaboration with federal

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agencies. These programs provide a framework for implementing good projects with broad conservationist and community support, and minimum controversy or conflict.

The Lands Council's Homeowner Wildfire Education Program

The Lands Council (TLC) is a non-profit conservation organization founded in 1983 and based in Spokane, Washington. They are a local grassroots partnership dedicated to protecting the quality of life and natural environment in the Inland Northwest by engaging in scientific research, document analysis, field monitoring, and public education projects. TLC regularly participates in Forest Service NEPA processes and occasionally appeals and litigates commercial logging projects; nevertheless, TLC saw opportunities for doing homeowner fire education by working in collaboration with the U.S. Forest Service (USFS). In 2001, TLC's Wildfire Education Project began with funding from a NFP grant that was administered by the Colville National Forest. Occasionally, circumstances caused TLC and the USFS to be adversaries, but to their mutual credit this did not inhibit each organization from becoming cooperators in addressing homeowner fire education and community wildfire protection. Common ground was forged between the two parties by taking a common sense approach to protecting private property from wildfire destruction: the sensibility that fire prevention begins at home.

Geographic and Social Context

TLC's Wildfire Education Program works with rural communities in Stevens and Pend Oreille counties in northeastern Washington. The landscape is dominated by dry forests composed mainly of ponderosa pine (*Pinus ponderosa*). Fire was once frequent in these forests, but that has been altered by past fire exclusion and suppression. The area is not composed of urbanized communities of small towns or subdivisions that fit the classic wildland/urban interface (WUI) concept; rather, it is more inhabited by dispersed residents living on large lots and disconnected small clusters of homes better described as the intermix zone. TLC has dubbed the area the "wildland/rural interface zone." A strong anti-government and anti-environmentalist sentiment prevails among area residents, who cherish their privacy and guard their properties against all trespassers.

Methodology of Wildfire Education Program

In order to educate residents about fire prevention issues, TLC originally attempted to meet with homeowner associations and host seminars in a town hall format, but attendance was very low at these meetings. At the same time, residents' fear and concern about wildfires was growing. Consequently, TLC devised a new strategy: going door-to-door for one-on-one sessions with individual homeowners. Because personal meetings are conducted with homeowners on their own properties, TLC fire educators have the ability to immediately assess the level of comprehension of the hazard awareness curriculum and directly clear up any questions or confusion in a non-threatening atmosphere. A beneficial outcome of the home visits, however, is that it raised homeowners' interest and stakes in utilizing the free fire risk assessments and subsequent fire prevention plans.

TLC believes that the key to their program's success is "a mixture of honest salesmanship and scientific backing." The honest salesmanship centers on an offer that homeowners find hard to refuse: free assessments of the homeowner's fire risks and fuel hazards. The scientific support is based on the FIREWISE program and the research of Jack Cohen at the USFS Fire Sciences Lab in Missoula, Montana. For each homeowner, the assessment includes an overview of building design and materials, vegetation, and other flammables within a 200 foot radius "home ignition zone" (Cohen, 1999) surrounding their house and associated structures.

Following the assessment, homeowners can opt for a free fire prevention and fuel reduction plan created between TLC and the homeowner that will help mitigate the risks and hazards identified by the assessment. In addition to the home protection information, some education in basic fire ecology processes is added, but there is no anti-logging message provided—nor is any required, for in general, nearly all homeowners are interested in keeping their big trees standing.

Another collaborator and component of the program's success is the Washington State Department of Natural Resources (WDNR). The WDNR will send small brush cutting crews to implement the homeowners' fire plans and develop defensible space around homes. The WDNR offers this service for free using grant money from the NFP. The program focuses on removing flammable brush and increasing the spacing of small-diameter trees. Only brush and small-diameter trees six inches DBH or less are removed, and these are often chipped and spread out on the property.

Summary

TLC and other conservationist NGOs believe that if private homes and rural communities are effectively protected from wildfire destruction, then both social and ecological benefits are realized. Reducing fuels on private lands and protecting homes might expand opportunities for reintroducing fire in forested areas adjacent to communities. The basic idea is that the sooner homes are protected from wildfire, the sooner forests can be restored with prescribed and wildland fire.

In the first year of the Wildfire Education Project, TLC completed over 2,000 free fire assessments and 80 fire plans, with 60 residents opting for the full package of fire assessments, fire plans, and fuels reduction. The ongoing collaboration of TLC, USFS, WDNR, and private homeowners is forging a fruitful partnership. Also, TLC is willing to work on private residential lands where federal agencies face many cultural and institutional challenges. The personal homeowner visits provide a useful model for doing collaborative fire prevention education and outreach in the rural intermix zone and are a role well-suited to conservationist and community service NGOs.

The Lomakatsi Restoration Project's Ecological Fuels Reduction Program

The Lomakatsi Restoration Project (LRP) is a non-profit conservation organization founded in 1995 and based in Ashland, Oregon. Lomakatsi is the Hopi word for "life in balance." The LRP provides environmental education and outreach, ecoforestry workforce training programs, and community-based watershed restoration projects. LRP is particularly focused on creating job opportunities in hazardous fuels reduction and ecological restoration for displaced forest workers and local residents. The founders of the LRP wanted to work on ecologically-oriented and community-based restoration projects, and discovered opportunities for doing ecological restoration via hazardous fuels reduction on private lands. When working on private lands, many bureaucratic regulations governing federal land management do not apply, and the LRP believes there is more room for developing ecological objectives and applying innovative techniques for fuels reduction. The LRP is mindful that fuels reduction for home protection is not the same thing as ecosystem restoration; nevertheless, the LRP attempts to develop and apply ecological principles and restoration objectives in its fuels reduction projects.

Geographic and Social Context

The LRP works in Josephine and Jackson Counties near the communities of Williams and Ashland in southern Oregon. The area is comprised of mixed conifer forests in an ecosystem that historically had a frequent fire return interval. Due to fire exclusion, flammable small trees, brush, and invasive weeds have increased, especially in previously logged and grazed areas. There are several wealthy landowners living in Ashland's interface and intermix zones. They cherish all of their remaining big trees, love the wildlife that inhabits their lands, and do not want to see them removed by fuels reduction efforts. But these homeowners also love their homes and are fearful of wildfire. The LRP's wealthy clients are willing to pay for the kind of relatively expensive manual fuels reduction treatments that can improve protection of their homes and resilience of their forests in case of wildfire, but do so in a sensitive, restoration-oriented manner.

Methodology of Ecological Fuels Reduction Projects

In the Ashland interface and intermix zones, dense thickets of flammable manzanita (*Arctostaphylos viscida* and *A. nevadensis*) and "chaparral" brush (*Ceanothus leucodermis*), and small pole-sized incense cedar (*Calocedrus decurrens*) and white fir (*Abies concolor*) trees, have grown in sites altered by past logging, grazing, and fire exclusion. In sites the LRP has been working, a typical stand contains an average of 80 big stumps that are smothered by up to 2,000 small-diameter stems per acre. When confronted by a contiguous field of manzanita or dense dog-hair fir thickets, part of the initial treatment is simply opening up stands enough for crews to walk through and assess the ecological needs of the site.

Workers are encouraged to first walk the ground and develop a relationship with the site before ever starting up their saws. This involves not only assessing the fuel loads, but identifying the native plants, learning the location of the large trees and riparian zones, and other notable natural features. The process is similar to sizing up a wildfire during initial attack, and workers may spend a whole day sizing-up a fuels reduction or restoration project. A key priority for LRP and their clients is to identify and protect all remaining large standing trees—especially snags—and protect water sources.

Thinning is done in a mosaic pattern, expanding all existing openings, and leaving some thickets behind. The LRP's thinning philosophy is to work in a circular, irregular, patchy fashion rather than a linear, grid-like system with uniform spacing between trees. To help protect remaining large, old trees, crews first thin around the large trees at least to the drip line of the outer canopy. Lower dead limbs and live ladder fuels are cut, piled and burned or swamper-burned. Crews do not eradicate all brush or small-diameter trees; rather, they try to retain some patch thickets in corridors for the sake of wildlife cover and species diversity. Additionally, some brush piles are not burned but are built around large relic stumps and preserved for use as wildlife habitat, primarily shelter for rodents. This helps build up the prey base for raptors and other carnivores, and mitigates the reduction in vegetative cover.

The objective of the LRP's ecological fuels reduction program is to perform multiple light entries, with monitored results guiding any subsequent activities.

Crews first lightly thin a stand; then step back and look at it; then come back in a couple hours or days to work more. Their intent is to return every few years to monitor and re-examine the site to determine whether or not it needs more thinning. The primary directive is to act conservatively: don't change a site too much or too quickly. The LRP believes that crews can always come back and cut more trees, if necessary, but if they cut too many trees too quickly, they cannot put those trees back living on the stump.

Workforce Training and Development

In addition to ecological concerns, the LRP has strong socioeconomic objectives with their fuels reduction work. The LRP is working to develop the kind of educated, skilled, and motivated workforce that is going to implement fuels treatments on the ground, as well as monitor and steward the land's restoration needs over the long term. The LRP's ecological fuels reduction projects are labor intensive and the hand-cutting and pile-burning is intensely hard work. The LRP considers their fuels reduction treatments to be a form of landscape artistry and desires to make ecological fuels reduction and restoration a high-wage, high-skill, high-status form of craft labor, as opposed to the low-wage, low-skill, and low-status industrial form of labor that characterizes traditional slash and brush disposal work.

The LRP conducts free public workshops, recruits interns, and tries to involve the community in its work for private clients. Field crews are organized like worker-owned cooperatives, and decisions are made collectively by consensus. The workers fully participate in each step of a project: pre-treatment biological surveys and environmental assessments, planning and design of prescriptions, implementation of treatments, and post-treatment monitoring. There is no division of labor between planners, implementers, and monitors of projects. Finally, the client landowners also go through training to understand the ecological and restoration objectives that are part of the fuels reduction work and to become an integrated part of the projects. After a project is completed, landowners are given a list of future recommended actions and are trained for some monitoring tasks. Assuming that most landowners will continue to reside on their properties for some period of time, they will become the primary monitors for the fuels reduction and restoration treatments.

Summary

The local community affectionately calls the LRP "the environmentalists with chainsaws." Their ecological fuels reduction program is being used in collaboration with other conservationists and ecoforestry professionals and put into practice. The LRP considers itself to be fortunate to have a relatively wealthy clientele that values their property and is willing to pay for this kind of work. Total costs per acre of the complete fuels reduction package depend on fuel loading and ranged from $800 to $1,600 per acre. However, work for private landowners does not fully cover all of the organization's educational and outreach activities; consequently, the LRP applies for grants from private foundations and has received a grant from the NFP. The LRP has organized three multi-regional fuels reduction field workshops in high risk, high hazard areas near the rural communities of southern Oregon. The Bureau of Land Management (BLM) administers the grant and oversees their work, and a very positive, mutually-beneficial relationship has developed between the LRP and BLM. The LRP hopes that the innovative and experimental techniques it is developing with private landowners will influence federal and state land management agencies in designing fuels reduction and restoration projects on public lands.

The Center for Biological Diversity's Mill Forest Restoration Demonstration Project

The Center for Biological Diversity (CBD) is a non-profit conservation organization based in the southwestern U.S. The CBD has a reputation for waging successful administrative and legal challenges against USFS logging and grazing projects. However, less well known is the fact that CBD also supports active forest management for ecological restoration and creation of local community-based wood products industries to utilize small-diameter trees that are by-products of forest restoration projects.

In the summer of 2000, CBD initiated the Mill Forest Restoration Demonstration Project in collaboration with Gila WoodNet and the Gila National Forest. Gila WoodNet is a nonprofit research and development corporation dedicated to developing new techniques and equipment for making value-added wood products from small-diameter trees removed specifically from forest restoration projects. The Gila National Forest has seen dramatic declines in commercial logging volume at the same time that unprecedented stand densities of previously sub-merchantable small-diameter trees are posing threats to forest ecosystem health. The aim of the Mill Forest Restoration Demonstration Project is to test treatment prescriptions and techniques for doing ecologically-based forest restoration that reduces wildfire hazards, produces jobs for local communities and resources for value-added wood products industries, but does not let pressures for commercial timber extraction become the economic or institutional driver for restoration projects.

Geographic and Social Context

The Mill Forest Restoration Demonstration Project is located in Grant County in southwestern New Mexico. According to the CBD, the area is home to over 30,000 people, of which more than half of the population is Mexican American, and many families live below the poverty level. Mining, ranching, logging, and farming have historically been the major industries in the area; however, approximately 700 local jobs have been lost in recent years when the mines were abandoned and two sawmills closed down. Unemployment levels currently exceed 12 percent in Grant County.

The Gila National Forest, like most of the interior West, is comprised mainly of dry forest types with historically frequent fire return interval. However, according to the USFS Forest Inventory Assessment, forests are uncharacteristically dense with approximately 90 percent of the trees being less than 12 inches DBH, making them susceptible to fire-caused mortality and posing a high risk of unnatural stand-replacing wildfires. In many cases in the Gila National Forest, tree density conditions are so far outside the historic range of variability that neither a passive or preservationist approach (e.g., "zero cut") nor a prescribed fire only method for restoration may be possible. Consequently, for both ecological and economic reasons, there is agreement among conservationists, local communities, and federal agencies on the need for some kind of forest restoration projects that will facilitate the creation of local wood products industries utilizing small-diameter trees.

Methodology for the Restoration Demonstration Project

Treatment prescriptions for the Mill Forest Restoration Demonstration Project were based on the "Natural Processes Restoration Principles" developed by the Bandelier Working Group, a collaborative effort among conservationists and scientists from a number of universities and agencies. According to these principles, ecological objectives rather than economic imperatives must guide the restoration prescriptions, and the primary objective must be to move existing forest structures towards conditions that will enable the restoration of natural processes. Existing native forest patterns are used as a guide for treatments, rather than trying to force human preconceptions of historic structures onto the contemporary landscape. Consequently, thinning efforts expand the existing openings between clusters of trees rather than create artificial uniform spacing between individual trees. No trees larger than 16 inches DBH are cut, with the majority of thinning targeted towards trees 9 inches DBH or less. The basis for this diameter limit came from the CBD's analysis that compared historic forest structures with current forest conditions and discovered that region-wide there is a deficit of trees greater than 16 inches DBH.

A conservative approach towards thinning and an integrative approach towards restoration is employed; thus, ecological restoration includes more than tree thinning, but also road closures and obliteration, grazing deferments, erosion control, invasive weeds eradication, and native seed planting. By ensuring that ecological objectives rather than economic interests guide project designs, the Natural Processes Restoration Principles addresses many of the concerns of the conservation community.

Rigorous multi-party monitoring protocols have also been established between the CBD, the Ecological Restoration Institute at Northern Arizona University, and the USFS Rocky Mountain Research Station. Plots have been established and data is collected before, during, and after treatments are implemented. Equally important, the socioeconomic effects of forest restoration efforts on local rural communities are also being monitored and evaluated. The Mill Project serves as the main supplier of small trees for the Gila WoodNet project, which sorts the trees into larger stems suitable for *vigas* (wood beams used in adobe house construction) and furniture, as well as those suitable for conversion into woodstove pellets, wood fiber composite building materials, and other value-added wood products. Gila WoodNet intends to achieve 100 percent utilization of restoration by-products through conversion of removed material into stove pellets, wood composite building materials, and other conventional wood products.

Summary

The CBD, Gila WoodNet, Gila National Forest, and area universities have joined together to craft a creative approach towards implementing forest restoration treatments and creating restoration forestry jobs. On test plots in the Mill Forest Restoration Demonstration Project, coalition members are formulating agreements on the kind of treatments that area forests need in order to restore natural processes and functions, while at the same time creating sustainable jobs and environmentally-sound commercial products from removal of small-diameter woody materials. Use of the Natural Processes Restoration Principles and multi-party monitoring provides the accountability needed to attract conservationist support for restoration products that yield some commercial utilization of small tree and biomass removal.

Conclusion

Assuming that fire education, fuels reduction, and forest restoration programs and projects will not be able to pay for themselves fully with commodity resource outputs, and thus assuming that they will rely to some degree on Congressional appropriations, it is in the interest of federal land management agencies to cultivate collaborative relationships with conservationist NGOs. Unlike federal employees, NGOs can lobby Congress for more appropriated tax dollars for projects, and they can also leverage those appropriated funds with donations, volunteer labor, and resources from private sources. Additionally, NGOs can bring innovative community-based approaches for doing this vital work. Collaboration among federal, state, and local agencies, and private stakeholders is not only a good idea, it is a principle articulated in the Appropriations Acts which fund the NFP. Hopefully, the example of The Lands Council, the Lomakatsi Restoration Project, the Center for Biological Diversity, and their federal partners will inspire others to emulate their model and engage in innovative collaborative programs and projects for the sake of educating fire-wise communities and restoring fire-adapted ecosystems.

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The Homeowner View of Thinning Methods for Fire Hazard Reduction: More Positive Than Many Think[1]

Sarah McCaffrey[2]

Abstract

With the focus of the National Fire Plan on decreasing fire risk in the wildland-urban interface, fire managers are increasingly tasked with reducing the fuel load in areas where mixed public and private ownership and a growing number of homes can make most fuel reduction methods problematic at best. In many of these intermix areas, use of prescribed burning will be difficult, and it is likely that thinning will be the dominant method for fuel load reduction. Yet little research has been done on acceptability of different thinning methods, and the current understanding is based primarily on accepted conventional wisdom. A limited number of surveys found that two-thirds of respondents thought thinning in general an acceptable fire management tool, but they did not examine differences in acceptability of specific practices. However, understanding what homeowners think about particular methods, and what is associated with more supportive views, can provide critical assistance to managers as they develop fuel hazard reduction plans. A survey of homeowners in Incline Village, Nevada found that support for most thinning methods, except herbicide use, was quite high, but varied across respondents. Factors associated with acceptability of specific methods include perception of fire risk, previous direct and indirect wildfire experience, perception of the role of various agencies in fire planning, and age. Individual responses also appeared to be influenced by the local character of the environment around Incline Village, particularly the desire to protect the water clarity of Lake Tahoe.

Introduction

Decades of successful fire suppression and the movement of more people into wildland areas have created a significant fire hazard throughout the United States and increased the complexity of trying to reduce the hazard. The growing number of houses within areas that retain much of their natural state creates a marked problem for public agencies, as it complicates both firefighting efforts and attempts to reduce the hazard through pre-fire fuels management. More people in the woods create not just more houses to protect, but also more views on resource management that must be taken into consideration. As a result, homeowner support of different fuels management practices will be integral to successful fire mitigation efforts.

In wildland areas with a large number of houses, use of prescribed burning will be particularly difficult, and it is likely that thinning will be the dominant method for reducing the fuel load. Many recent newspaper headlines suggest that in many ways thinning is no less controversial a practice than prescribed burning. However, although a limited number of surveys have found that over two-thirds of respondents thought thinning in general an acceptable fire management tool (Shindler and others 1996; Shelby and Speaker 1990), they did not examine differences in acceptability of

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specific thinning practices. In part to fill this critical gap and provide a more nuanced understanding of public views on fuels management practices, a mail survey was administered to homeowners in Incline Village, Nevada (Incline 2001). The town was considered by many within the fire management community to have one of the more proactive and effective fire management programs at the time (National Commission 1994). This program worked closely with town residents and relevant government agencies to encourage vegetation management and use of defensible space protocols.

Study Site

Incline Village is located on the northeast shore of Lake Tahoe, an area renowned for its great beauty, clear water, and abundant recreational opportunities. Approximately 77 percent of the Tahoe watershed is USFS managed land, 12 percent is private, and the remaining 11 percent is California or Nevada state parks (Elliot-Fisk and others 1996). With such valuable natural amenities, the area has been subject to growing population pressures and urbanization since WWII. Most environmental concerns in the Basin revolve in some way around arresting the decrease in the remarkable water clarity of Lake Tahoe that has accompanied development.

Incline Village itself is a resort community surrounded by numerous possibilities for skiing, hiking, and boating. Reflecting its resort status, roughly half the residences are vacant for part of the year. Reflecting the recent mobile demography of the West, only 30 percent had lived in the same residence in 1985. Estimates for 1998 show that the resident population was 87 percent white, had a median age of 41, and a median household income of $62,347 (Incline 2001). Although the area is un-incorporated, and most local government is conducted in Reno, fire responsibilities, including fire education, are administered locally by the North Lake Tahoe Fire Protection District. Essentially a series of homes, condominiums, and shopping strips interspersed throughout the forest, Incline Village is a clear example of residential wildland intermix (RWI). With a high proportion of vacation homes and recent permanent residents in a recreation area surrounded by public lands, it fits well into the demographic portion of the current wildland fire hazard equation. As a wealthy and predominantly white community, Incline Village may not have the broad distribution of income levels and racial groups of many metropolitan areas; however, it is also not an entirely unique representation of a residential-wildland intermix community. Throughout the U.S., the West in particular, scenic rural areas are attracting significant numbers of affluent migrants (Riebsame 1997).

Incline Village is also a good example of the ecological portion of the RWI equation as it sits in the middle of a very significant wildfire threat created by past management practices. Clear-cutting in the late 1800’s for local silver mines, and subsequent fire suppression, have created an even-aged, overly dense, white fir forest where drought and bark beetle attacks have left a significant portion of the Basin’s trees dead or dying (Huntsinger and others 1998). Such a remarkably uniform and dense fuel load composed of weak and fire-susceptible trees creates conditions most favorable to a wildfire but unfavorable for those living in the Basin. Unwilling to wait for a catastrophic fire to create public support for fuels management, Incline’s fire marshal in the late 1980’s, Gerald Adams, began to take steps to decrease the town's exposure through active education and mitigation work that included thinning and use of prescribed burns.

Methods

A mail survey was posted in June 1998 to a random sample of 643 individual property owners following Salant and Dillman's (1994) three wave approach. After bad addresses and undelivered surveys were subtracted, the study had a 46 percent response rate with a total of 279 usable questionnaires. Specific questions were then selected representing various factors—such as sense of responsibility, risk perception, knowledge levels—thought to influence attitudes and behavior in relation to fire management. Significant associations were then examined between these factors using Pearson's chi-square. Because the focus of the study was to identify *potential* factors that could help understand support for wildfire mitigation activities, a relatively low significance level (90 percent) was chosen: $p \leq 0.10$.

Although non-respondents were not contacted, survey respondents closely reflected two key variables that define the population of Incline Village, thereby reducing the risk of significant non-respondent bias. Fifty four percent indicated that they used their Incline property for less than 8 months of the year. This is comparable to the 53 percent of housing units found vacant in the 1990 census and to 1998 calculations that Incline Village had a permanent population of 9,354 residents and a summer population of around 18,000 (Incline 2001). In addition, 64 percent of respondents owned single family residences while 34 percent owned condominiums. This parallels 1990 census data where 34 percent of condominiums and 62 percent of non-condominiums were owner occupied.

In other respects, respondents were wealthier, better educated, and older than recent census demographics: over 57 percent of respondents had an income over $100,000 (versus 23 percent of 1998 census estimates), 58 percent were over 55 (versus 34 percent of 1998 census estimates), and 47 percent had some post-graduate education (versus 8 percent from the 1990 census). Much of this is likely accounted for by the fact that U.S. Census data includes both homeowners and renters while this survey was sent only to homeowners who generally would be expected to be older and have higher income and education levels than the general population. For instance, the 1990 census data indicates that 67 percent of owners were over 45 whereas only 20.5 percent of renters were over 45. The proportion of male respondents was also higher (66 percent) than that of the 1998 census estimates (51 percent). As the survey requested that the person most responsible for landscape maintenance fill it out, and it is likely that men are often responsible for this, this difference was not unexpected. It is generally observed by those living in Tahoe that much of the population is wealthy and retired, so these results fit that expectation.

Results and Discussion

A portion of the survey focused on acceptability and knowledge related to six thinning methods (*table 1*). No definitions or photos were provided of any of the methods, so responses reflect individual respondent perceptions of what the terminology means. Overall, there was a high level of awareness of thinning as a fire mitigation technique (86 percent), which was also seen as a generally acceptable fuels management activity. Over 75 percent of respondents found four of the five listed thinning methods at least somewhat acceptable. Hand thinning was most acceptable, with 80 percent of respondents finding its use fully acceptable. Contrary to beliefs that timber harvest is a controversial option, salvage logging and selective timber harvest were both fully acceptable to roughly three-quarters of respondents. Thinning with heavy equipment and use of grazing animals tended to be somewhat less acceptable. The one clearly unacceptable method was use of herbicides, with 50 percent finding them unacceptable.

Table 1—*Respondent views on acceptability of different thinning practices.*

How acceptable is each thinning method? (n=237)	Acceptable pct	Somewhat acceptable pct	Not acceptable pct	Not sure pct
Hand thinning by work crews	80	15	3	3
Salvage logging	75	18	1	6
Selective timber harvest	73	22	3	2
Undergrowth thinning with heavy equipment	52	26	15	7
Grazing animals	48	29	14	9
Herbicides	13	27	50	10

Respondents were asked to give reasons for any practice they found unacceptable. The few responses on timber harvest and salvage logging indicate that the main problem is distrust of commercial logging interests and the likelihood that they would take only mature trees that minimally contribute to the fire hazard, or leave the landscape damaged for decades. While heavy equipment was an issue due to air and noise pollution, the most frequently expressed concern was its potential to damage the soil and increase erosion. Grazing animals were seen by respondents as not practical in Incline given its steep slopes and resort nature: "Cows and goats roaming through Incline? Get real." By far the most comments were regarding herbicides which were seen as being completely unacceptable because they caused, as one respondent put it, "too much collateral damage." Specific reasons for unacceptability of herbicide use included potential negative effects on habitat and health of wildlife and humans, uncertainty of long-term effects, and potential toxic contamination of air, soil, and water.

One element unique to the area that clearly influenced non-acceptability of certain practices was concern about the water quality of Lake Tahoe. This was cited particularly in terms of herbicides getting into the lake and the potential erosion from use of heavy equipment and grazing animals. The surprisingly large proportion of residents who found timber harvest and salvage logging fully acceptable may be related to current logging practices in the Tahoe Basin. Any logging that takes place occurs under very strict conditions in order to minimize potential erosion. Tree removal in sensitive areas is only carried out by helicopter or when there is a snow pack to act as buffer between harvest equipment and soil. Such closely regulated logging may make residents less suspicious of potential environmental damage from timber harvest. The mixed response on grazing animals is possibly reflective of less familiarity with the process as grazing animals are not used much in the Tahoe Basin or, as indicated earlier, by the sense that it may be appropriate but in places other than Incline Village.

In terms of predictive factors for approval of different thinning methods, risk perception and experience were associated with support for mechanical thinning methods. Respondents who found the fire hazard in Incline more severe were more likely to find salvage logging (80 percent vs. 65 percent), selective timber harvest (83 percent vs. 72 percent), and hand thinning (86 percent vs. 76 percent) acceptable than those who saw the hazard as less severe. Experience with wildfire also appears to be associated with support for use of timber harvesting and salvage logging. Direct experience made a respondent 15 percent more likely to find timber harvest acceptable, and indirect experience had a similar effect on increasing the acceptability of both timber harvest and salvage logging.

The clearest set of associations with acceptability of thinning practices was on the appropriate level of involvement of the individual, state, or federal government in local fire planning. Those who favored a major state or federal role were more willing to accept management methods that, by their nature, are most easily managed at the state or federal level. Those who favored either a major federal or state role were 12 to 16 percent more likely to find hand thinning, salvage logging, and selective timber harvest acceptable than those who favored a more limited federal or state role. Notably, favoring a major federal role had an even stronger effect on acceptability of using heavy equipment (23 percent more likely) but favoring a major state role had no significant relationship with acceptability of heavy equipment use.

Of demographic variables, age was a fairly consistent predictive factor and is likely due to generational differences—the immediate post WWII emphasis on large scale government resource management and Smokey Bear and the more individual oriented, less “government trusting” approach that became more common in the 1960's. This is reflected in the association found between age and support for a major federal role in fire management, with those over 65 fourteen percent more likely to support a major federal role in local fire planning than respondents 45 to 65 and 18 percent more likely than those under 45 (*table 2*). Further reflecting this pattern is the positive association between age and approval of salvage logging, selective timber harvest, and heavy equipment thinning, with an increase in respondent age associated with a higher degree of acceptability for each method. All are activities requiring some level of large-scale (both financial and administrative) resource management.

Table 2—*The influence of respondent age on views about responsibility and acceptability of thinning practices.*

Age yr	**The federal government should have a major role in local fire planning** pct (n=262)	**Practice is an acceptable thinning method**[1]		
		Selective timber harvest pct (n=222)	**Salvage logging** pct (n=209)	**Heavy equipment** pct (n=210)
<45	65[b]	59[b]	71[a]	32[a]
45-64	69	77	76	58
<65	83	80	93	66

[1] Somewhat acceptable and not acceptable were combined for this analysis; not sure was excluded
[a] p<0.05; b: p<0.10

Associations with views on herbicide use showed a very different, although often complementary, pattern (*table 3*). No relationship was found between risk perception or experience with fire and approval of herbicide use. Age showed a similar relationship to that of other thinning practices with older individuals more likely to find it acceptable, although with herbicides the balance is tilted towards the practice being unacceptable rather than acceptable. Retirees also were more likely to find the practice acceptable. Herbicide acceptability also had a significant association with gender, with women 15 percent more likely to find their use unacceptable. Although favoring a major state or federal role in planning for local fire management showed no association with acceptability of herbicide use, opinion on the appropriate individual role did, with those who favored a major individual role much more likely to find herbicides not acceptable than those who favored a lesser role.

Table 3—*Influence of respondent demographic traits and sense of responsibility on unacceptability of herbicide use.*

Age yr	**Herbicides are an unacceptable thinning method**[1] pct (n=202)
<45	77[a]
45-54	68
55-64	44
<65	49
Retired	49[b]
Not retired	63
Female	67[b]
Male	52
Role respondent felt the individual should have in local fire planning	
Major role	64[a]
Supporting or no role	47

[1] Acceptable and somewhat acceptable were combined for this analysis; not sure was excluded
[a] p<.05; b: p<.10

Conclusions

Results provide evidence that members of the public are capable of being supportive of most thinning practices. Although caution should be used in generalizing the results given Incline Village's relatively high wealth and education levels, it should also be noted that a growing number of communities threatened by fire bear similar attributes and also have initiated education programs—so results may be more broadly applicable than at first glance. Notable is the importance of local environmental priorities in shaping acceptability, with concerns over a practice's potential negative impact on Lake Tahoe's water clarity being the most consistently stated reason for finding a practice unacceptable. While it is hardly surprising that herbicide use is unacceptable, it is surprising that salvage logging and selective timber harvest were acceptable to the vast majority of respondents. This may again be related to water quality concerns, with both these practices heavily regulated in the Basin to minimize erosion. Also notable is the relationship between age, sense of responsibility, and acceptability of certain practices where there appears to be a division in views based on age, with older respondents more supportive of government involvement and larger scale thinning practices while younger residents favor greater individual involvement in fire planning and hold a stronger dislike of herbicides. These results suggest that no thinning practice, except perhaps herbicide use, is an automatically unacceptable method. Rather they suggest that acceptability can be tempered by local dynamics that may make some practices that are usually unacceptable less (or perhaps more) problematic in certain regions. It is thus important for fire managers to pay attention to local environmental issues, as well as the general age of the local population, in deciding the best thinning practices to use to reduce the fire hazard.

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Prescribed Fire Education at Oklahoma State University: Training Our Future Pyros[1]

John R. Weir[2]

Abstract

The Rangeland Ecology and Management program at Oklahoma State University recognized the need for a practical, hands-on course designed for undergraduate and graduate students, with instruction on conducting prescribed fires for vegetation management, wildlife management, and livestock management. Two separate prescribed fire courses were initiated in the spring of 2000. The first course instructs the students on the how, why, when and where of conducting prescribed fires and is taught in a formal classroom setting. Students also are required to participate in a minimum of five of the numerous fires conducted throughout the semester. This provides the hands-on training and experience to complement classroom instruction. The second course is an advanced prescribed fire course that provides the student experience as fire boss. Other course requirements include writing fire plans, writing fire management plans, serving as crew leader, and assisting with training of students in the first prescribed fire course. With these courses we have attracted students from varied disciplines across the university as well as graduate students from across the US. Students from our program possess prescribed fire skills better preparing them for the careers and challenges in fire ecology and in land management today.

Introduction

Why do we need prescribed fire education or prescribed fire classes in the university? In my opinion there are two good answers to this question. First, with man's innate fear of fire, there has to be some way to calm that fear. Man has a natural fear of the unknown, and fire has become unknown or unlearned in our society today. Therefore, education and instruction of the future landowners and land managers to become comfortable with fire and use it responsibly will in turn help the general public overcome its fear of fire. The main problem with this is that it will take time, and how much time do our ecosystems have?

The second reason is invasive plant species. In Oklahoma, we are losing 782 ac of our prairies and shrublands per day to eastern redcedar infestation (Mosley, 2002). Not only is this affecting our native plant and wildlife communities, it is also affecting our human communities in the form of increased health problems (i.e. allergies) and volatile fuel build-up that is a risk to life and property (Engle and others, 1999). The probability of a catastrophic wildfire increases daily in Oklahoma's wildland/urban interface without some type of fuel management. The most cost effective method for redcedar control is prescribed fire (Strizke and Bidwell, 1990). We have the research on cedar and fire, and we have the knowledge to use fire wisely. So how do we increase the amount of fire being used in Oklahoma and the Southwest in general? Education and experience lead to confidence, and confidence in the people conducting the burns leads to public support and acceptance.

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Education and experience is the only way we will increase the amount of fire on the ground and smoke in the air. We have to train our future landowners, land managers and policy makers that prescribed fire can be used safely and is the most effective and economically viable way to maintain our natural ecosystems.

Public land states typically have several agencies that try to use fire and spread the word about fire. They have budgets for training, education and public service programs. In the private-land states, universities, through research and extension programs, are the main source for prescribed fire education. Mark Twain once wrote "*All schools and all colleges have two great functions: to confer and to conceal valuable knowledge.*" In the past we may have been guilty of the latter part of what Mr. Twain said, but now we are trying not to conceal anything about prescribed fire, but rather, get the word out.

Oklahoma, Texas, and New Mexico, as well as the rest of the Southwest and Great Plains, are in need of wise fire management, especially on privately owned wildands. Most of this region is made up of private lands or non-federal ownership. Oklahoma's land area is over 90 percent private land ownership, with the rest being urban areas, Indian lands, state lands and US Forest Service lands. Private landowners and non-federal land managers generally are unwilling to accommodate the federal bureaucracy to attend Federal prescribed fire schools and vice versa. Federal agencies have little influence within a state that is predominately private lands other than to provide technical assistance, incentives, and cost share programs to the landowners. So how do you educate the future managers and put fire on the ground? Educate them before you hire them.

Fire Courses

We have had students helping us for years on our research burns. They have always asked for more formal training and more prescribed burns on which to participate. In response, we developed a prescribed fire course in 1999, and in the spring of 2000 we enrolled 21 students in the Prescribed Fire course and five students in the Advanced Prescribed Fire course. Since that time, we have instructed 65 students in Prescribed Fire and 16 in the Advanced Prescribed Fire. The students are mainly undergraduate seniors and graduate students in Rangeland Ecology and Management, Wildlife, Forestry, Animal Science, Environmental Sciences, and Political Science. The course fills up rapidly every spring with many students waiting a year to enroll.

Prescribed Fire (RLEM 4983)

The Prescribed Fire class is limited to 20 students a semester. Lecture can accommodate several hundred people, but when it comes to conducting a fire you can have too many students on site, so this is the reason the course is limited to 20. Normally half to three-quarters of the class show up at each burn. This is a manageable size, and each student can actually participate in a task and not stand around being bored or causing a safety hazard.

Safety, history of prescribed fire, reasons why we burn, and prescribed fire laws and policies are covered in lecture. In addition, prescribed fire weather, sources of forecasts, appropriate fire prescriptions for the vegetation regions of Oklahoma, burn techniques and proper ignition devices are covered. We review the basics of fire plans and what should be included in every plan and step-by-step methods for smoke management. Lectures on prescribed fire economics and the development of prescribed fire cooperatives are also included. Quizzes and tests are given over these subjects throughout the semester to account for 40 percent of the class grade.

Each student is required to attend a field session in Prescribed Fire Equipment and Spotfire Training, a five to six hour lab, in which each piece of equipment we use on prescribed fires is reviewed for safe operation and utility. We also instruct on how spotfires occur and how to fight them safely and effectively. The students are then given a short test over this subject matter. Following this, we set a series of fires so the students can operate all equipment and learn the effectiveness, and the suitability of each piece of equipment. Simulated spot fires are ignited for the students to extinguish and to demonstrate safe suppression and mop-up techniques. We finish by burning a small unit to bring all of these techniques together. This training counts 15 percent toward the course grade and is required before a student attends any burns.

The students are also required to go on five burns for 25 percent of the course grade. This requirement allows the students to really learn about prescribed fire. We normally conduct 15 to 20 burns per spring. A few burns count as two credits, depending upon size and complexity. The burns units range in size from seven acres to 900 acres. Students who return the next day for mop-up of a burn will also get credit for a burn. If we arrive at a burn and the decision is made not to burn due to weather conditions, the students still get credit for a burn. Learning when not to burn is an important part of the learning experience. We make sure each student uses all the equipment and performs all the tasks throughout the semester.

We have several experienced research technicians who assist with each burn and provide individual instruction to the students. Some students are satisfied with just participating in five burns, while several participate in several burns. Each year we have one or two students who attend every burn conducted. For each burn over the required five, a student receives one bonus point toward the final grade. At the end of the year “The Pyro of the Year” is awarded to the student or students who have participated on the most burns that year.

The final project the students must complete is a fire plan. The students are given a burn unit and an outline of requirements for a fire plan. The students are required to make-up a fire plan, complete with ignition plans, escaped fire contingency plans, smoke management plan and crew assignments as 20 percent of the course grade.

Advanced Prescribed Fire (RLEM 5993)

We offered Advanced Prescribed Fire for those students wanting to extend their fire experience and take a leadership role in conducting fires. This course does not concentrate as much on formal classroom lectures as it does on learning by experience. Requirements for this course are completion of Prescribed Fire RLEM 4983 and consent of the instructor. One of the first class requirements is to assist with teaching the Prescribed Fire Equipment and Spotfire Training to the semester’s Prescribed Fire RLEM 4983 students. This helps the student build confidence in their ability to lead and assign people to tasks, as well as, learn how to work with people who do not yet understand the concepts of prescribed fire. This requirement counts toward 15 percent of the course grade.

The student is also required to be a crew leader on at least one burn during the semester. The student is in charge of an ignition and suppression crew. The student assigns crew members to the assigned tasks, follows the ignition plan, oversees the safety of each crewmember, and handles any problems that may arise. This counts as 15 percent of the grade.

The main part of this course and 35 percent of the grade is the fire plan and serving as fire boss. Each student is assigned a burn unit the first week of class for which they write a detailed burn plan and serve as fire boss. The student is required to make sure the weather is acceptable for that day and for notifying adjoining landowners, fire departments and crewmembers and making sure the equipment is ready. The student then reviews the ignition plan and assignments with crew leaders and crewmembers. The burn is that student's from the time it is ignited until mop-up is complete. It is interesting to watch how nervous and quiet the students become when they are in charge. Most students tell me it is not as much fun being in charge as it is being just a crewmember. This is part of the experience gained in being fire boss, and this will enable the students to make better decisions in the future.

Two case studies are assigned during the semester. These case studies involve a nearby management area. We have used private ranches, recreation areas, and a golf course that encompassed considerable wildland area. We meet with the manager of the area and students ask questions about management goals and objectives. The student then writes a prescribed fire plan, public relations plan, fuel management plan, budget for equipment purchases, and a crew-training plan for each area. These case studies are 40 percent of the grade for the semester.

Results from Courses

Students from our program possess prescribed fire skills better preparing them for careers in fire ecology and in land management. The students not only have more formal classroom training than most Federal and State agencies require, they also have considerable actual prescribed fire experience when they finish. This experience would take several years to acquire on most jobs. With the condition most of our lands are in now, we cannot afford to wait any longer. Many different employers are hiring the students who have completed these courses and many more prescribed fire jobs appear on the horizon (*table 1*).

Table 1—*Employers of former students who have taken courses in prescribed fire at OSU and who are applying skills acquired in the courses.*

Private	Universities and State Agencies	Federal Agencies
The Nature Conservancy	Rangeland Ecology & Management, OSU	US Fish & Wildlife
Red Buffalo, LLC	Oklahoma Dept. of Wildlife Conservation	US Forest Service
Bluestem, LLC-Turner Ranches, Inc.	Texas Parks & Wildlife	USDA-Agricultural Research Service
Private Ranch and Wildlife Managers	Texas Division of Forestry	USDA-Natural Resource Conservation Service
	Missouri Department of Conservation	Bureau of Land Management
		Bureau of Indian Affairs
		US Army Corp of Engineers

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Small-Diameter Timber Alchemy: Can Utilization Pay the Way Toward Fire-Resistant Forests?[1]

Jeremy S. Fried, R. Jamie Barbour, Roger D. Fight, Glenn Christensen, and Guy Pinjuv[2]

Abstract

There is growing interest in using biomass removed from hazardous fuels reduction treatments in wood-fired electrical generation facilities. An application of FIA BioSum to southwest Oregon's Klamath ecoregion assessed the financial feasibility of fuel treatment and biomass generation under a range of product prices and fire hazard-motivated silvicultural prescriptions. This simulation framework consisted of linked models developed on a foundation of Forest Inventory and Analysis data. Small diameter woody biomass and merchantable volume and pre and post-treatment fire hazard were characterized from a systematic sample and combined with transportation infrastructure information to estimate potential biomass delivered under different objectives, constraints, and assumptions about costs and benefits. The FIA BioSum model allowed users to evaluate the financial feasibility of locating biomass plants in specific places or, alternatively, to identify the lowest cost processing plant locations within a landscape. Only a small fraction of the total forested landscape in the southwest Oregon study area could be treated via operations that generate positive net revenue, though there was potential to expand treated area via substantial subsidy of logging and/or hauling costs.

Introduction

Using woody biomass derived from hazardous fuel reduction treatments for financially viable products is not easy, yet there is increasing pressure on managers to find ways to do this. As a result, various interests in almost every part of the country come forward with proposals to study or implement all manner of processing facilities to handle small diameter timber or other woody biomass. Managers need reliable ways to sort through these proposals to determine which ones make sense.

Several researchers have reported efforts to assess the impact of silvicultural prescriptions designed to reduce the risk of catastrophic fires or the impacts on forests when fires occur. A few of them have addressed the impacts of treatments on the ground, in terms of post-treatment fire effects attributes like tree mortality, char depth, fuel consumption and fire intensity (Omi and Martinson 2002, Oucalt and Wade 1999, Pollet and Omi 2002). A few have compared fire effects across treatment areas affected by a single wildfire, such as seedling establishment (Chappell and Agee 1996) and runoff and sediment production (DeBano and others 1996). Others have relied on simulation to assess the impacts and efficacy of alternative prescriptions through simulations in Sierra and Rocky Mountain coniferous forests

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[1]An earlier version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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with the Forest Vegetation Simulator (Forest Management Service Center 2001). Hollenstein and others (2001) found that removal of some fraction of the large (>30 inch dbh) tree population was critical to the maintenance of a sustainable stand structure and to the efficacy of fuel treatments. These stand level analyses provide the foundation for integrated landscape scale analyses that have not been attempted before now.

There is also a small body of literature on landscape-scale biomass availability (Noon and Daly 1996, Downing and Graham 1996, Graham and others 1997, Graham and others 1996) that involves assessing potential biomass supplies in a location-specific fashion, and draws on FIA plot data as part of the biomass supply picture. However, in these studies, which were conducted primarily for Tennessee and included non-forest biomass sources such as mill-wastes and agricultural byproducts, forest-produced biomass was handled as an undifferentiated commodity (e.g., short rotation woody conifers grown for sale as biomass, not timber), and there was no specification of silvicultural prescriptions, evaluation of removal costs, or intent to modify or evaluate fuel or fire attributes.

With support from the National Fire Plan, and the Western Forest Leadership Coalition, and building on results from a previous Joint Fire Sciences project (Barbour and others 1999), we developed the Forest Inventory and Analysis biomass summarization modeling framework (FIA BioSum) to estimate biomass availability, financial returns, and fuel treatment efficacy associated with silvicultural prescriptions devised to reduce fire hazard to forest stands (i.e., reduce the likelihood of stand replacement fire). FIA BioSum uses Forest Inventory and Analysis (FIA) plot data to: 1) Identify and evaluate the economic feasibility of potential sites for woody biomass processing facilities, 2) Provide economic analysis of alternative treatments, and 3) Predict the effectiveness of alternative treatments in improving fire hazard-related indices and attaining specified post-treatment stand conditions.

Methods

The FIA BioSum modeling framework (*fig. 1*) consisted of a linked series of generally available, documented models, including the Forest Vegetation Simulator, FVS, and its fire and fuels extension, FFE (Beukema and others 2000), and STHARVEST (Fight and others 2003), a spreadsheet model composed of regressions and look-up tables for logging cost components derived from empirical data on timber sales. It also included a series of GIS data inputs (i.e., FIA plot locations and comprehensive road networks), GIS processing steps, databases, and linear programming optimization protocols.

This framework was tested in the Oregon portion of the Klamath ecoregion, 4.6 million acres of mostly forested land that contains diverse coniferous and evergreen hardwood forest types and a heterogeneous distribution of landownership, including National Forest, BLM, industrial, and non-industrial private lands classes. Most of this study area has been characterized as fire regime condition class 2 or 3, indicating that fire regimes have been moderately or significantly altered from those of pre-settlement forests, and that risk of stand replacement fire is substantial (Schmidt and others 2002). Since this analysis was completed, the Biscuit Fire (2002) burned a significant area in the study; the official fire perimeter contains over 500,000 acres.

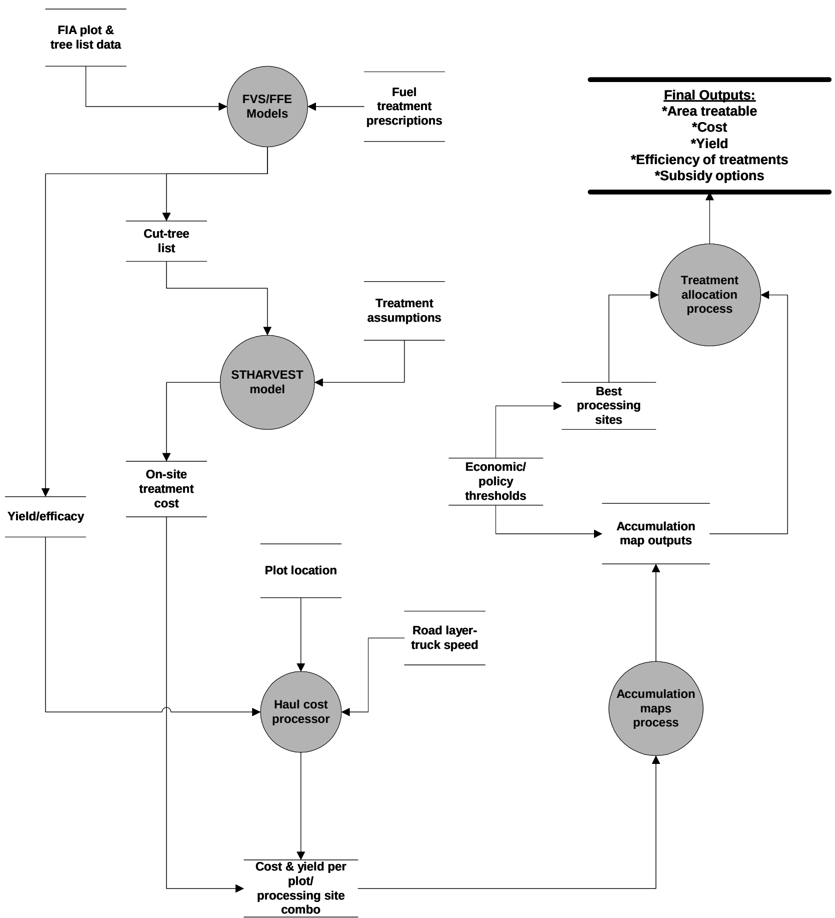

Figure 1—Flow diagram of the FIA BioSum modeling framework.

We characterized pre- and post-treatment fire hazard, biomass removed (by size class), and residual stand conditions for over 800 FIA and CVS (Continuous vegetation survey plots installed on National Forest and Bureau of Land Management lands) forest inventory plots located outside of designated wilderness and roadless areas. Two fuel-treatment motivated silvicultural prescriptions were simulated in FVS using tree lists from these plots: Prescription A, a stocking reduction in which stands were thinned proportionately across diameter classes to a residual basal area of 125 ft^2 ac^{-1}, and Prescription B, a thin-from-below which left a residual basal area of 80 ft^2 ac^{-1}. No trees larger than 21 inches dbh were removed in either treatment to be consistent with current policies on National Forests in portions of the study area. Biomass-sized trees (dbh<7 inch) and most hardwoods were valued at 26 dollars per green ton (\$ gr. ton^{-1}), and merchantable-sized trees (7 inch<dbh<21 inch) were valued at \$62 gr. ton^{-1}, a rate roughly equivalent to \$300 MBF^{-1}. Biomass and merchantable removals from each plot were summarized in a database along with residual stocking, and pre- and post-treatment fire hazard, which was represented in the estimates of torching and crowning indices generated by FFE. Torching index is the wind speed in miles hr^{-1} at which a surface fire would climb into the crowns of individual trees, and crowning index is the wind speed at which a crown fire would spread from crown to crown. *Larger* values for both indices are indications of *lower* fire hazard. Plot expansion factors were used to extend plot outcomes to acres in the landscape.

Fuel treatment costs for each plot were estimated using a combination of the harvest cost simulator STHARVEST (Hartsough and others 2001), other published information, and judgments of local experts. Cost components included felling, yarding, preliminary processing (e.g., limbing, bucking, and chipping), brush-cutting, and rehabilitation/remediation (water-barring of roads). Whole tree harvesting, cut-to-length harvesting and combinations of these were assigned to each plot based on the diameter distribution of the removals and plot slope. Cable yarding was assumed on plots with slope >40 percent Trees <3 inch dbh were always cut and left on the ground; trees 3 to 5 inch dbh were cut and removed as biomass on tractor yarded plots and left on cable yarded plots; trees 5 to 7 inch were always removed as biomass; trees 7 to 21 inch were removed as merchantable volume if selected by the prescription, and their tops and limbs utilized as biomass only if whole-tree harvested (which did not occur on steep slopes); trees >21 inch were never removed.

To evaluate delivered raw material costs and identify promising locations for siting a biomass-to-energy generating plant (e.g., where biomass accumulation potential for a given delivered unit cost is greatest), a systematic 10 km grid of potential processing sites was established. Potential processing sites within designated wilderness and roadless areas were omitted from consideration. Unit round-trip haul costs for merchantable and biomass-sized material were estimated by 1) tessellation of comprehensive GIS road layers to produce a transportation cost surface of 500 m grid cells and assigning the haul cost associated with the highest standard road within a cell, 2) processing this haul cost surface with a cost-distance GIS function to produce an accumulated-to-the-potential-processing-site haul cost map for each potential processing site, and 3) spatially joining these accumulated haul cost maps to the inventory plots. The end result was a table of costs inclusive of harvesting, skidding, loading, and hauling costs for delivering a ton of biomass from every plot location to every potential processing site.

When the removals, costs and fire hazard tables were combined, it was possible to evaluate the desirability of every potential processing site from multiple perspectives (e.g., biomass accumulation, net revenue, area of forest treated) and to develop maps depicting how the levels of these attributes varied over the landscape. Tradeoffs among costs, merchantable- and biomass-sized yield, area treated, and treatment effectiveness were evaluated for the most promising potential processing sites via linear optimization in which the model is allowed to choose among prescriptions (including the no treatment option) for each forested acre.

In this pilot study, we used the FIA BioSum modeling framework to address five questions: 1) Can we reduce fire risk? 2) How much of the landscape could be feasibly treated? 3) Will there be enough biomass to fuel a power plant? 4) Where are the best places to site a power plant? and 5) Would a subsidy help?

Results

Can we reduce fire risk?

Both prescriptions improved torching index and crowning index on most acres, but prescription B (thin from below to 80 ft^2 ac^{-1}), shown in *figure 2*, was more effective on nearly every plot.

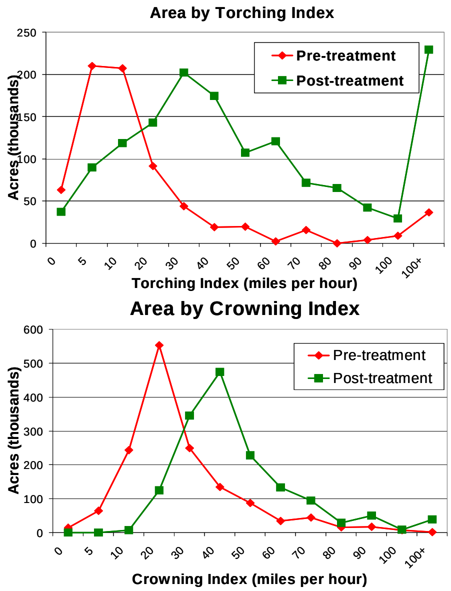

Figure 2–Distribution of acres by torching and crowning index for prescription B for all treatable acres in the southwest Oregon study area.

How much of the landscape could be feasibly treated?

After subtracting out the roadless areas, non-forest (e.g., agriculture and urban) and forests with insufficient basal area to implement either treatment only 1/3 of the Klamath ecoregion (i.e., 1.6 million acres) were potentially eligible (*fig. 3*). These were acres on which the distributions of torching and crowning indices in *figure 2* were based. But the reality was that even after accounting for revenue from sales of merchantable- and biomass-sized material, costs exceeded revenue most of the time, and there were only 270,000 acres of federal and non-federal land where estimated net revenue was positive.

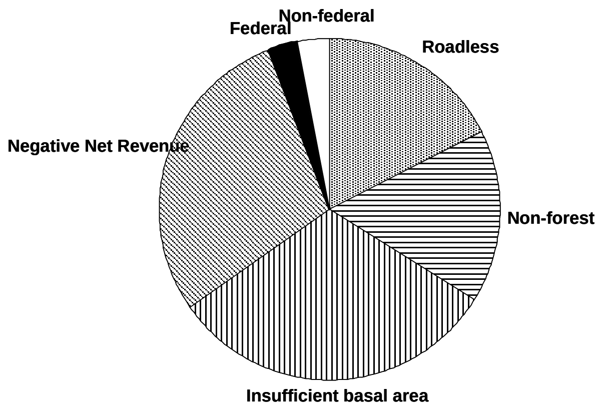

Figure 3—Fuel treatment opportunities that generate positive net revenue occur in only a small fraction of the 4.6 million acres of the Klamath ecoregion in southwest Oregon (the solid black and white slices).

Will there be enough biomass to fuel a power plant?

Under both prescriptions, nearly all of the removed material is in merchantable trees (*fig. 4*). Removals are nearly always greater for prescription B, most likely due to its specification of a lower residual basal area. Even under the most optimistic assumption that every landowner would treat every acre that could yield positive net revenue with prescription B, there would be sufficient biomass generated to fuel a 20 Megawatt biomass-based electrical generating plant for only 5 years.

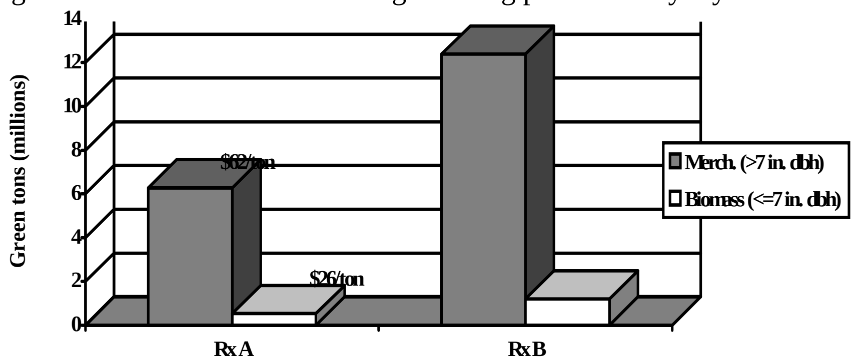

Figure 4—Amounts of merchantable- and biomass-sized material, by prescription (Rx), accumulated from acres that generated positive net revenue.

Where are the best places to site a power plant?

The best location depended on assumed product prices, prescription and one's objective (*fig. 5*). Maximizing biomass-sized accumulation gave one location, maximizing merchantable-sized accumulation another, and maximizing area treated or net revenue yet another. We evaluated every potential processing site on the 10 km grid, and found that the best locations were on the east side of the study area. In part, this reflected the locations of the forests needing treatment, but it also accounted for transportation infrastructure and lack thereof (i.e., the large designated wilderness and roadless areas in the southwest quadrant of the study area).

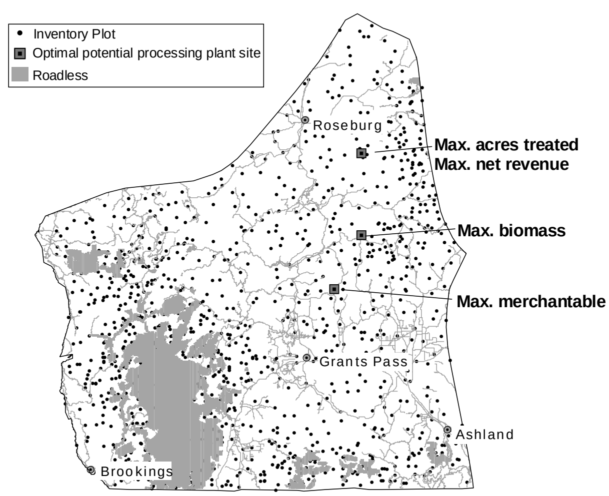

Figure 5—Map of the study area showing approximate locations of inventory plots and the locations of potential processing sites that maximize net revenue, biomass or merchantable material accumulation, or area treated.

Would a subsidy help?

A histogram of acres by net revenue class confirms that the vast majority of treatable acres would generate negative net revenue (*fig. 6*). Subsidies of up to $100 ac^{-1} would result in almost no increase in treated area, and even subsidies of $1,000 ac^{-1} would leave two thirds of the treatable landscape untreated. The highly negative net revenues were partly the result of the high costs of operating on steep ground: about half of the inventory plots had slopes over 40 percent. Every ton of biomass-sized trees on every acre had negative net revenue, so, in a sense, the harvest of merchantable-sized trees represented a subsidy already, although, in most cases, these removals also contribute to reducing fire hazard.

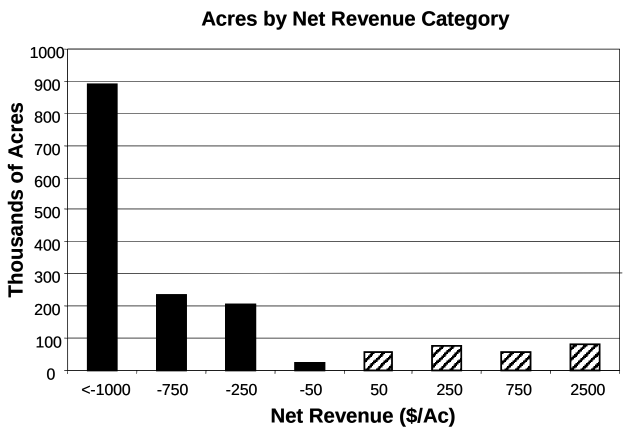

Figure 6—Frequency distribution of acres by net revenue class under prescription B for all treatable acres; solid bars represent acres with negative net revenue under the assumptions embedded in this analysis.

Comparing Policies

FIA BioSum presents policy makers with the opportunity to display various policy options and discuss them in terms of costs, volume produced, and effectiveness in fire hazard reduction. A partial set of possible policies for the pilot area is presented in *table 1*. Each row in the table represents an alternative, and the columns describe the details used in the analysis. The first two rows represent policies that focus on production of raw material where there are no restrictions on the plots that are selected for treatment, but in one net revenue is maximized, while in the other total recoverable biomass is maximized. When net revenue is maximized there is a net return of $211 million, and 290,000 ac are treated with significant fire hazard reduction on 162,000 of them. When biomass recovery is maximized, there is a net loss of almost $1.7 billion, and 1.5 million acres are treated with significant fire hazard reduction on nearly 1 million of them. The third policy shown in *table 1* includes creating a package of treatments where the net revenue is zero, there is no subsidy in total, treatment acres are maximized, and revenues from treatments with positive net revenue can be used to subsidize treatments with negative net revenue. This policy basically reflects the ability to trade goods for services. In the 4th row of *table 1*, only plots with positive net revenue are treated while maximizing biomass yield. The final three rows represent policies that repeat policies depicted in rows 2 through 4 but maximize area treated that results in significant fire hazard reduction rather than biomass yield. Some of these policies require subsidies but treat more acres or produce more biomass. Others require no subsidy but are less effective. Depicting alternative policies in this way can help policy makers, landowners, managers, and the public discuss outcomes in an objective and consistent manner.

Table 1–*Seven sets of alternative objective function/constraint combinations and model outputs for potential biomass processing site #181.*

Maximize	Constraint	Biomass generated	Area treated	Effective area treated[1]	Net Revenue
		10^6 tons	10^3 acres	10^3 acres	10^6 dollars
Net Revenue	None	1.3	290	162	211
Biomass	None	9.7	1490	943	-1697
	In aggregate Net Rev.≥0	4.7	559	350	0
	Each acre Net Rev.≥0	1.3	217	134	116
Effective area treated	None	5.2	1035	1035	-1053
	Each acre Net Rev.≥0	0.7	178	178	67
	In aggregate, Net Rev.≥0	2.7	636	519	0

[1] Includes only area represented by plots where torching index is improved by at least 20 mph.

Conclusions

In the Klamath ecoregion, utilization can pay the way towards fire resistant forests in some cases. It is the utilization of merchantable-sized material, not the biomass-sized material, that makes this possible. Only a small fraction of the landscape can be treated without infusions of considerable additional subsidy or incentives. Energy generation at least provides a place to haul biomass-sized material. Leaving such material on the ground in the woods would not be acceptable to most fuel managers, and disposal by burning would add other costs and risks.

A few caveats are necessary. FIA BioSum is not a spatially explicit model in the sense that it does not track the location of every acre or evaluate hazard from the perspective of the off-site values at risk (e.g., nearby homes in a wildland urban interface setting, or an adjacent, irreplaceable habitat) associated with any plot or acre in the landscape. Nor is there any dynamic component in this strategic fuel treatment model—all treatments are assumed to happen at the outset. Furthermore, considerable planning costs would likely be incurred before any kind of treatments occurred on the ground, and these are omitted from this analysis, not because they are unimportant but because their magnitude is unknown. And, the example policy comparisons outlined above are simplistic—maximizing acres treated makes little sense unless, for example, priority acre groupings (e.g., high initial risk or characterized as wildland-urban interface) are incorporated into the optimization framework, certainly a feasible extension of the analysis presented here.

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Estimating Forest Fuels in the Southwest Using Forest Inventory Data[1]

Krista M. Gebert,[2] Ervin G. Schuster,[3] Sharon Woudenberg,[4] and Renee O'Brien[5]

Abstract

Catastrophic wildfires occurring over the last several years have led land management agencies to focus on reducing hazardous fuels. These wildland fuel reduction projects will likely be concentrated in shorter interval, fire-adapted ecosystems that have been moderately or significantly altered from their historical range. But where are these situations located? What are their fuel characteristics? Who owns them? Describing fuel characteristics on these lands is not simple, but Forest Inventory and Analysis (FIA) data may be helpful. One objective of this study was to demonstrate the linkages between forest inventory data and hazardous fuel characteristics and to identify information gaps and needed relationships. A second objective was to estimate and contrast overstory and understory biomass, especially in high fire-risk areas. Restricting analysis to Arizona, New Mexico, and Utah, we estimate that understory biomass accounts for 4 to 8 percent (20 to 42 million tons) of total forest biomass. Additionally, we estimate that around 57 percent (619 million tons) of the estimated 1.08 billion tons of biomass is found on high fire-risk forest lands. Of these 619 million tons, approximately 434 million tons is associated with larger diameter ($\geq$ 10 inches) overstory trees, both live and dead, and most is found on non-reserved forestlands (lands where tree utilization is not precluded by statute or administrative designation) administered by the USDA Forest Service. We found that FIA data provides useful data on 92-96 percent of biomass, but we did encounter problems with estimating understory biomass. Some of the problems we encountered included a lack of widely applicable understory biomass equations, no equations for estimating tree seedling biomass using percent cover, and many biomass equations for shrubs that use diameter, a measurement that is not collected by FIA.

Introduction

Years of fire suppression have led to forest densification and unnatural buildups of forest vegetation. Consequently, a major focus of the National Fire Plan (USDA Forest Service and U.S. Department of Interior 2000) is to reduce hazardous fuels. According to the Cohesive Strategy (USDA Forest Service 2000), wildland fuel reduction projects will be concentrated in shorter interval, fire-adapted ecosystems that have been moderately or significantly altered from their historical range of fire frequency, severity, and density of understory vegetation. But where are these hazardous situations located? What are their fuel characteristics? Who owns them? This information is needed for broadscale assessments and fuel treatment planning.

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[1] A version of this aper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Space of the Southwestern United States, December 2–5, 2002, San Diego, California.

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Describing fuel characteristics on these lands is not simple, but forest inventory data may be helpful. The USDA Forest Service's Forest Inventory and Analysis (FIA) program is administered through five regional centers (Southern, Northeastern, North Central, Pacific Coast, and Interior West), each of which conducts resource inventories on all forested land within their geographical area. This system of inventory allows for a consistent set of data that can be used to produce a wide array of forest measurements, such as acres of forest and timber land, growing stock volume, and biomass, along with a wide range of resource characteristics (ownership, species, size, etc.). Consistency in data collection allows for nationwide comparisons of forest attributes. If one wishes to estimate, for example, the volume of small-diameter timber for various states, differences found between states can be attributed to actual, physical differences, not to inconsistencies in what data are available or how those data are collected or measured. Additionally, FIA data include forest inventory information on all ownerships, allowing comparisons among agencies or between Federal and private lands.

Our main objective was to assess the extent to which FIA data could be used to describe forest fuels on a broad scale, especially fuels associated with high-risk forest situations. Forest fuel, measured as tons of biomass, will be described for both forest overstory and understory vegetation. The overstory contains large and small diameter trees. Understory vegetation focuses on three layers. The lowest layer, from ground to the knee, contains grasses, forbs, and low shrubs. The next layer, between the knee and eye level, contains forbs and medium shrubs. The highest layer includes plants above eye level, usually seedlings, saplings, and tall shrubs only. Information gaps and needed relationships are identified.

Methods

We wanted to use FIA data to describe forest fuels, not only in the overstory, but also in the understory. To do this, we needed to determine the types of information collected by FIA and how that information could be used to describe forest fuels. We decided upon biomass as a measure that could be used to estimate amounts of forest fuel and allow us to compare the overstory and understory. Analysis was restricted to Arizona, New Mexico, and Utah, the three southwestern states for which we could obtain relatively recent and complete FIA data, but the same process could be applied to other FIA regions. A major focus was on forest lands with a high potential for catastrophic wildfires.

Data Compilation

We began data compilation by investigating the types of FIA data available. FIA collects and estimates a large variety of information on the forest overstory (trees greater than 1" diameter) including biomass for both live and dead trees. The term diameter in this publications refers to the diameter of the tree at the point of measurement, which is either breast height or root collar depending upon species. (For more information about IWFIA, see http://www.fs.fed.us/rm/ogden/). However, the only widely-available understory information is percent cover by vegetation layer (0-1.5 feet, 1.6-6 feet, and 6.1 feet or greater) and life form (trees, shrubs, grasses, forbs), and this information is collected for live vegetation only. Within the last several years, information on down woody debris has begun to be collected, but the collection of this information is in its infancy and, therefore, was not available for any State in its entirety. This information is collected on what are called "P3" or Phase 3 plots, which are a by-product of the Forest Health Monitoring/FIA merger in 2000. Approximately one in 16 FIA plots is also sampled for P3 data using one to two additional field personnel to collect data items and samples, including information on down woody debris (Rogers, personal communication).

Once we determined availability of fuels information in the FIA data, we selected a metric of forest fuels that would allow us to compare fuels in the overstory to those in the understory. We decided on biomass for several reasons: (1) fuel loadings are often measured in terms of biomass, (2) biomass was already estimated for the overstory by FIA, (3) biomass is a useful measure of potential material for bio-based products, and (4) we knew of equations for computing biomass of forbs and grasses using percent cover. We then obtained overstory data from the Interior West Forest Inventory and Analysis (IWFIA) unit in Ogden, Utah. For the understory data, IWFIA calculated average percent cover by layer and life form for each inventory plot, averaged across all subplots on which information was collected. Understory data were then merged with overstory plot data.

We were particularly interested in assessing the amount of forest fuels on lands at risk for severe wildfires. The Cohesive Strategy states that its aim is to "reduce losses and damages from wildland fires by concentrating treatments where human communities, watersheds, and species are at risk" (USDA Forest Service 2000). To determine the location of these areas, we used a map of coarse-scale spatial data developed by Hardy and others (1999). These data designate both Historic Natural Fire Regime (historic fire frequency and severity) and Current Condition Class (an indication of the degree of departure from the historic fire regimes). This map was sent to IWFIA where it was overlaid with FIA plots, and each plot was assigned a Fire Regime/Condition Class depending upon where the plot fell on the map. We designated plots falling in Fire Regime I or II and Condition Class 2 or 3 as "high risk." According to Schmidt (2002), almost all of the lower elevation zones in the United States, which are the areas most affected by human intervention and where resources and communities are at highest risk, fall under Fire Regimes I and II. In addition, Condition Classes 2 and 3 have been moderately or significantly altered from their historical range and, therefore, are more at risk for severe wildland fires and generally require some degree of mechanical treatment before prescribed fires can be successfully used to control fuels (Schmidt and others 2002). It should be noted that FIA plot information could have been used directly to evaluate forest conditions and assess fire risk, but that was not feasible given the time frame and scope of this study; that procedure is more suitable to smaller land areas.

We encountered several problems during data compilation, including the lack of useable information on down woody debris and no dead understory vegetation information for the states analyzed. Additionally, several plots were missing understory vegetation information due to snow cover at the time the plot was field visited. For these plots, we used the average percent cover by layer and life form for the appropriate forest type.

Calculating Biomass

To calculate the biomass (in oven-dry tons) of live and dead overstory vegetation (trees $\geq$ 1" diameter), we used the algorithms designed by FIA for use with their data (Miles and others 2001); this was a straightforward process. However, for understory vegetation, we had to use equations relating biomass to percent cover. This turned out to be much more difficult than originally envisioned.

Several problems were encountered in converting FIA understory vegetation attributes of percent cover by layer and life form to biomass. First, we were unable to locate any equations for estimating the biomass of tree seedlings based on percent cover or the number of seedlings. This meant we were unable to estimate biomass of tree seedlings (trees < 1" diameter), thus leaving a gap in our analysis. Second, many of the equations for estimating biomass of shrubs or other woody plants used diameter as an estimation parameter (Alaback 1986, Reeves and Lenhart 1988). Shrub stem diameters were not collected by FIA. Third, most biomass equations for the understory were developed for very specific geographical areas and specific species (Alaback 1986, Alexander 1978, Means and others 1996, Reeves and Lenhart 1988). Little work has been done to develop more generally applicable equations or to validate them for estimating biomass of other species.

In the end, we used understory biomass equations that were based on a mix of species, rather than equations developed for one specific species. These mixed-species equations had been developed for other regions of the country and their applicability to the southwest is questionable; however, we felt these equations could be used for illustrative purposes and give some idea of the relative understory biomass of the three states. For grasses and forbs, we used equations developed by Mitchell and others (1997) for estimating biomass based on percent cover. For shrubs, we adapted an equation developed by Olson and Martin (1981) that used two parameters, percent cover and height (*table 1*). Although FIA does not specifically measure height of understory plants, percent cover by vegetation layer is recorded. We used the midpoints of the first two layers (0.75 feet, 3.8 feet) as an estimate of the height of the understory vegetation. For the third layer ($\geq$ 6.1 feet), we assumed that, on average, shrubs ranged from 6.1 feet to around 15 feet, so we used a midpoint of 10.5 feet.

Table 1— *Regression equations used for estimating biomass of forest land understory vegetation in Arizona, New Mexico, and Utah*

Life form	Parameters	Equation[1]	Source
Shrubs	% Cover (x_1) Height (x_2)	$g/0.5m^2 = -.62689 +$ $(0.05778^* x_{1*} x_{2})$	Olson and Martin 1981
Forbs	% Cover (x_1)	kg/ha = 13.66* x_1	Mitchell and others 1987
Grasses	% Cover (x_1)	kg/ha = 8.17*x_1	Mitchell and others 1987

[1]Dependent variable (y) = oven-dry plant weight. Units varied, but all results were converted to pounds/acre.

We encountered an additional problem in the calculation of biomass for saplings --- inclusion of saplings in the FIA understory data varied by inventory year. Therefore, when saplings were included in the understory, we subtracted them out of the calculations of overstory biomass to prevent double counting.

Results

The results presented in this section are largely illustrative of the types of forest fuel information that can be extracted from FIA data. Estimates of area and overstory biomass are standard outputs from the FIA database. Understory estimates of biomass involve more extrapolation. We also provide information on the relative amounts of biomass found on lands at risk for severe wildland fires by merging the FIA data with current condition class and historic natural fire regime information.

There are 51.8 million acres of forest land in Arizona, New Mexico, and Utah (*fig. 1*). Forest land is land at least 10 percent stocked with trees. Forest land is made up (by IWFIA definitions) of timberland and woodland. The majority of the forest land in these three states (68 percent) is classified as woodland. Woodland is forest land where the majority of stocking is in woodland tree species---species such as pinyon and juniper that are not usually converted into industrial wood products. Only one-fourth of the forest land is classified as timberland. Timberland is forest land where the majority of stocking is in timber tree species---species such as ponderosa pine and Douglas-fir that are used for industrial wood products. Each of the three states have more than 10 million acres of woodland, with Arizona containing both the largest area of woodland and hence, the largest area of forest land. None of the states have more than 5 million acres of timberland. Only 8 percent of the forest land has been withdrawn from tree utilization through statute or administrative designation (reserved land). Reserved land could include either timberland or woodland--that distinction was not used in this paper.

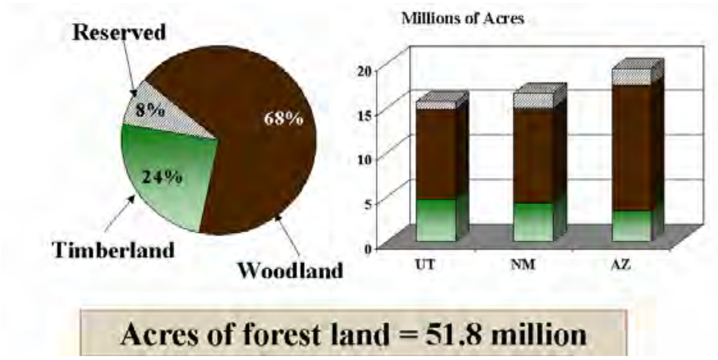


Figure 1—Acres of forest land in Arizona, New Mexico, and Utah

Total biomass found on forest land in these three states comes to an estimated 1.08 billion tons (not counting tree seedlings) (*fig. 2*). Most of this, 96 percent, is found in the overstory, and the majority of the overstory biomass comes from live trees. Interestingly, although Arizona contains the largest area of forest land (*fig. 1*), Utah contains the largest amount of total biomass, more than 390 million tons, and also contains the largest amount of understory biomass.

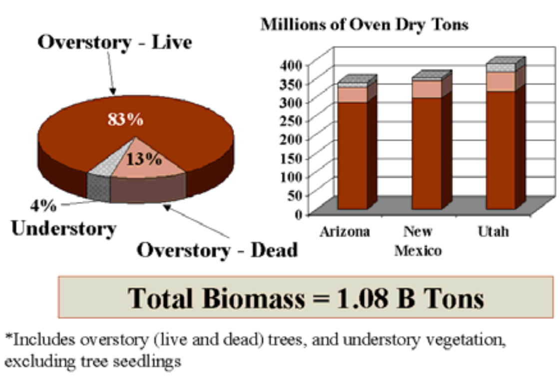


Figure 2—Estimated biomass of overstory and understory vegetation*on forest land in Arizona, New Mexico, and Utah

Estimates of understory biomass do not include tree seedlings (trees < 1" diameter) due to the lack of equations for estimating seedling biomass. Just how much did this likely affect our estimates? We found one reference that suggested that seedling biomass comprises anywhere from 5 to 50 percent of total understory biomass (Telfer 1971). To test how the missing seedling information affected our estimates, we recalculated our estimates by including tree seedlings as 5, 25, and 50 percent of total understory biomass (*table 2*). Assuming that tree seedlings account for 5 percent of understory biomass, there would be no noticeable affect on understory biomass estimates. If tree seedlings were 25 percent of understory biomass, the total understory percentage increased to 5 percent, as opposed to 4 percent with no tree seedlings included. When we assumed that tree seedlings make up 50 percent of understory biomass, the percentage of biomass accounted for by the understory increased to 8 percent. At this level, the importance of the understory began to vie with that of the dead overstory trees, and our overall biomass estimates increased from 1.08 billion tons to 1.13 billion tons.

Table 2— *Sensitivity of forest land understory biomass estimates to missing tree seedling information: Arizona, New Mexico, and Utah.*

Tree seedling assumption	Estimated understory biomass using tree seedling assumptions	
Pct. tree seedling biomass of total understory biomass	Total understory biomass	Understory biomass as a pct. of total biomass
	Millions of tons	Percent
w/o seedlings	20.8	4
5	21.9	4
25	27.8	5
50	41.7	8

A major focus in this study was forest lands at risk for catastrophic wildfires because these are the areas where fuel treatments are most likely to occur. Using the maps developed by Hardy and others (1999) to classify FIA plots according to historic natural fire regime and condition class, we estimated the amount of biomass on these lands. Results show that of the estimated 1.08 billion tons of forest land biomass, approximately 57 percent (619 million tons) is found on lands we classify as high risk (Fire Regime I or II, and Condition Class 2 or 3); 23 percent is found in the highest risk category (Condition Class 3) (*fig. 3*).

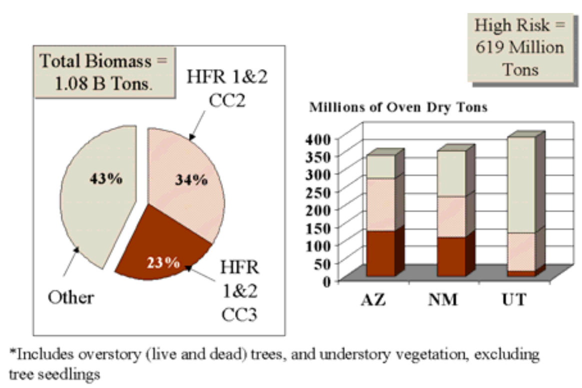


Figure 3— Estimated biomass of overstory and understory vegetation* on high fire risk forestland in Arizona, New Mexico, and Utah.

This is not surprising considering the amount of land falling in these categories (*fig. 4*). More than two-thirds of the forest land in Arizona and New Mexico falls into these high-risk categories. For Utah, the percentage is much smaller, with less than one-third of the forest land classified as high risk. In fact, the breakdown of biomass by state shows that Utah, although having the largest amount of forest land biomass, has only 30 percent of its biomass in these high-risk areas and only a very small percentage (less than 5 percent) in the highest risk category. Conversely, 81 percent of the forest land biomass in Arizona is found in these high-risk areas, with about an equal amount falling on Condition Class 2 and Condition Class 3 lands. In New Mexico, the biomass is about evenly split between the two fire risk categories and all other lands.

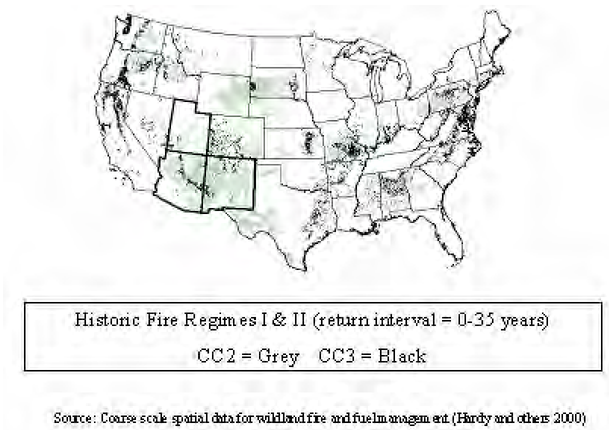


Figure 4— Historic Fire Regimes I & II, Condition Classes 2 & 3

Next, we looked in more detail at biomass in these high-risk areas, focusing on overstory versus understory, ownership, and reserved status. Looking first at vegetation layers (*fig. 5*), the majority of the biomass found on these high-risk lands is in the overstory (96 percent) with 84 percent coming from live overstory trees and 12 percent from dead trees. Of the 4 percent of understory biomass, half is found in Layer 2 (1.6 - 6 feet). However, it is important to note that because this estimate does not include tree seedlings, the distribution of biomass among layers is affected. Yet, when we looked at the plot data, more plots reported tree seedlings in Layer 2 than the other two layers, so the relative ranking may be correct.

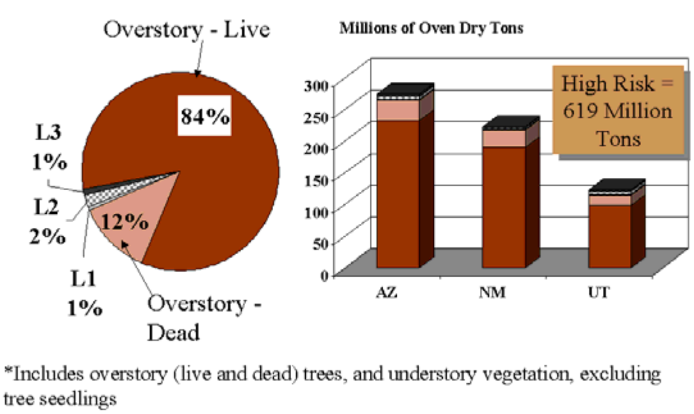


Figure 5— Estimated biomass of overstory and understory vegetation*on high fire risk forestland in Arizona, New Mexico, and Utah by vegetation layer

Looking at the overstory separately (*fig. 6*), there are at least two points of note. First, 73 percent of the overstory biomass is found in trees (both live and dead) 10 inches or greater in diameter. If stand density is a concern, it is difficult to reduce it by removing only small trees when such a large proportion of the biomass is in larger trees. Second, the relative percentage of dead to live trees does not change much with size. Twelve percent of the biomass of the larger trees comes from dead trees, while the percentage of dead tree biomass is 11 percent for the smaller trees. There is little difference in the distribution of the overstory in the three states.

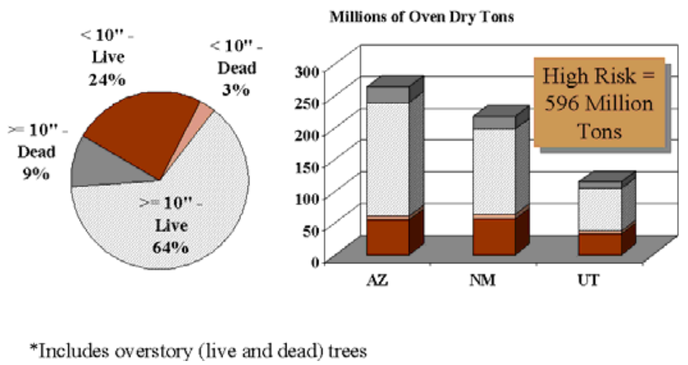


Figure 6— Estimated biomass of overstory vegetation* on high fire risk forestland in Arizona, New Mexico, and Utah, by size

Turning to understory biomass, nearly 90 percent comes from shrubs, followed by grasses and then forbs (*fig. 7*). The breakdown by state shows, however, that the relative ranking of states has now changed. When looking at total biomass on these high-risk lands, or just overstory biomass (*fig. 5 and fig. 6*), the ranking from largest to smallest amount was Arizona, New Mexico, and Utah. When focusing on understory biomass only, Arizona still ranks first, but Utah overtakes New Mexico to rank second in terms of understory biomass on high-risk lands.

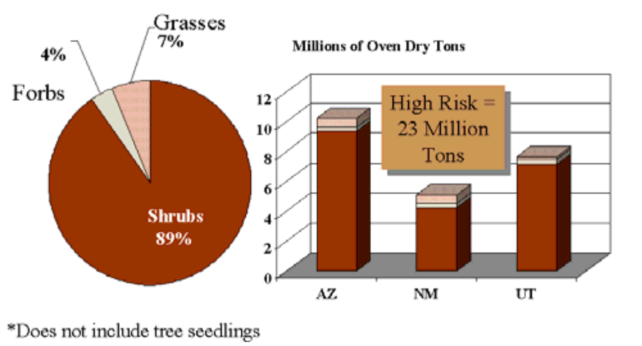


Figure 7— Estimated biomass of understory vegetation* on high fire risk forestland in Arizona, New Mexico, and Utah, by life form

Aside from the physical characteristics of forest fuels on high-risk lands, other important dimensions in the management of these lands are ownership and reserved status. Nearly 60 percent of the biomass occurs on lands administered by the Forest Service, with another 28 percent being found on private land (*fig. 8*). A comparison of states shows that Arizona has a larger percentage of this material occurring on lands administered by Federal agencies other than the Forest Service or the Bureau of Land Management (mainly the National Park Service) than the other two states. In Utah, a larger percentage of biomass is found on lands administered by the Bureau of Land Management than in either Arizona or New Mexico. Most of the biomass in these three states (88 percent) occurs on lands currently available for timber utilization (non-reserved lands) (*fig. 9*).

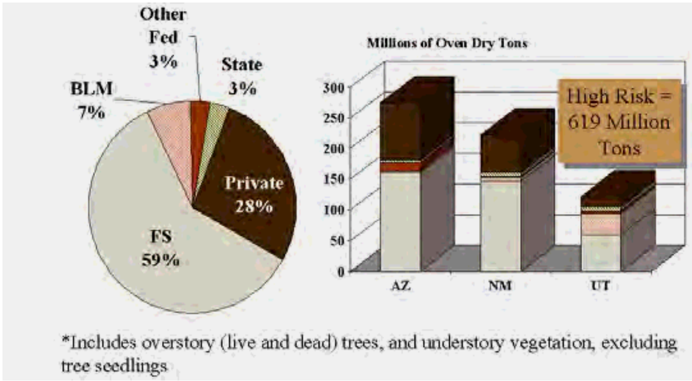
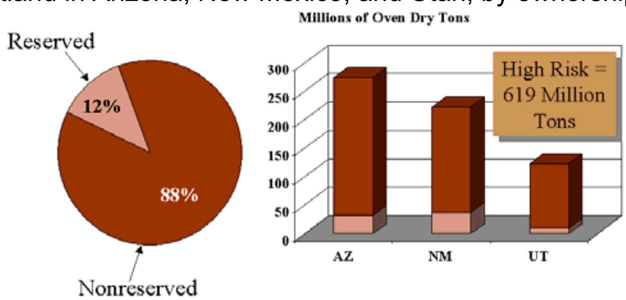


Figure 8— Estimated biomass of understory and overstory vegetation* on high fire risk forestland in Arizona, New Mexico, and Utah, by ownership



*Includes overstory (live and dead) trees, and understory vegetation, excluding tree seedlings

Figure 9— Estimated biomass of understory and overstory vegetation* on high fire risk forestland in Arizona, New Mexico, and Utah, by reserved status

Conclusion

Broadscale assessments of forest fuels would be useful for fuel treatment and bio-based product planning. The general conclusion of this study, however, is that more research is needed to accurately estimate biomass from FIA understory vegetation data. While overstory biomass can be adequately estimated using FIA data, the understory biomass estimates presented here are only illustrative, giving a general impression of the amount of understory biomass in Arizona, New Mexico, and Utah. These understory estimates are not completely satisfactory. They are based on equations developed for other areas of the country, for different species, and do not include tree seedling biomass. Biomass of dead understory plants is also not available.

Given the fact that the understory appears to be a fairly small component of total forest biomass, FIA data can be used to provide reasonable estimates of amounts and location of forest fuels for broadscale assessments. However, to develop more complete estimates of biomass, including the understory, research is needed to develop equations for estimating the biomass of tree seedlings based on percent cover or the number of seedlings. Additionally, there is a lack of equations for estimating the biomass of shrubs based on percent cover; many of the available equations for shrub biomass use other parameters, such as diameter, which is not a measurement collected by FIA. Finally, and perhaps most importantly, research is needed to test the applicability of species- and area-specific understory biomass equations to other species or areas and/or to develop more generally applicable understory biomass equations.

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The Maintenance of Key Biodiversity Attributes Through Ecosystem Restoration Operations[1]

Robert W. Gray[2] and Bruce A. Blackwell[3]

Abstract

The requirement to manage for key biodiversity attributes in dry forest ecosystems is mandated in the Forest Practices Code Act of British Columbia. These attributes include snags, large old trees, and large organic debris. In the Squamish Forest District dry forest restoration activities center on the use of thinning operations followed by prescribed fire to restore stand structure and species composition to conditions closer to the historic range of variability. Various strategies have been tested to retain or create key biodiversity attributes. These strategies include wrapping fire-scarred trees with fire shelter material, digging firebreaks around attributes, using "avoidance firing" ignition techniques, and setting prescription limits around fuel moisture content. Some strategies have proven to more successful than others. We present our findings on cost effectiveness and retention success for several attributes and treatment strategies plus a discussion of recommended policy changes to make expectations of retention more in line with operational realities.

Introduction

Many western interior forested ecosystems in North America are considered to be in an unhealthy state. The decline in ecosystem health has been attributed to fire exclusion, livestock grazing, excessive harvesting, and the introduction of exotic species (United States General Accounting Office 1999). Intensive management is required in order to restore ecosystem structure, composition, and functions to a more sustainable condition (Covington and others 1997). Ecosystems are considered best able to respond to disturbances if they are resilient, sustainable, and biologically diverse (Bourgeron and Jensen 1994).

Most definitions of biodiversity include three distinct components: composition, structure, and function. The compositional component represents the variety of fauna and flora within an area. The structural component refers to the arrangement of fauna and flora, including their spatial and age-class distribution. The functional component characterizes the processes and mechanisms occurring within an ecosystem including, but not limited to, nutrient cycling, decomposition, and energy flows (Franklin 1988). The task for resource managers involved in ecosystem restoration is to either preserve biodiversity through static set asides, or to promote biodiversity through restoration efforts in a dynamically-managed landscape (Everett and others 1996).

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Identifying the appropriate location and proportion of attributes to retain or create is the first step in planning restoration operations. The second step is determining how, through restoration strategies of mechanized thinning and prescribed burning, the various associated attributes can be retained or created. Since 1998, the Squamish Forest District Small Business Forest Enterprise Program has carried out a variety of ecosystem restoration trials (Gray and Blackwell, this volume). Employing an adaptive management philosophy to dry forest management has enabled the targeting of specific restoration issues and the building of monitoring plans to address these issues in subsequent operations. This paper presents the results of adaptive efforts to retain and promote a variety of biodiversity attributes, including Coarse Woody Debris (CWD), snags, and large diameter green trees through mechanical thinning and prescribed burning operations in the southern interior of British Columbia.

Study Area

The Haylmore Creek drainage is located in the northeast corner of the Squamish Forest District in southwestern British Columbia. Forested ecosystems range from the Interior Douglas-fir wet warm subzone (IDFww) at the valley floor and lower half of the slope to the Engelmann Spruce/Subalpine Fir moist warm subzone (ESSFmw) above approximately 1400 m. Valley floor elevation is 300 m, with the highest points in the local Cayoosh Range exceeding 2200 m. Climate in Haylmore Creek is characterized as continental with a mean annual precipitation of 549 mm. Mean monthly air temperature ranges from a low of -1.4°C in January to a high of 23.7°C in July. Summer droughts are not uncommon. This general area is referred to as the Coast-Interior Transition Zone due to the climatic overlap of the two systems.

Low elevation forests in Haylmore Creek contain a mixture of Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) and ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) on dry sites and predominantly Douglas-fir, with a minor component of western redcedar (*Thuja plicata* Donn ex D. Don) and paper birch (*Betula papyrifera* Marsh.), on mesic sites.

Methods and Results

Each attribute investigated and managed is described individually in this combined section. Methodology used in inventory and restoration operations is described followed by the operation results.

Coarse Woody Debris

Coarse woody debris (CWD) plays a significant role in forest ecosystem ecology, including the provision of habitat for many autotrophic and heterotrophic species; providing a food source for many decomposer bacteria and fungi; and acting as a sink for important nutrients (Stevens 1997). For the purposes of this report we characterize CWD as large (>15 cm), downed logs and describe their condition using five decay classes (British Columbia Resources Inventory Committee 1997).

The retention of adequate quantities of CWD throughout the restoration process,and into the future is of concern to researchers who hold the belief that long-term site productivity and wildlife habitat may be impacted if too much CWD is removed during the restoration process (Tinker and Knight 2001). However, questions arise over how much CWD should be retained in these ecosystems and how the quantity of CWD will affect fire severity during either prescribed burning operations or wildfires (Brown and others 2003).

Retaining a predetermined amount of CWD through the thinning phase of restoration is not difficult. However, retaining this material through the prescribed burning phase of restoration is problematic due to its flammability.

Consumption of material in decay classes 1 and 2 was only 9 percent, while the consumption rate for more advanced decay classes was 31 percent. Decay class 1 and 2 material was sound and often still suspended off the ground. Decay class 3 to 5 material was spongy, elliptical in shape, and fully in contact with the ground. Fuel moisture content (FMC) at the time of ignition ranged from 14 to 17 percent in the decay class 1 and 2 material and 20 to 25 percent for the decay class 3 to 5 logs. FMC was measured using a protimeter on the outside of the log. Higher consumption rates in decay class 3 to 5 CWD were most likely explained by the low bulk density of material, high surface-area-to-volume ratios, high flammability due to resin impregnation, and oxygen availability between particles. These qualities contributed to material in decay classes 3 to 5 exhibiting increased flammability when compared to decay class 1 and 2 material. All pieces sampled were either Douglas-fir or ponderosa pine.

Snags

Dead standing trees are recognized as a vital component of wildlife habitat, providing perching, feeding, nesting, over-wintering, and hiding structure for birds, and nesting/denning, feeding, and over-wintering habitat for mammals (Bull and others 1997). Wildlife value is often dependent on the stage of decay. Retaining dead trees, especially tall, large diameter snags, through the restoration process is difficult owing to worker safety regulations, structural soundness during thinning and yarding operations, and flammability of the structure. In restoration treatments in British Columbia, snags are either felled or retained in designated Wildlife Tree Patches (Ministry of Environment, Lands and Parks and Ministry of Forests 1995). The patch dimension is based on snag height and its structural soundness.

Analysis of snag retention on restoration units in this study was limited to thinned-and-burned and burned-only units. Any of these units that included harvesting had all snags felled as per Worker’s Compensation Board (WCB) regulations (Workers’ Compensation Board of British Columbia 1997). Prior to treatment, the density of large (>50 cm) snags averaged two per hectare, with most classified in conditions five to nine.

Snag retention was measured on two spring burns; one carried out in 2001 (burn only), and the second in 2002 (thin and burn). The 4.3 ha burn only unit contained eight snags prior to treatment, while the 11 ha thinned and burned unit contained 22 snags. Snag survival was 50 percent in the burn only unit and 36 percent in the thin and burn unit. Snags lost in the burn only unit were attributed to snags burning and falling over, and snags that had to be felled during project mop-up as per worker safety regulations. All 14 of snags lost in the thin and burn unit were felled prior to burning due to safety issues. In the thin and burn unit, thinning crews would be working in the vicinity of snags for several days, and as a result 64 percent of snags were felled. The only option open for retaining the remaining eight snags was to designate “no work” zones around two small snag patches. In these patches, ingrowth trees, which were the target of the restoration treatment, could not be thinned.

Live Green Trees

The category "live green trees" refers to the older tree cohort left following restoration thinning. This is in contrast to smaller diameter, younger cohort trees that are retained as future legacy structures. Live green trees in the study area are typically characterized by large diameters (>70 cm), clear boles, thick bark, moderate crown ratios (0.4 to 0.6), high crown areas, large diameter limbs, and often fire scars, or cavities. There may be large accumulations of bark scales, needles and cones, especially in the case of ponderosa pine, at the base of these trees. In addition to their aesthetic and genetic values, these trees are considered to have very high wildlife values.

Retaining live green trees during the thinning phase of restoration is achieved through marking to leave guidelines. On rare occasions trees are lost to falling or yarding damage, or they are intentionally felled because they are located in a yarding corridor. However, the proportion of live green trees lost during this phase of restoration operations is minimal.

Retaining live green trees during the prescribed burning operations is more difficult. Experience has shown that there are significant issues of survivability based on tree species, whether or not the tree contains an open fire scar, and the fuel complex in the unit. An analysis of differences between legacy retention in thin and burn units versus burn only units yielded a noticeable difference between the two treatment types. Mortality of legacy trees in thin and burn units reached 19 percent by year 2 following the burn. Both Douglas-fir and ponderosa pine experienced a mortality rate of 19 percent, which in the case of Douglas-fir was partially attributed to Douglas-fir bark beetle (*Dendroctonus pseudotsugae* Hopkins) and structural failure. Immediately post-burn (all burns were carried out in the spring) a number of trees were attacked by bark beetles, but only 5 percent were killed. All mortality was associated with the thin and burn unit. Fire damage and associated bark beetle attacks are well documented in the literature (Edmonds and others 2000). In order to limit tree mortality associated with fire and beetle interactions precautions were taken during the burn to limit crown scorch. These are principally burning under higher moisture contents, and limiting ignition strip widths.

Douglas-fir legacy trees did not experience significant crown scorch in either treatment type; however, in many cases fire ran up the bole burning resin on the outside of the bark. Structural failure of trees was primarily due to fire entering large rotten limbs and burning into rotten heartwood. If the bole was weakened by internal burning, the top of the tree would break off, creating a large diameter case-hardened snag. This resulted in the mortality of 14 percent of the Douglas-fir in the thin and burn unit and 6 percent of the Douglas-fir in the burn only unit.

Ponderosa pine mortality for all causes was 19 percent in the thin and burn treatment and 6 percent in the burn only treatments. Of those killed, seven percent was due to a combination of red turpentine beetles (*D. valens* LeConte), western pine beetles (*D. brevicomis* LeConte), and mountain pine beetles (*D. ponderosae* Hopkins). In some cases trees were not attacked until the year following the burn. Bark beetles attacked only 2 percent of ponderosa pine trees in the burn only treatment. In both treatment types crown scorch was minimal.

Structural losses are the most significant management concern for ponderosa pine legacy trees. Unlike Douglas-fir, where fire runs up the bole and enters heartwood through rotten branches, fire seldom runs up the bole on ponderosa pine.

Open cavities at the base of the bole, caused by repeated fire-scarring, is the structural weak point of these trees. Following an earlier burn in 1999, where all fire-scarred pine were lost, an attempt was made to mitigate future losses through a combination of fuel removal at the base of the trees and by covering the fire scar area with an ember barrier. Fuel removal limited the heat residence time at the base of the tree, while the barrier prevented embers from entering the scar cavity.

In a heavily thinned restoration unit burned in the spring of 2001, 123 trees were wrapped with fire shelter material purchased from Cleveland Laminating Ltd. Of the 123 trees, 22 (18 percent) were Douglas-fir, and 101 (82 percent) were ponderosa pine. The shelter material was placed over the wound and stapled into place. The bottom of the material was anchored to the ground with rocks. Total cost of the project was $21.09/tree (Cdn): $848.00 for two rolls (1 yd x 300 yd) of material, and $284.00/day x 6.2 man-days to apply the wrap. Total mortality post-burn was 30 trees (24 percent of the total): seven Douglas-fir (32 percent of fire-scarred trees) and 23 ponderosa pine (23 percent of fire-scarred trees). All seven Douglas-fir were burned through the wrap and then burned out at the base and fell over. Of the 23 pine that were killed, 13 (57 percent) had been burned through the wrap and were hand felled because they were deemed a worker hazard, while 10 (43 percent) burned through the wrap, burned out the base, and fell over.

In an attempt to determine whether the wrap would be more effective in lighter fuels, the experiment was repeated in the spring of 2002 on an 11 ha thin and burn restoration unit (*Table1*). In this unit the majority of Douglas-fir ingrowth trees were felled and scattered. A total of 18 trees were randomly located and wrapped in the unit; 14 were pine and four were Douglas-fir. Surface fuels were also pulled back 1 m from the bole of the tree. No wrapped trees were lost to the burn in this second experiment.

Table 1—*Fuel bed characteristics for the heavily thinned and lightly thinned restoration treatment types.*

	Fuel loading (kg m^{-2})						
Treatment Type	**0-0.6 cm**	**0.6-2.5 cm**	**2.5-7.5 cm**	**<7.5 cm**	**>7.5 cm rotten**	**>7.5 cm sound**	**Total**
Heavy							
pre-burn	1.66	0.65	1.86	4.17	1.44	3.48	9.09
post-burn	0.29	0.16	0.70	1.15	1.19	3.23	5.57
Light							
pre-burn	0.38	0.58	0.16	1.12	0.31	0.00	1.43
post-burn	0.31	0.34	0.00	0.65	0.31	0.00	0.96

The survivability of legacy trees is heavily influenced by the fuel bed characteristics in the restoration units. There was a significant difference (p=0.011) in total fuel loading between the two treatment types (*table 1*). Pre- and post-burn analysis indicates that there was a significant difference in burn intensity between the two units. High levels of fine fuel consumption–82 percent, 76 percent, and 63 percent for each of the 1-hr, 10-hr, and 100-hr fuel size classes–were recorded in the heavily thinned unit compared to very low levels–18 percent, 42 percent, and 100 percent–in the lightly thinned unit.

Discussion

The retention of coarse woody debris, snags and large diameter old green trees, are important elements of biodiversity in dry forest ecosystems. Tactically managing these attributes is problematic when restoration activities reach the operations level. Difficulties arise when trying to maintain or promote biodiversity attributes that may be a function of current biological conditions, or are associated with inflexible government regulations. Both were encountered when carrying out restoration operations in Haylmore Creek. Beginning with coarse woody debris, we present a set of discussion points detailing where we had successes and failures and recommend some possible solutions to these problems.

While it is recognized that CWD has an important function in these ecosystems, maintaining appropriate levels of it while not jeopardizing other resources is key. Too much CWD on a site that will be exposed to frequent fire as part of an ongoing restoration program can lead to long-term site productivity impacts. Maintaining adequate amounts on the site through the thinning phase of restoration is not difficult. Logging contractors rarely remove large old logs unless they are very solid or are a valuable species such as western red cedar. Additional amounts of CWD are added to the site in the thinning phase, most of which is very solid decay class 1 to 2 material. Snags not located in Wildlife Tree Patches or “no work” zones will be felled and incorporated into the CWD volume. Through the burning phase, the more decayed and pitch-impregnated material has a high probability of being consumed. This process is more pronounced during drought cycles. Very little in the way of mitigation can be done to prevent this. Less decayed material, however, has a very high probability of surviving the first burn. As we have seen on our projects, there is an additional input of large CWD to the site following the burn due to the loss of green retention trees.

Retaining snags was less a function of flammability than of regulations guarding worker safety. All snags in thinning units had to be felled unless they were located in “no work” zones. In restoration units that did not include pre-burn thinning, snags were lost if they were deemed to be “unsafe” to work around during mop-up. In these cases snags had caught fire and were structurally unsound. The only other option open for retaining snags is to locate them in reserve patches. Our hypothesis concerning this strategy is that reserve patches don’t mitigate the factors causing accelerated decomposition of snags; namely, the high moisture content in closed canopy reserves provides a hospitable environment for insects, bacteria and fungi. These thickets of trees surrounding snags need to be thinned in order to dry out snags and make them less hospitable to decay agents. The only positive note regarding snag management is the recruitment of new snags from the live green tree population. Many of these old, case-hardened trees exhibit fairly high longevity (Smith 1999).

There are several fairly inexpensive but successful strategies available for mitigating the losses in live green trees during the burning phase of restoration operations. These include fuel pull back at the base of the tree, wrapping scar cavities with fire shelter material, applying certain ignition techniques around these trees and aggressive mop-up immediately following ignition. Longer-term post-burn issues around bark beetle attacks could be addressed with pheromone traps, which can be successful for certain species of bark beetle. Retaining additional future legacy trees is another option if there are significant concerns over the losses of these trees.

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Fuel Management Strategies in 60-Year-Old Douglas-Fir/Ponderosa Pine Stands in the Squamish Forest District, British Columbia[1]

Robert W. Gray[2] and Bruce A. Blackwell[3]

Abstract

The restoration of dry forest ecosystems in the Squamish Forest District in the past has focused on treating stands with no prior history of selective harvest and containing a large population of remnant historical stand structure. Many 60 to 90 year old stands that date to turn of the century selective harvest operations also exist in the district. These stands contain very high densities of Douglas-fir and ponderosa pine and few historical structures. Silviculture prescriptions were developed to thin these stands to a remnant structure that would be more resilient to future wildfires that may occur between the maintenance prescribed burn schedule. Several strategies were employed, including manual thinning only, thinning and fuel wood harvest, and thinning and mulching. All thinning treatments were followed by prescribed fire. From a strategic perspective, the treatment results indicate that options exist for creating wildfire resilient stands. The best overall option is to thin, remove the material, and then burn the site.

Introduction

Historically, dry forests in the northern end of the Squamish Forest District, Haylmore Creek, British Columbia (BC) contained a higher proportion of ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) than Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) and were structurally dominated by well-spaced, large diameter trees. Structure and species composition were maintained by a short-interval, low- to moderate-severity fire regime (Gray 2001). The historic dry forest type was located on southerly aspects at low to mid-elevations.

Forest stands in Haylmore Creek have experienced a significant increase in stand density and an increase in aerial and surface fuels in a pattern similar to many other sites in western North America (Covington and others 1994, Harrod and others 1999, Gray and others 2004). The homogenization of surface and aerial fuels means that a wildfire can travel for long distances at high-intensity levels and result in large-scale environmental damage. Fire suppression personnel, public and private property, natural resources, and local economies within these landscapes are all at significant risk from large, high intensity fire within these ecosystems. Where fire once promoted biodiversity and heterogeneity, it now threatens biodiversity and promotes homogeneity of structures and species (Martin and Sapsis 1991).

In the fall of 2001 the Squamish Forest District Small Business Forest Enterprise Program (SBFEP) began a strategic assessment of wildfire threat (Blackwell and

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others 2004) to identify high hazard areas of the landscape with significant values (biological, social and economic) at risk. This assessment was followed by a prioritization plan detailing the location and timing of restoration and fuel treatments. The second initiative was an operational study of dry forest restoration treatments in 60 year old mixed stands of Douglas-fir and ponderosa pine. Four treatment strategies were tested and included: 1) thinning combined with understory burning, 2) understory burning with no thinning, 3) thinning, fuel wood removal and understory burning, 4) thinning, mechanical mulching and understory burning. The creation of a wildfire resilient stand through the restoration of dry forest structure and composition, which more closely represents the historical range of variability (Morgan and others 1994), is the primary objective in the TERP study. More specific, tactical objectives of the study were to test the short-term and long-term effects of a range of fuel treatments on stand-level wildfire resilience.

Study Area

The strategic assessment of wildfire threat was conducted at a larger landscape scale of 103,202 ha, and encompassed two Landscape Units (LUs) (Ministry of Environment, Lands, and Parks and Ministry of Forests 1999); the Birkenhead LU and the Gates LU. Each of these LUs lie on the east, or leeward, side of British Columbia's Coast Mountain Range and is comprised of a second order watershed. Climate, vegetation and stand structure details are outlined in *table 1*.

Table 1—*Study area climate, vegetation and stand structure details.*

Climate Variables	**mean**
annual precipitation	549 mm
monthly air temperature: January	-1.4 C
monthly air temperature: July	23.7 C
Vegetation Classification	
biogeoclimatic subzone[1]	Interior Douglas-fir wet warm (IDFww)
vegetation community	Douglas-fir and ponderosa pine on warm sites; predominantly Douglas-fir with minor western red cedar (*Thuja plicata* Donn ex D. Don) and paper birch (*Betula papyrifera* Marsh.) on moist sites
Stand Structure	
study area	42.3 ha total; 22.3 ha of old high-grade logging, and 20 ha of unharvested with scattered old pine and Douglas-fir
stand density: high-grade area	2,600 t ha^{-1} to 10,000 t ha^{-1}
species proportions	Douglas-fir 90 pct, ponderosa pine 10 pct
canopy closure: total unit	>90 pct

[1](Green and Klinka 1994)

Methods

The first step in the strategic assessment of wildfire threat relied upon the use of a spatial threat rating model known as the WTRS (Blackwell and others 2004). A full description of the WTRS assessment used in this project can be found in Blackwell and others (2004).

The current forest structure contains a high surface fuel load in duff, litter, and branchwood fuels, and a very high aerial fuel load. Historic surface fuels were likely 1 to 3 kg m^{-2}, while the current fuel load averages 9.8 kg m^{-2}. Canopy fuels, measured as canopy bulk density and the total weight of crown fuels, were never likely to have been high enough to sustain an active crown fire. Computing this fuel layer was achieved using the CrownMass subroutine in the Fuels Management Analyst™ (Fire Program Solutions, L.L.C. 1999). The current stand structure is highly conducive to an active crown fire, with a canopy bulk density of 1.40 kg/m^3/ha (0.028 lb/ft^3/ac) and 5.3 kg m^{-2} (23.5 t ac^{-1}) of crown fuels measured as a single layer across the entire study area. The critical canopy bulk density required to sustain a crown fire is measured at 0.1 kg/m^3/ha (0.006 lb/ft^3/ac) (Sando and Wick 1972, Harrod and others 1998). Contemporary stands are at >1400 percent of the critical threshold value. Surface fuel load must also be reduced to a point where the convective heat from a wildfire does not result in excessive crown scorch damage (Saveland and others 1990). Therefore, creating a wildfire resilient structure is the result of reducing canopy bulk density below the critical active crown fire threshold, and reducing surface fuels to the point where a wildfire will not result in unacceptable crown scorch.

Meeting the study objectives centered on developing thinning and burning prescriptions that were easily understood and implemented. Implementation of the prescribed burn post-thinning was the most constraining element in the development of treatment prescriptions. Existing surface fuel load was high, and the various thinning treatments would increase fuel load significantly. Crown scorch, even though several of the treatment units were to be thinned, was still the most critical element in developing both the thinning and burning prescriptions. The critical step in developing a crown closure-driven thinning prescription that still met the restoration objectives relied on developing a correlation between diameter at breast height (DBH) and crown area. Prior field work in Haylmore Creek yielded a strong relationship (r^2=0.70) between crown area and DBH. Field crews then established fixed radius inventory plots in the portions of TERP that were to be thinned. Variables collected included stand density, diameter distribution, tree height, and crown area. A linear regression of crown area over DBH yielded an r^2 of 0.86.

The next most critical factor in developing the prescription was setting limits on crown scorch damage and mortality. The crown scorch model in the First Order Fire Effects Model (FOFEM) (Reinhardt and others 1995) was used in combination with professional judgment to develop a maximum canopy closure parameter for burn prescription success. A target of 40 to 60 percent canopy closure was arbitrarily set by the burn planning team based on past understory burning experience. Target canopy closure was to be achieved by summing crown area by diameter, starting at the largest DBH class recorded, and adding crown area until the target was reached (*fig. 1*). The canopy closure prescription target was theoretically met at a lower diameter limit of 20 cm and resulted in a mean retention level of 270 t ha^{-1}. Based on this prescription approximately 2,600 trees ha^{-1} <23 cm DBH were to be felled.

It was assumed that, at a minimum, the thinning prescription would result in reduced canopy bulk density below the critical active crown fire threshold (i.e., <0.1 kg/m^3/ha). This would be accomplished by using canopy closure as a surrogate for the density of aerial fuels instead of a more rigorous inventory and analysis of canopy bulk density and crown weight by species and structural characteristic. The hypothesis was that if canopy closure was reduced to 40 to 60 percent from >90 percent, there would be a reciprocal reduction in canopy bulk density. Once thinning was completed to the canopy closure target, resulting stand structure could be re-compiled in FMA™ to determine post-thinning canopy bulk density and crown weights.

A series of four thinning and fuel management treatments was applied to 20 ha of the TERP study area. The thin and mulch (Unit TM) area was conducted on gentle slopes (<30 percent), where the operation could be carried out with a small tracked forwarder. In the thin and removal (Unit TR) unit the area was thinned and fuels were either manually or mechanically removed to the roadside. The thin only (Unit TO) unit was thinned only, with no fuel removal. The last unit, burn only (unit BO), contained only natural fuels and was prescribed burned unthinned. Burn objective monitoring quantified surface fuel consumption, immediate tree mortality by probable cause, crown scorch, and understory vegetation response. Prescribed burn monitoring methodology follows the U.S. Department of Interior National Park Service Fire Monitoring Handbook and included a series of $1/10^{th}$ ha plots randomly located in each treatment unit. Surface fuel inventory followed Brown and others (1982), immediate tree mortality was measured over a 100 percent sample of the study area, percent crown scorch was measured from all trees within the $1/10^{th}$ plots, and vegetation response, which is not reported here, was measured using point-intercept transects in the $1/10^{th}$ ha plots.

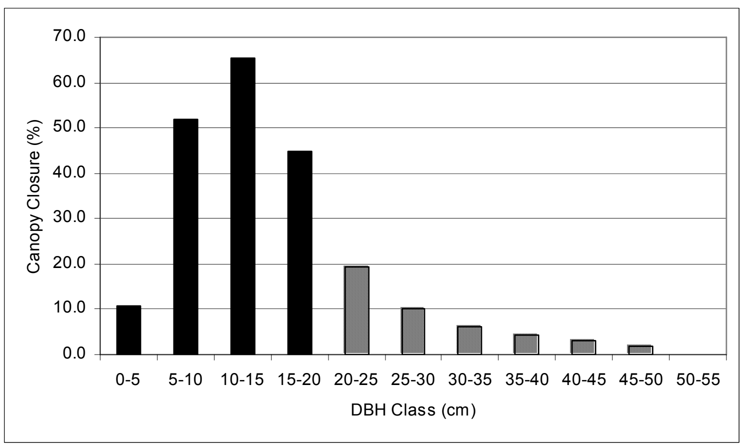

Figure 1—Range of canopy closure by 5 cm DBH class. The target, 50 percent, was reached by summing all trees starting at 45 to 50 cm and descending in DBH.

Prescribed burn operations took place in March and April of 2002, with fire effects monitoring carried out immediately post-burn. Surface fuel consumption focused on removing fine fuels <7.5 cm in order to reduce the short-term surface fire rate of spread. The ignition strategy for meeting this objective was driven by the need to limit overstory crown scorch.

Results

Manual thinning treatments carried out in units TM, TR, and TO resulted in variable crown closure by subunit. The prescription target of 40 to 60 percent canopy closure was met on average, with some gaps being larger than others (*table 2*).

Variability was attributed to the spatial pattern of tree diameters, especially in unit TO where a high density of >20 cm DBH trees are located. Along the south edge of TM where the unit abuts a steep, southwest slope the density of <20 cm DBH trees was often >5,000 t ha^{-1} with few trees >20 cm DBH; leading to larger post-thinning canopy gaps.

Table 2—*Post-thinning canopy closure by treatment subunit in the TERP study area.*

Treatment unit[1]	N[2]	Mean (pct)	SD	Minimum (pct)	Median (pct)	Maximum (pct)
TM	10	56.1	20.6	23.3	59.1	82.8
TR	16	60.1	15.3	33.5	59.1	91.3
TO	6	81.1	11.4	64.0	81.3	94.0

[1] Treatment unit abbreviations correspond to the following: TM=thin and mulch; TR=thin and remove; and, TO=thin only.
[2] Each plot involves 4 cardinal direction measurements.

The reduction in crown closure translated into a significant reduction in canopy bulk density and crown fuel load (*fig. 1*). Post-thinning canopy bulk density, measured for the entire 20 ha manual thinning area, was 0.004 lbs/ft^3/ac. The post-thinning stand also contained many significant gaps. Thinning reduced canopy fuel load by 84 percent, from 5.3 kg m^{-2} (23.5 t ac^{-1}) to 0.8 kg m^{-2} (3.8 t ac^{-1}). Unfortunately, this fuel load was added to the surface fuel load, which defeats the purpose of the prescription. However, this result was anticipated.

Fuel consumption in activity fuels was highly variable, not only in total consumption, but also in rates of consumption by fuel size class (*table 3*). The post-burn fuel inventory indicated that the highest consumption rate of the three thinned treatments occurred in the TR unit, while the only unthinned unit, BO, experienced a moderate consumption for total fuel load.

Table 3—*Fuel consumption rate by treatment unit and fuel size class.*

Treatment Unit	0 to 0.6 cm	0.6 to 2.5 cm	2.5 to 7.5 cm	<7.5 cm total	>7.5 cm sound	>7.5 cm rotten	Total
Pre-burn (kg/m²)							
Thin and mulch	1.62	0.74	1.37	3.73	3.41	0.00	7.14
Thin and removal	1.73	0.56	0.40	2.69	0.00	0.04	2.73
Thin only	1.82	0.27	0.63	2.72	6.06	0.07	8.85
Burn only	1.77	0.18	0.54	2.49	0.00	1.42	3.89
Post-burn (kg/m²)							
Thin and mulch	0.92	0.49	1.24	2.65	3.35	0.00	6.00
Thin and removal	0.18	0.18	0.27	0.63	0.00	0.00	0.63
Thin only	0.76	0.20	0.56	1.52	4.78	0.07	6.37
Burn only	0.25	0.07	0.34	0.66	0.00	1.21	1.87

The level of scorch damage in the treatment units was variable, ranging from none in the TM unit to widely scattered but low-level scorch in the TR and TO units, to more consistent but still scattered scorch in the BO unit. There are several small patches (<1 ha) of either moderate or high crown scorch, especially in the TO and BO units. From a pool of approximately 5,500 live trees in the treatment units, <200 trees were killed outright in the burn. It is expected that subsequent tree mortality will occur as Douglas-fir beetles (*D. pseudotsugae* Hopkins) and red turpentine beetles (*D. valens* LeConte) attack fire-damaged trees.

Discussion

At the stand level, the use of thinning, mulching and prescribed fire reduced fine surface fuels, and therefore the short-term threat of fire spread, but left a long-term large fuel legacy that must be dealt with through subsequent burns. The period of reduced fire spread risk is currently unknown and is dependent on the development of a fine fuel complex. Considering that these sites were depauperate prior to thinning, many of the herbs and grasses that would contribute to a future fine fuel complex will have to invade the site. The rate of invasion and accumulation of cured material is part of ongoing monitoring in the study site. Once a consistent fine surface fuel complex is developed, subsequent burns used to incrementally reduce the remaining unnatural fuel accumulation can be more easily carried out.

The TO treatment left a problematic fuels legacy as did the TM treatment. The significant difference between TM and TO treatments lies in the arrangement of fuels. Mulching in the TM treatment resulted in residual fuel lying flush with the ground, providing some opportunity for microbial and fungal decay between follow-up burns. Most, if not all, residual fuels in the TO treatment remain suspended off the ground and air dried. The breakdown of fuel will be prolonged by this effect. However, invasion of a fine fuel complex consisting of grasses and herbs is likely to occur at a rate comparable to the TM treatment unit. This could create a significant fire threat before optimum conditions present themselves for subsequent fuel reduction burns. Once conditions become favorable, burning will still need to be carefully conducted in order to limit overstory damage and minimize risk of escape.

The TR treatment provided the best results for creating a long-term, wildfire resilient stand structure that would serve as a wildland-urban interface prescription model. The post-thinning, pre-burning fuel complex was light and relatively easy to manage from a prescribed burning perspective. Surface fuel reduction and crown integrity objectives were also easy to achieve due to the low volume of pre-burn fuels. The TR treatment area is more likely to survive a wildfire occurring prior to any future maintenance burning.

The BO treatment resulted in a low post-burn fuel load, but unlike any of the other treatments this unit will likely see a substantial increase in surface fuels in the short-term due to fire-caused overstory mortality. The rate of input of these fuels varies by species and diameter (Everett and others 1999); however, within 10 to 15 yr, this treatment unit is likely to have a higher surface fuel load than it did prior to treatment. The BO unit also had the highest overall rate of crown scorch, which could lead to additional bark beetle caused overstory mortality.

Conclusions

From a strategic perspective, the treatment results indicate that options exist for creating wildfire resilient stands. The best overall option is to thin, remove the material, and then burn the site. Because one of the primary objectives in the study was the fire effects resulting from a range of stand treatments, the option of thinning and removal and no follow-up prescribed burn (a burn control unit) was not considered. While economic indicators were not tracked in detail during this study, it is abundantly clear that thinning overstocked, young Douglas-fir, forwarding the material to a road, giving it away at no cost, and then prescribed burning the site is likely to be expensive.

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A Comparison of Visual and Quantitative Changes From Rotational Prescribed Burning in Old-Growth Stands of Southwestern Ponderosa Pine[1]

Sally M. Haase[2] and Stephen S. Sackett[2]

Abstract

Two long-term prescribed fire studies were established near Flagstaff, Arizona in 1976 and 1977. One of the sites, Chimney Spring, is located on a basalt soil type and had not received any natural fires for the previous 100 years. The other site, called Limestone Flats, located on a sedimentary soil type, has a similar fire-free period but received a sanitation cut to remove the dead overstory in the early 1960s. The study was designed to test 1-, 2-, 4-, 6-, 8-, and 10-yr burning rotations in southwestern old growth ponderosa pine stands. The primary objective was to determine if a particular burning rotation would reduce and maintain the natural accumulation of fuels and reduce the stand density to a condition that would withstand a wildfire under average worst conditions. Permanent photo points were initiated at the time each study area was established, and they have been retaken periodically. Visual changes in stand structure correspond to the reduction in number of stems but don't reflect the continuation or increase in basal area per acre. The photos also show the initial reduction in large woody fuels followed by their incremental return.

Introduction

Southwestern ponderosa pine forests historically developed with the natural occurrence of frequent fire. The overstory groupings have been shown to be even- aged or even-sized. This stand structure resulted primarily from thinning by reoccurring natural fires that also stimulated the grass community and kept the accumulation of fuels in check. With the active suppression of natural and human-caused fires, the continuous accumulation of fuels and increased stand density have created a very unnatural hazardous situation in southwestern forests. Two long-term fuel reduction studies were initiated in 1976 (Chimney Spring, basalt soil type) and 1977 (Limestone Flats, sedimentary soil type) to determine if a rotational prescribed burn program would be a viable management option to reduce these hazardous conditions found in natural stands of southwestern ponderosa pine. The rotations selected reflected the fire history for the Chimney Spring area (Dieterich 1980) and included 1-, 2-, 4-, 6-, 8-, and 10-yr rotations along with a "no-burn" control.

Original fuel loadings were determined prior to the application of the burn treatments for each study area and are summarized in Sackett (1980) and Sackett and Haase (1998). Briefly, the Chimney Spring study area had a total of 22.3 tons ac^{-1} (50.0 Mg ha^{-1}) of fuel. Of this amount, 15.2 tons (34.1 Mg) were of organic material ≤1 inch (2.54 cm) in size, and the remaining 7.1 tons (15.9 Mg) was of the large woody fuel >1 inch (2.54 cm). Total fuel loading was reduced by 65 percent in the initial burn. The Limestone Flats area had a total fuel loading of 32.3 tons ac^{-1} (72.4 Mg ha^{-1}). Material less than 1 inch (2.54 cm) in diameter was 15.7 tons ac^{-1} (35.2 Mg ha^{-1}), and the larger woody fuels greater than 1 inch were 16.6 tons ac^{-1} (37.2 Mg ha^{-1}). Forty-three percent of the fuel loading was removed with the initial burn. Both areas had very similar fine fuel loadings but contrasting large fuel loadings and

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quite different consumption rates. This was most directly related to the difference in fuel moisture conditions at the time of the initial entry burns (Sackett 1980).

Various fire effects are being followed during the length of these studies. They include overstory changes in growth and mortality, understory vegetation responses, soil ammonium- and nitrate-nitrogen changes, and others. This paper was developed after evaluating the complete retake of photo points on the two study areas. The visual effectiveness of prescribed burning became obvious when comparing previously taken photos and seeing how the structure of stands can actually change with repeated prescribed fires and that this action still needs to be considered a viable management tool.

Methods

Permanent photo points were established on each of the 2.5 ac (1 ha) treatment plots at both study sites, and additional points were added when situations warranted documentation (*fig. 1*). The photo points were established from each corner looking toward the closest permanent sample point (1 to 5) and from various permanent sample points to other sample points. All trees were measured in a 0.1 ac circular plot at each of the five permanent sample points located on each treatment plot. For this paper the treatment plots that demonstrated significant visual changes were measured again in the fall of 2002, calculating the BA ac^{-1} (ft^2 ac^{-1}) of these treatment plots.

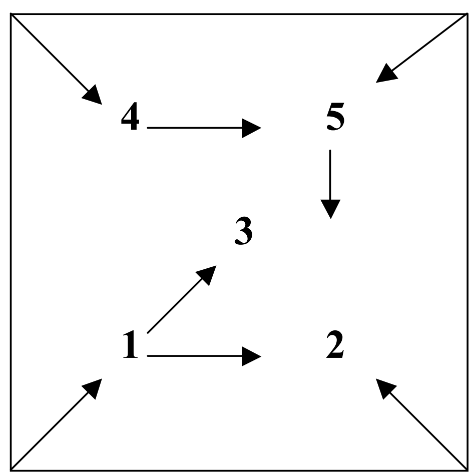

Figure 1—Diagram of possible photo point and permanent sample point orientation.

Results

The following sets of photos (*figs. 2-8*) are grouped by treatment and study site. The left-hand photo is what the photo point looked like prior to the initial burn, and the second photo is what the photo point looked like in fall 2002. Only one example is given due to the restriction of space. The stem count and basal area data are representative of the particular treatment plot represented in the photo set for that treatment and are summarized (*table 1*).

The number of times the treatments have been applied are as follows:

	Burn rotation					
	1 yr	2 yr	4 yr	6 yr	8 yr	10 yr
Chimney Spring	26	14	7	5	4	3
Limestone Flats	25	13	6	5	4	3

1 year rotation treatment

Figure 2—Top photos are taken on plot 2E1 of the Chimney Spring site, and the bottom photos are taken on plot 3A1 of the Limestone Flats research site.

2 year rotation treatment

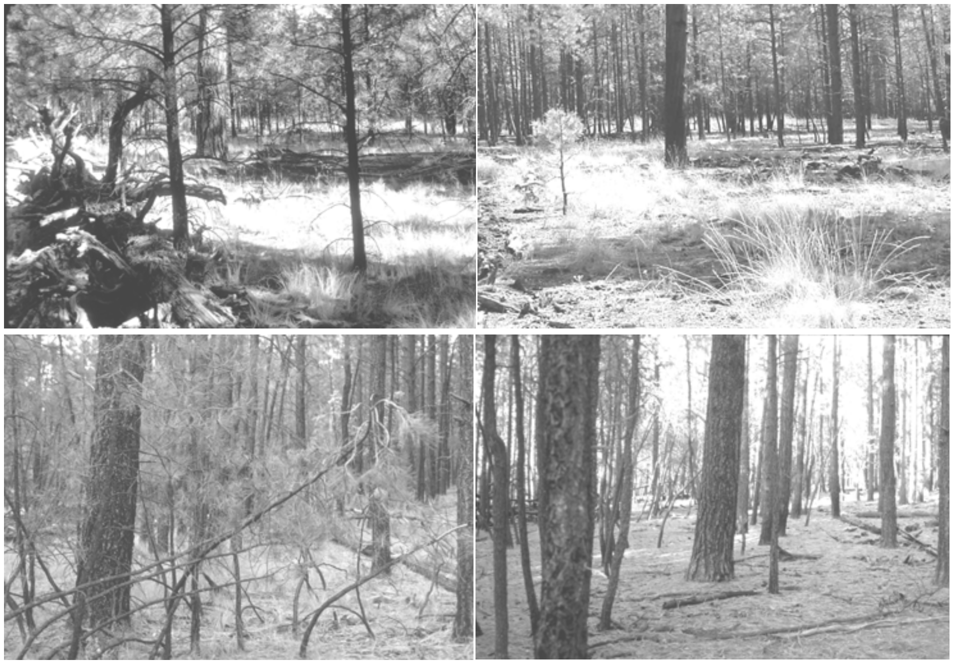

Figure 3—Top photos are taken on plot 2G2 of the Chimney Spring site and the bottom photos are taken on plot 4D2 of the Limestone Flats research site.

4 year rotation treatment

Figure 4—Top photos are of plot 1A4 of the Chimney Spring site and the bottom photos are taken on plot 4E4 of the Limestone Flats research site.

6 year rotation treatment

Figure 5—Top photos are of plot 2F6 of the Chimney Spring site and the bottom photos are taken on plot 4F6 on the Limestone Flats research site.

8 year rotation treatment

Figure 6—Top photos are taken on plot 2B8 of the Chimney Spring research site and the bottom photos are of plot 2D8 from the Limestone Flats research area.

10 year rotation treatment

Figure 7—Top photos are of plot 1B10 of the Chimney Spring research site and the bottom photos are from plot 5D10 of the Limestone Flats research site.

Controls

Figure 8—Top photos are of control plot 2CC on the Chimney Spring research site and the bottom photos are of plot 5CC of the Limestone Flats research site.

***Table 1**—Changes in number of trees per acre and basal area (BA=ft^2 acre^{-1}) of specific treatment plots of both study areas. Chimney Spring study area was initially sampled in 1976 and Limestone Flats study area in 1977. Selected plots of both areas were then sampled in 2002.*

	Chimney Spring			Limestone Flats		
Treatment sample period	**Stems ac^{-1}**	**Plot ID**	**BA ac^{-1}**	**Stems ac^{-1}**	**Plot ID**	**BA ac^{-1}**
1 year rotation		2E1			3A1	
Initial	2,808		171.8	2,132		131.7
2002	674		202.3	350		164.1
2 year rotation		2G2			4D2	
Initial	1,658		153.0	2,056		149.4
2002	664		166.1	754		186.8
4 year rotation		1A4			4E4	
Initial	1,168		160.7	2,700		152.7
2002	282		164.8	668		188.7
6 year rotation		2F6			4F6	
Initial	2,636		171.7	1,826		103.3
2002	682		181.9	314		106.7
8 year rotation		2B8			2D8	
Initial	2,334		157.8	2,298		121.9
2002	574		167.3	560		142.9
10 year rotation		1B10			5D10	
Initial	2,708		166.8	1,848		146.4
2002	282		159.0	696		172.5
Control		2CC			5CC	
Initial	2,434		175.8	2,584		119.5
2002	1,356		197.6	1,830		218.0

Discussion

The number of trees per acre and BA information (*table 1*) is based on the plots corresponding to the photos and cannot be interpreted as treatment averages, but similar visual and quantitative responses were found on the other treatment replications. Prescribed burning affects many aspects of a ponderosa pine ecosystem, but most are not as easily assessed as what these photos show (Harrington 1991, Sackett and Haase 1998). The greatest prescribed burning effect seen in the photos, both visually and quantitatively, is a decreased number of trees per acre. The reduced tree numbers appear significant for the reported treatment plots both visually and quantitatively. The smaller size classes were the most affected, as seen in the photos where the sapling thickets were greatly reduced. This reduction was achieved with subsequent applications of prescribed fire. Once the initial fuels were reduced, fire was applied more aggressively, altering the stand more significantly. Another important response that was contrary to the reduced number of trees per acre was the general increase of basal area on most of these treatment plots. This would suggest that the larger trees can continue to put on annual increments with the removal of competing smaller trees and with improved soil moisture conditions (Clary and Ffolliott 1969, Harrington 1991).

Another obvious result was the removal of large woody material. Fire managers have had to work with the requirement made by wildlife managers that to conserve this fuel size class for small mammals and other users. As the prescribed fire thins these stands, the amount removed is restored with newer, sounder material. Although not easily seen in the photo sets shown, large woody material is being added to the system on a regular basis. A portion of mature yellow pines were killed by the initial prescribed burns because of the heavy fuel loadings around the bases of these trees, and they will all eventually fall, adding to the coarse woody debris component of the ecosystem. Tops of pole size and larger trees are often struck by lightning and, as seen in the photos of the controls (*fig. 8*), demonstrate that these large fuels are being added to the system and not being removed. Also seen in these same photos is the breakdown of the large woody component overtime, but even in this condition the fuel still remains a concern for the fire manager.

Stand structural changes are evident in these photos that support the use of prescribed burning, if the objective is to reduce natural fuel loadings and remove the small thickets typically found in stands of southwestern ponderosa pine represented here. Crown base heights are also raised, as seen in the photos, reducing the stands' crown fire potential. This is achieved by removing the lower branches of the pole size trees through scorching and the removal of ladder fuels. It is evident that prescribed burning alone is not sufficient to return existing uncut stands into stands that would significantly reduce the probability of uncharacteristic catastrophic fires. Mechanical thinning either before or after the removal of natural fuels by prescribed burning may be needed to address the issue of making stands safe from catastrophic wildfire in a more timely manner.

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Survivorship of Raked and Unraked Trees Through Prescribed Fires in Conifer Forests in Northeastern California[1]

William F. Laudenslayer, Jr.,[2] George N. Steger,[2] and Jonathan Arnold[3]

Abstract
Large diameter, old trees are an important component of functioning forests, as they provide habitat for many wildlife species and add value to the scenery along roads and trails that cross our National Forests and Parks. Tree mortality, from prescribed or wild fire, is of great concern to forests managers, especially mortality of those of large diameter. Raking away the litter and duff around the base of these desired trees has been advocated as a way to reduce tree mortality during prescribed burns. We tested the effectiveness of raking to reduce tree mortality in two burn areas in Lassen Volcanic National Park (Lake and Lost Creek) by randomly selecting large and small diameter trees and removing the litter and duff to bare mineral soil for 1 m in radius around the base of each sample tree. The areas were prescribed burned in the fall of 1997. Results varied considerably between study plots, with 37 percent of all the sample trees (raked and unraked) in the Lake burn dying, in contrast with the Lost Creek burn where 25 percent of the sample trees died. Raked trees, however, did survive better than unraked trees in both study areas. Twenty-seven percent of the raked trees died in the Lake burn in contrast with 47 percent of the unraked trees. In the Lost Creek burn, none of the raked trees died whereas 50 percent of the unraked trees died.

Introduction

Reintroducing fire into western forests, whether by permitting naturally ignited fires to burn or applying prescribed fire, is widely being employed to reduce fuel loadings and to restore fire as a process to forests where fire historically has had a profound influence. However, because of the absence of fire, fuel accumulation may increase fire severity in prescribed burns, causing unacceptable mortality in large trees. As an example, following a prescribed burn in an eastside pine forest comprised mostly of Jeffrey pine (*Pinus jeffreyi*), all 14 of the large, old Jeffrey pines in a study plot died, apparently from effects of the fire, within 3 yr of the fire (Laudenslayer 2002). The crowns of these trees did not appear to be scorched severely enough to have caused death, but the fire appears to have been intense and fire-elevated temperatures prolonged at the soil surface where litter and duff accumulations were as deep as 30 cm.

The literature reports a number of methods that might be effective in reducing large tree mortality during and following prescribed fire (Weatherspoon and others 1989, Fulé and others 2002a, Jerman and others 2004). One of these methods is to

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physically remove the heavy deposits of litter and duff from around the bases of large, old trees that have not seen fire in decades. We, in cooperation with Lassen Volcanic National Park, decided to test this method on two upcoming prescribed burns. Our objective was to determine if removal of litter and duff down to bare mineral soil in the fall can improve survivorship of both large and smaller diameter trees following a prescribed fire.

Study Areas

The prescribed burn areas are located in the southern extremity of the Cascade Range in California. Both burn areas are within Lassen Volcanic National Park. The Lake burn is just northeast of Butte Lake, and the Lost Creek burn is adjacent and to the south of the Crags Group campground, approximately 8 km east of Manzanita Lake (*fig. 1*).

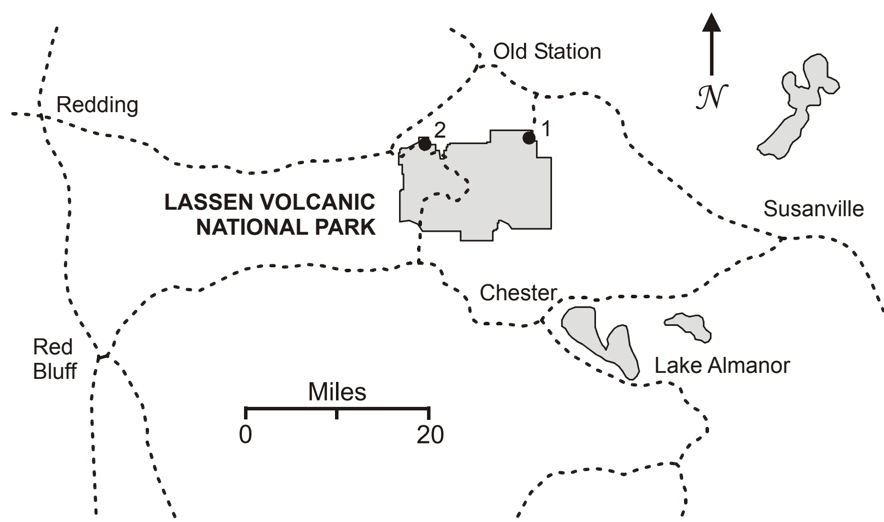

Figure 1–Vicinity of Lassen Volcanic National Park in Northeastern California; locations of the burns are at the closed circles: 1 is the Lake burn and 2 is the Lost Creek burn.

The Lake burn area is dominated by an eastside pine forest comprised primarily of Jeffrey pine with smaller numbers of ponderosa pine (*P. ponderosa*), white fir (*Abies concolor*), and red fir (*A. magnifica*). The stands are very open and dominated by relatively few, sparsely distributed, large diameter trees, with very few seedlings, saplings or smaller trees. Shrubs are rare and herbaceous growth is uncommon. Topography is variable, ranging from nearly level to very steep. Elevation generally increases from the periphery of the burn toward several high points. The soil surface is dominated by a layer of volcanic ash (approximately 25 to 35 cm deep), which resulted from an eruption of a nearby cinder cone sometime between A. D. 1630 and 1670 (Clynne and others 2000). Frequency of rocky outcroppings of older volcanic rocks increases as elevation increases. Prior to the burn, deposition of needles and other organic debris was generally restricted to within a few meters of the boles of large diameter trees and areas between trees were relatively free of dead organic material.

The Lost Creek burn area is dominated by mixed conifer forest, including white fir, ponderosa, sugar pine (*P. lambertiana*), and incense cedar (*Calocedrus decurrens*). Canopy closure is much higher than in the Lake burn area, and trees ranging from saplings to very large are present. Shrubs are rare and herbaceous growth is uncommon. Topography is variable, but sample trees were all on rather gentle slopes (<10%). The soil surface, before the burn, was dominated by a continuous layer of needles and other organic debris with deposits accumulating around the bases of the large diameter trees.

Methods

In September 1996, we randomly selected pine trees (Jeffrey, ponderosa, or sugar) for the experiment in the Lake burn and Lost Creek burn areas to be treated by prescribed fire. Trees were selected by choosing a random direction and distance from an initial start point, and then that process was repeated from selected tree to selected tree. On occasion, a direction/distance for a tree would place that tree outside the proposed burn unit, and for these trees a new random direction and distance were selected. When arriving at the location of the next tree, a random number generator determined if the tree was to be of large diameter or small and whether or not it was to be raked. Once a tree was selected, it was tagged with a uniquely numbered stainless steel label and flagged so it could be readily relocated. The raked trees had all of the litter-duff within 1 m radius of the tree removed to bare mineral soil using pitchforks and McClouds. The litter-duff was then scattered beyond the duff mound but generally within the dripline.

The sample size for the Lake burn was 30 trees: 20 large trees (>61 cm in diameter at breast height [dbh]) and 10 small trees (<61 cm dbh); the sample size for the Lost Creek burn was 20 trees: 10 large and 10 small. Half of each of the four groups of trees was raked and half was left with the litter and duff intact.

Prior to the burns, information collected on each tree included tree species, tree dbh, tree risk (a visual evaluation of the "health" of each subject tree using criteria developed by George T. Ferrell from Dunning [1928], Keen [1936], and Salman and Bongberg [1942]) [*table 1*]), description of fire scars, and bark thickness at the litter-duff layer taken at 4 points 90^{o} apart with two coinciding with litter-duff depth measurement points. Information about the fuels associated with sample trees included subjective characteristics of the litter-duff and other adjacent fuels (especially the presence of downed wood >12 cm in diameter within 3 m of the tree) and litter-duff depth on the upslope side of the tree and on the downslope side of the tree both at the bole and 1 m away from the bole (for the raked trees, these measurements were taken after raking whereas duff pins were set for the unraked trees so that depths could be measured after the fire). Information on the topographic context of each sample tree included an estimate of the slope of the landscape to the nearest 5 degrees and an estimate of the aspect in 45^{o} classes (0, 45, 90, 135, 180…).

Table 1–*Risk Rating Criteria summarized from George T. Ferrell (pers. com) afterDunning (1928), Keen (1936), and Salman and Bongberg (1942).*

Risk rating	Characteristics of risk
1-Low	Crowns: full foliage, healthy appearing; no weakened parts Foliage: thick and continuous; most twigs with normal foliage complement Needles: long and dark green in color
2-Moderate	Crowns: incomplete in spots, moderately healthy Foliage: some areas thin or bunchy but not concentrated Needles: average length or better; color fair to good
3-High	Crowns: somewhat ragged or thin Foliage: in parts of crown thin or bunchy; some to many twigs fading or dead Needle: length average to shorter than average; color fair to poor
4-Very High	Crowns: in poor condition, ragged or thin Foliage: thin or bunchy; twigs/branches dead or dying; weakening crown parts Needles: short or sparse, color poor

Immediately after the burns, data collected from each tree included scorch height on tree trunk at both upslope and downslope points, insect activity less than 3.5 m high on the bole (where *Dendroctonus valens* activity is generally found) and at greater than 3.5 m where other bark beetle activity is generally found, tree risk, and any noticeable change in fire scars. Bark beetle activity was categorized as clear or red in color and placed into 1 of 4 categories: none, <5 pitch tubes, from 6 to 20 pitch tubes, and >20 pitch tubes in the respective height band. The change in the depth of litter-duff layer was measured from the duff pins, and degree of litter-duff consumption was determined.

In the years after the fires (1997 through 2002), each tree was examined yearly. Each tree was classified as alive or dead and each tree was rated for risk. Bark beetle activity, by each of the four categories described above, below and above 3.5 m, as well as bird foraging activity were noted for each tree.

Prescribed fire was applied to both study areas in the fall of 1996. The Lake burn was ignited late in the afternoon on 28 September 1996 and continued into 29 September until the entire unit was burned. Firing was by spot ignition. Ignition was generally started at the higher elevations and allowed to back down the slope with 1-2 ft. flame lengths. Torching was common in small trees and occasionally occurred in large trees. Winds were relatively light and supported the ignition tactics. Mature ponderosa pines in the burn area, including both raked and unraked large trees, were ring-fired and fire behavior at 5 study trees was observed (Rankin 1996a) (*table 2*).

The Lost Creek burn (identified as the Crags Management Ignited Prescribed Fire by Lassen Volcanic National Park) was ignited late in the afternoon of 1 October 1996 and continued into 2 October. Firing was a combination of spot and strip ignitions. Ignitions proceeded generally from higher to lower elevations, resulting in a backing fire with flame lengths of 1 to 2 feet. Torching, common in small trees and occasional in large trees, resulted from jackpots of heavy fuels and frequent fuel ladders. Winds were relatively light and supported the ignition tactics. Mature ponderosa pines in the burn area were ring-fired, and fire behavior at 5 study trees was observed (Rankin 1996b) (*table 2*).

Table 2–*Weather and fire characteristics (from Rankin 1996a and 1996b).*

Ignition		**Relative Humidity**	**Temp**	**Wind**		**Fuel moisture (pct)**		**Rate of Spread (chains/hr)**		**Flame length**
Date-Time	Method	pct	°F	Dir.	Speed (mph)	Log	Duff	Backing fire	Head fire	ft
Lake burn										
Sep 28-29, 1996-17:40	spot & ring	24-49	50s-70s	NE to SE	3-5	not recorded		1/2	3-4	1-3
Lost Creek burn										
Oct 1-2, 1996-15:55	spot & strip	30-53	50s-60s	SE to SW	0-4	15	29	1/4	>1	0.5-2

Results

Sample trees in both burn areas were primarily Jeffrey pines; only the Lost Creek burn included sugar pines (*table 3*). Risk ratings increased in both burns following the fires, especially in the trees that eventually died. Scorch heights were generally greater for the unraked trees, although the highest scorch heights were recorded for the raked trees in the Lake burn. Both scorch heights and bark beetle activity were greater in the trees that died. Bark thickness did not vary appreciably before and after fire.

Twelve (40 percent) of our sample trees within the Lake burn died within 2 to 3 years after the prescribed burn, and more unraked trees (8) died than raked (4). Mortality was greater among the larger than the smaller diameter trees, especially in the unraked group (*fig. 2*). In contrast, 5 (25 percent) of our sample trees within the Lost Creek burn died in the same time period after the prescribed burn, and all mortality was among the unraked trees (*fig. 2*).

Amounts of litter and duff close to the sample trees prior to raking and burning was similar for members of each size class; the average litter depth for trees <61 cm dbh was substantially less than for trees >61 cm dbh. The amount of litter and duff associated with the unraked dead trees was greater than that associated with the unraked live trees (*fig. 3*). All trees that died in the Lost Creek burn were in close proximity (within a meter or two) to a relatively large log or snag, making it unclear how much the litter and duff or the logs and snags contributed to the mortality of the trees.

Discussion

In the Lake burn area, tree mortality was twice as likely to occur on the unraked trees as the raked; although mortality did occur in the raked group, the larger diameter trees were more likely to die than smaller diameter trees. Heavy deposits of litter, duff, and larger dead material were associated with the majority of the mortality events. Some surviving unraked trees were also associated with relatively thick deposits of litter and duff, albeit substantially less than the trees that died. However, topography and subtle shifts in fire behavior, related to winds and fuel loadings, and the layer of volcanic ash likely played a role in tree mortality (Fulé and others 2002).

Table 3–*Tree characteristics before and after the prescribed fires.*

					Diameter (cm)		Before fire	After fire				
							Risk rating median	Risk rating median	Scorch height (m)[3]		No. pitch tubes[2] (range)	
	n=	Pije[1]	Pipo	Pila	Mean	St. dev.			Mean	St dev	<3.5 m	>3.5 m
Lake burn												
Dead Trees												
Unraked Lg Trees	6	1	5	0	108.5	16.45	2	2.5	8.2	8.57	0-H	0-H
Raked Lg Trees	2	0	2	0	100.2	1.06	3.5	4	12.0	4.24	M-H	L-M
Unraked Sm Trees	2	1	1	0	30.5	14.85	2	4	4.0	0.71	0-L	0-L
Raked Sm Trees	2	2	0	0	29.0	1.41	2	Dead[4]	0	-	L-H	L-H
Live Trees												
Unraked Lg Trees	4	3	1	0	79.2	15.31	2	3	5.5	5.17	0-H	0-M
Raked Lg Trees	8	4	4	0	105.3	23.17	2	2	1.4	1.71	0-H	0-H
Unraked Sm Trees	3	3	0	0	33.8	6.33	2	3	3.2	1.04	L-H	0-M
Raked Sm Trees	3	3	0	0	34.5	13.48	1	2	1.0	1.73	0-M	0-L
Lost Creek burn												
Dead Trees												
Unraked Lg Trees	3	1	2	0	118.5	32.97	1	1	2.1	2.24	H	L-M
Raked Lg Trees	0	0	0	0	-	-	-	-	-	-	-	-
Unraked Sm Trees	2	2	0	0	36.5	2.12	2	3	2.8	2.40	L-H	0-M
Raked Sm Trees	0	0	0	0	-		-	-	-	-	-	
Live Trees												
Unraked Lg Trees	2	0	2	0	113.2	16.62	1	1	3.2	1.61	M	0-L
Raked Lg Trees	5	1	3	1	108.5	21.04	2	2	0	-	0-L	0
Unraked Sm Trees	3	1	0	2	45.5	17.39	1	1	5.2	8.93	0	0-M
Raked Sm Trees	5	3	1	1	30.4	9.67	1	2	0.01	0.02	0-L	0

[1] Pije=Jeffrey pine; Pipo=ponderosa pine; Pila=sugar pine.
[2] 0=no pitch tubes detected; L= <5 pitch tubes counted; M=6-20 pitch tubes; H>20 pitch tubes.
[3]total scorch height measured from the ground; generally scorched bark but may include scorch in crown if trees are short enough
[4]both trees were dead on first visit following the prescribed burn.

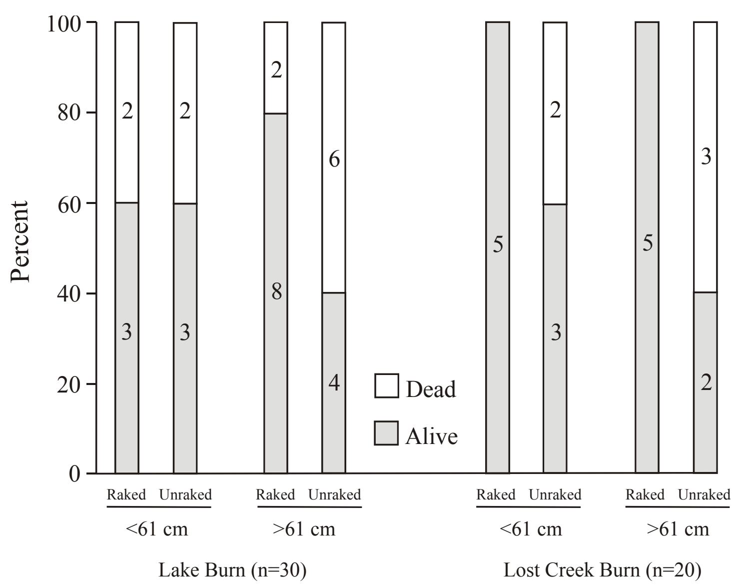

Figure 2–Tree mortality of sample trees in the Lake burn (Eastside burn) and Lost Creek burn (Westside) by tree diameter class and whether raked to bare mineral soil or left unraked.

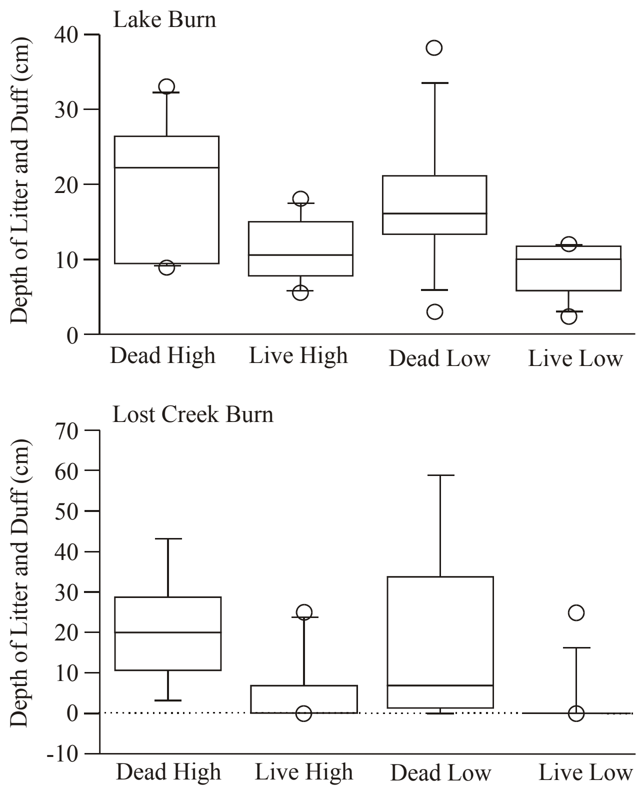

Figure 3–Distribution of litter and duff of the unraked trees in the Lake (upper figure) and Lost Creek (lower figure) burns contrasting the amounts associated with the trees that died with that associated with the trees that survived on both the uphill (High) and downhill (Low) sides of the trees.

Within the Lost Creek burn, tree mortality only occurred in the unraked trees, and all of the trees that died were closely associated with a relatively large log or snag that contributed to the fire's intensity close to the experimental tree. In addition, the unraked trees that died had substantially thicker deposits of litter and duff than the unraked trees that survived–range of 0 to 59 cm and 0 to 25 cm, respectively.

Removal of the litter and duff from around the bases of trees prior to initiating a prescribed burn may be beneficial in preventing undesirable effects such as tree girdling and bole scorching, thus benefiting tree survival (Sackett and others 1996, Kolb and others 2001, Jerman and others 2004) or possibly to have no effect or increase tree mortality through damage to the fine rootlets (Fulé and others 2002b). Our work suggests that removal of litter and duff from an area around the bole of a tree, in both eastside and westside burns, may reduce mortality that is related to burning in areas where fire has been excluded for many years.

Timing of litter and duff removal may also be a factor in tree response. In this study, we removed the litter and duff in the fall, when soils were dry, the trees inactive, and when the trees may have been less susceptible to rootlet damage. In our initial experimental design we had prepared three sites for the raked vs. unraked tree experiment. Trees within the third area, pines as well as firs, were raked in the fall in the same manner as the two experimentally burned plots. This third plot was not burned in the fall of 1997 because the weather conditions did not meet the fuels prescription and was not burned until the spring of 1999. To date, we have not observed any mortality of any of the sample trees whether raked or not. Whether these results are related to the timing of the burn or some other factor is not known but it suggest that further experimentation on the season of burning and raking, size of area raked and tree health are needed.

While our results indicate that litter and duff removal may improve survival of selected trees during prescribed burning, our design may not have been sufficient to permit very strong conclusions. Fire intensity and the resulting severity varies greatly relative to variables such as the distribution of fuels, weather conditions during the burn, topographic variation, and characteristics of surface soils at the site. Pairing of sample trees, with a raked and unraked tree within close proximity of each other, may help filter out some of these variables that could influence survival.

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We greatly appreciate the contributions of Jim Baldwin, Ken Castro, George Ferrell, Pat Shea, Carl Skinner, Nicole Tancreto, and Phil Weatherspoon to the design, execution, and data collection for this study. We also thank Ken Castro and the Lassen Volcanic National Park fire personnel for conducting the prescribed burns and the Park administrators for cooperating with this project. Finally we thank Sally Haase and Marcia Narog for their helpful comments on the draft manuscript.

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The Effects of Prescribed Fire and Ungulate Herbivory 6 and 7 Years Postburn in the Upland Bitterbrush (*Purshia tridentata*) Communities of Rocky Mountain National Park, Colorado[1]

Kelly Nesvacil[2] and Charles E. Olmsted[3]

Abstract
A controlled, manipulative study utilizing exclosures was initiated in 1994 by Zeigenfuss and others (2002) to assess the effects of prescribed fire and ungulate herbivory in the bitterbrush communities of Rocky Mountain National Park (RMNP). Four sites were chosen randomly from all available bitterbrush communities in RMNP wherein exclosures were erected (March/May 1995) and paired grazed plots established. Prescribed burns were performed on half of each exclosed and grazed area in 1995 and 1996. Using the sites and exclosures already present in the bitterbrush communities, additional data were collected from July 2001 to August 2002 to determine the communities' responses to prescribed fire and herbivory 6 and 7 yr post-burn. Bitterbrush canopy volume and estimated annual production remain lower 6 and 7 yr post-burn in burned treatments as compared to their corresponding unburned treatments. Total shrub canopy area, volume, and estimated annual production did not significantly differ due to burning, but differed due to grazing at least 1 yr of the study (p <0.10). Data support that the ambient level of herbivory present in RMNP is affecting post-burn successional patterns by impeding shrub regeneration. We caution against the use of any type of prescribed fire that would burn a substantial portion of the shrub component in these communities while ungulate herbivory remains high due to the alteration of post-fire successional patterns that could result in the loss of a major component of the community.

Introduction
The effects of prescribed fire on the plant communities within Rocky Mountain National Park (RMNP) are not well understood (NPS 1992). Because prescribed fire is utilized as a management tool within RMNP, it is important not only to understand the effects of prescribed fire on the plant species within the Park, but what impact the resulting changes in vegetation may have on the wildlife within the Park. The upland bitterbrush communities within Rocky Mountain National Park comprise 3.3 km^2 of the winter range for ungulates on the Park's east side and are characterized by antelope bitterbrush (*Purshia tridentata*), ponderosa pine (*Pinus ponderosa*), rabbitbrush (*Chrysothamnus viscidiflorus*), mountain muhly (*Muhlenbergia montana*), and needle-and-thread grass (*Hesperostipa comata*). Despite their small area, bitterbrush communities are important habitat for many species of wildlife found in the Park, such as mule deer (*Odocoileus hemionus*) and the green-tailed towhee (*Pipilo chlorurus*). Thus, proper management of bitterbrush communities is

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[1]An earlier version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.
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essential not only to conserve community biodiversity within RMNP, but also for conservation of wildlife populations that depend on the communities.

Historically, bitterbrush communities were characterized by patchy, low-intensity surface fires (NPS 1992, Pyne and others 1996). There is much variation in the literature regarding the fire frequency for bitterbrush communities, but the RMNP Fire Management Plan cited a fire frequency of 30 yr (NPS 1992). According to the 1992 Fire Management Plan for RMNP, bitterbrush communities fall into the fire suppression zone where prescribed fire is utilized. The type of prescribed fire used in these communities involves a majority of the shrub canopy during burning and does not mimic the type of fire thought to occur historically.

Due to the fact that bitterbrush and other plant species in these communities may have limited tolerance to fire, RMNP managers need to assess the use of widespread prescribed burns as a valid management tool in the upland bitterbrush communities. As stated previously, bitterbrush communities are utilized by many animal species within the Park. For example, bitterbrush is a main source of winter forage for mule deer in RMNP, accounting for approximately 24 percent of mule deer winter diets, with rabbitbrush accounting for 13 percent and wax currant (*Ribes cereum*) 7 percent (Stevens 1980). Therefore, Park managers must take into account all effects prescribed fire may have on the productivity and physiognomy of the plant community and in turn what these effects will have on animals in the Park.

Confounding the use of prescribed fire in RMNP is an increase in Park elk numbers. The elk (*Cervus elaphus*) population wintering in RMNP is estimated at 1,069 individuals, with an additional 1,975 individuals outside RMNP in the Estes Valley that utilize park winter ranges during spring and fall migrations (Lubow and others 2002). The elk population in RMNP is estimated to have increased nearly three-fold since being released from artificial controls within the Park in 1968, and heavy herbivory and plant damage to winter range vegetation has been documented (Olmsted 1997, Zeigenfuss and others 1999). However, the mule deer population within the Park has decreased from a count of 1,021 mule deer in 1937 to 1938 to approximately 200 individuals in RMNP and 300 individuals in the surrounding Estes Valley (Conner and others 2002, Stevens 1980). Herbivory at moderate to heavy levels (>50 percent herbaceous offtake), as occurs on the winter range of RMNP, could alter successional patterns after prescribed burning by decreasing shrub regeneration, resulting in an additive effect between prescribed fire and herbivory.

Plant communities respond variably to the effects of fire and herbivory over time. Long-term data are needed to ascertain the effects of fire and herbivory on a specific community and then make effective management decisions for the entire community. If long-term data can be collected on the effects of fire and herbivory in upland bitterbrush sites, stronger inference of the effects on plant species productivity and community structure will be possible. This study collected 2 yr of post-treatment data that assessed the condition of the vegetation 6 and 7 yr post-burn and provides Park managers with a more reliable representation of the impacts of prescribed fire and herbivory on the upland bitterbrush communities.

Materials and Methods

To assess the impacts of prescribed fire and herbivory on Rocky Mountain National Park's upland bitterbrush communities a controlled, manipulative study began in 1994. Four sites were chosen randomly from all available bitterbrush communities in the Park using GIS (Zeigenfuss and others 2002). The site locations lie in the areas of Aspenglen Campground, Deer Ridge, Hollowell Park, and Beaver Meadows Entrance Station (*table 1*). In March-May of 1995 researchers set up 30.5 × 45.7 m exclosures on the four sites with paired plots that were allowed to be grazed (Zeigenfuss and others 2002). A prescribed burn was done on half of the exclosed and paired plots at each site in late fall 1995 or early April 1996, resulting in the treatment combinations of grazed-burned, grazed-unburned, exclosed-burned, and exclosed unburned.

The previous researchers collected 1 to 2 yr pre-burn data and 2 years post-burn vegetative data. We collected an additional 2 yr post-burn data, 6 and 7 yr post-burn in the summers of 2001 and 2002 on the same parameters as the previous researchers. One year of winter offtake data was collected in the spring of 2002. Our methodology follows that of the previous researchers (Zeigenfuss and others 2002).

Table 1—*Site descriptions for the locations of the four study sites in RMNP, Colorado.*

	Aspenglen Campground	Hollowell Park	Beaver Meadows Entrance	Deer Ridge
Elevation (m)	2558	2625	2544	2648
Ave. slope[1] (^{o})	10	22	8	14
Aspect (^{o} from N)	131	162	170	166
Soil type[2]	Rofork-Chasmfalls complex	Isolation gravelly sandy loam	Rofork-Chasmfalls complex	Rofork-Chasmfalls complex

[1]Slope taken at the middle of each treatment then averaged.
[2] Unpublished data on file at Rocky Mountain National Park, Estes Park, Colorado.

Measurement of Shrub Parameters

Shrub data were collected from three randomly placed 9.3 m^2 circular plots in each treatment. Data on species-specific estimated annual shrub production (kg ha^{-1}), canopy area (m^2 ha^{-1}), and canopy volume (m^3 ha^{-1}) were collected in July/August of 2001 and 2002 for all treatments at each site. Estimated annual shrub production data were collected and calculated following Peek (1970). Log-log regression equations were developed for each shrub species each year using site, treatment, and canopy volume as predictors of production. Adjusted R^2 values ranged from 0.41 to 0.97 in 2001 and from 0.54 to 0.87 in 2002. In addition to shrub plot data, three bitterbrush plants per treatment per site were tagged from the previous research. These tagged plants had all data collected all years.

Measurements of shrub offtake by ungulates from the previous winter were collected in May 2002 from the shrub plots in the grazed treatments only. Percent leader use was calculated by dividing the number of browsed leaders by the total number of leaders and multiplying by 100. Percent twig winter utilization was calculated following Jensen and Urness (1981). Total consumption was estimated by multiplying percent leader use by percent twig utilization.

Measurement of Herbaceous Parameters

Data for annual herbaceous biomass were collected in July/August 2001 and 2002 for all treatments at each site. Three randomly located 0.25 m^2 circular plots per exclosed treatment were used for sampling. To ascertain annual herbaceous biomass on grazed treatments, three 1 m^2 moveable grazing cages were randomly placed in each grazed treatments in April, and samples were collected using the same 0.25 m^2 circular plots from under the grazing cages in July/August. All herbaceous biomass within the circular plots was clipped 5 cm above ground level and sorted by species. The litter, graminoids, and forbs were dried at 55°C for 48 hr and weighed. Percent cover data were collected from the same 0.25 m^2 circular plots used to measure annual herbaceous biomass.

Herbaceous overwinter offtake by ungulates was measured in late April 2002 in grazed treatments. Grazing cages were randomly relocated in August after annual herbaceous biomass collection. The procedure follows that of collection for annual herbaceous biomass in grazed treatments with 3 additional measurements per treatment taken outside the grazing cages. Clippings were sorted by vegetation type (forb, graminoid, litter), and percent differences in dry weight between grazed and caged locations were determined to measure winter consumption by ungulates.

Statistical Analyses

The data were analyzed for significant differences between treatments in SAS Statistical Software Package, version 8 (alpha=0.10). Voucher specimens were collected and archived at the Rocky Mountain National Park Working Herbarium (ROMO). All plant identifications were made using Weber and Wittmann (2001).

Results and Discussion

Overwinter herbaceous offtake for 2001 to 2002 was 64 percent, with 21 percent total shrub consumption and 49 percent leader use. A significant negative effect due to grazing on total estimated annual shrub production was detected in 2002 (p=0.07), with no effects detected due to burning both years. From 2001 to 2002, both exclosed treatments increased estimated annual shrub production, while both grazed treatment decreased. Also, the grazed-burned treatment had the lowest estimated production value both years. Tagged plant production in unburned treatments was twice as great as their corresponding burned treatments in 2001 and four times as great in 2002 (*fig. 1*, p=0.004; p=0.01). These data suggest a long-term negative effect due to prescribed burning on individual bitterbrush plants. Tagged plant data reveal more about individual plant responses to treatments, because shrub plot data are confounded by dispersion of shrubs within and among sites. Precipitation in 2002 was five times less than 2001 from May to September, as measured at RMNP.

Grazing decreased total shrub canopy volume in 2001 and 2002 (p=0.06; p=0.04) with no effect detected due to burning. Moreover, the exclosed-burned treatment averaged 25 percent lower canopy volume values than the exclosed-unburned treatment in 2001 and 2002, while the grazed-burned treatment averaged 50 percent lower values than the grazed-unburned treatment both years (*fig. 2*). Grazing decreased total shrub canopy area in 2002 (p=0.04) with no effects in 2001 detected due to grazing or burning. However, the grazed-burned treatment had the lowest value for both parameters both years and burned treatments had lower values than their corresponding unburned treatments both years (*fig. 2*).

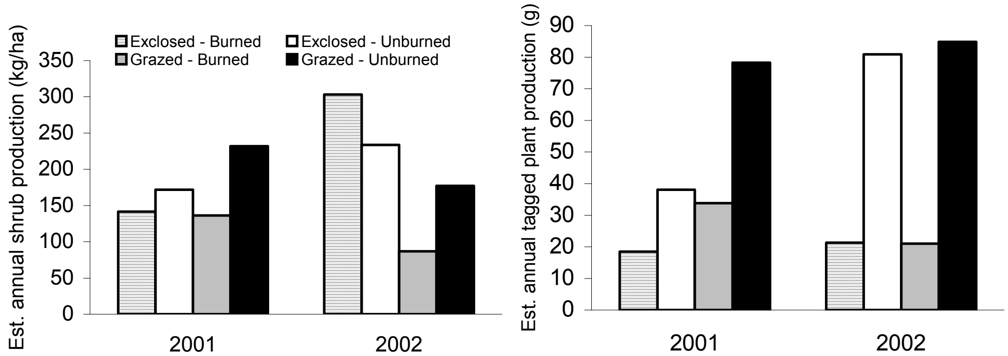

Figure 1—Estimated annual production for total shrub (kg/ha) and tagged plant (g) parameters 6 and 7 years post-burn.

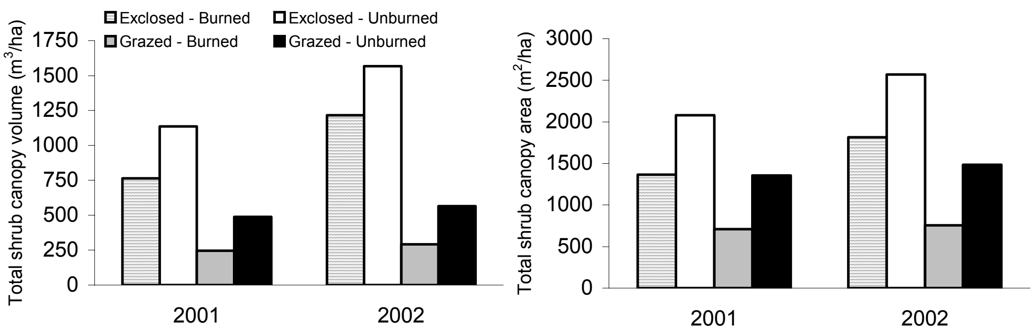

Figure 2—Total shrub canopy volume (m^3/ha) and canopy area (m^2/ha) 6 and 7 years post-burn.

Tagged plant canopy area remained decreased due to burning 6 and 7 yr post-burn (p=0.02; p=0.004) as did tagged plant canopy volume (p=0.01; p=0.03). Unburned treatments had two to four times as much tagged plant canopy area as their corresponding burned treatments and three to seven times the tagged plant canopy volume as their corresponding burned treatments. No effects on either parameter due to grazing were detected. Burning decreased shrub plot bitterbrush canopy volume in 2001 (p=0.08) and a significant effect due to grazing was detected in 2002 (p=0.04). Again, the grazed-burned treatment had the lowest treatment value both years. Burned treatments averaged 40 to 60 percent less canopy volume than unburned treatments over both years (*fig. 3*). Seven years post-burn, 54 to 61 percent of the bitterbrush plants sampled were resprouting in burned treatments. However, the resprouting plants in the grazed treatments were prostrate, with a highly pruned appearance and small canopy area and volume. This suggests that bitterbrush can resprout adequately after burning, but the ambient herbivory levels present in the Park are impeding regeneration. However, unburned treatments still have greater bitterbrush canopy volume values 6 and 7 yr post-burn than their corresponding burned treatments.

No significant effects due to burning were found for total annual herbaceous biomass or percent cover 6 and 7 yr post-burn (p>0.10). Annual herbaceous biomass ranged from approximately 30 to 50 g/m^2 in each of the treatments, with no trends present. In 2001, grazing decreased the percent cover of litter and moss and increased the percent cover of bare ground and basal graminoid cover (p<0.10).

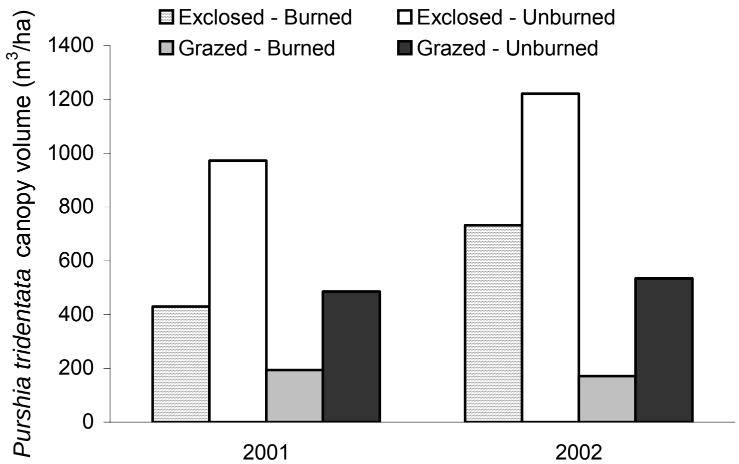

Figure 3—Shrub plot bitterbrush canopy volume (m^3/ha) 6 and 7 years post-burn.

Conclusions

The grazed-burned treatment had the lowest values for total shrub canopy area and volume, estimated annual shrub production, and bitterbrush canopy volume. This suggests that ambient herbivory levels are enough to affect post-fire successional patterns, specifically by inhibiting shrub regeneration. Due to the long-term negative effect of prescribed fire and ungulate herbivory on shrub canopy area and volume, we would not recommend the use of prescribed fire in these communities; the integrity of community structure and function can not be maintained under both influences. RMNP is considering using point-source burns in these communities that would better mimic the type of fire thought to occur historically. However, we caution against the use of any type of prescribed fire that would burn a substantial portion of the shrub component in these communities while ungulate levels are elevated. The additive effect between prescribed fire and ungulate herbivory causes the loss of essential components in the bitterbrush communities. This loss not only results in the degradation of habitat for mule deer and numerous other plant and animal species, but also a degradation of community biodiversity within the Park.

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Aerial Mulching Techniques—Trough Fire[1]

Robert Faust[2]

Abstract

The Trough fire occurred in August 2001 on the Mendocino National Forest of northern California. A burned area emergency rehabilitation team evaluated the fire effects on the watershed. Concerns were soil from the denuded slopes moving into streams affecting fishery values, reservoir sedimentation and storm runoff plugging culverts leading to road wash outs. Past evaluations have shown hand straw mulching to be a superior method in protecting soil from raindrop impact, surface runoff and erosion. However, hand mulching is a long and expensive process with safety factors dictating mulching only on ridges and not steep stream banks. On the Trough fire two aerial mulching techniques were evaluated on stream banks. Both evaluations were done in cooperation with USDA, Forest Service, San Dimas Technological and Development Center. One technique was in conjunction with the California Straw Supply Co-op. Work consisted of straw baling techniques and dispersal of straw by helicopter. Field testing on the Trough fire involved 60 ac being treated at a cost of \$450 ac^{-1}. This cost included unit layout, work crews, straw, trucking costs and helicopter flight time. In March 2002, San Dimas and Erickson Air-Crane contacted the Forest to test a new hydromulch formulation in a burned area. The helicopter hydromulch drops were effective in creating swaths along two steep burned streams. Cost for the treatment was \$3,000 ac^{-1} plus mobilization. During treatment, some germinated native plants were covered with hydromulch. Over time, most of these perished. The Forest Botanist surveyed treated and untreated land. Untreated areas had more plant diversity than the treated area. In November 2002, the hydromulch area was inspected for product effectiveness. After a 6 inch rain, the mulch was mostly intact with native plants germinating through the mulch. Surface erosion and some rill erosion were halted by the hydromulch.

Introduction

There are two types of aerial mulching techniques, straw and hydromulch, that were evaluated on the Trough fire. In August 2001, the fire occurred on the Grindstone Ranger District, Mendocino National Forest (Forest), west of the town of Stonyford, CA. Elevations burned were between 1,500 and 6,000 ft. Low elevation vegetation consisted of chaparral, while upper elevations consisted of conifers. Upper elevations of the fire were in the Snow Mountain wilderness.

A burned area emergency response (BAER) team was assembled to evaluate the effects of the wildfire on resource values. Values that could be protected by aerial mulching were the road system (County and Forest Service), OHV (off-highway vehicle) trails, a resident trout stream, and sedimentation of Stony Gorge reservoir.

The USDA, Forest Service, San Dimas Technological and Development Center (SDTDC) contacted the Forest about evaluating the use of helicopters in the dispersal of straw and hydromulch. The Forest agreed that there was a need to spread straw in a timely manner on remote locations and that use of a helicopter could be the most efficient method. Aerial straw mulching occurred in the fall of 2001. In the spring of 2002, SDTDC once again contacted the Forest about evaluating the use of helicopters

[1]An earlier version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.
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in dispersing hydromulch on burned areas. As the Forest was interested in furthering technology, locations were provided to SDTDC and Erickson-Air Crane to do evaluations.

Aerial Straw Mulch

In the past, straw mulch has proven to be very effective in protecting burned slopes. Robichaud and others (2000) documented this finding, as did visual observations by Luckow and Jaramillo (2001) on the Cerro Grande fire in New Mexico and Faust on the Fork fire,[3] Mulch dissipates rain drop impact; keeps soil, nutrients and seed in place; and detains surface runoff to increase infiltration of water into the soil.

In 1996, the Mendocino National Forest BAER and implementation team recommended and applied rice straw mulch to 600 ac of the Fork fire. Hand mulching was very expensive (\$1,200 ac^{-1}) using California Conservation Crews and very time consuming carrying bales to the mulch sites and spreading the straw. Also, due to worker safety, only the ridges were treated. Steeper stream bank areas were left untreated.

After the Trough fire, to protect downstream values the BAER team recommended use of straw mulch. However, based on past experience and cost on the Fork fire, straw mulching was not selected.

Kim Clarkin, Research Hydrologist, at the San Dimas Technological and Development Center contacted Forest Hydrologist Faust about a burned area to evaluate aerial straw mulching. While burned treatment areas were being laid out, SDTDC was working with the California Straw Supply Co-op, located in Colusa County, on straw bale preparation and delivery systems.

The evaluation was to determine if helicopter application would be less expensive than hand treatment, if the mulch coverage would be similar to hand dispersal, and if steep slopes could be treated. A four hook carousel used to carry cargo nets was also evaluated.

Preliminary test drops showed that straw would give somewhat uniform coverage. Straw in the drops ranged from fine layers of individual straw stems to clumps about 3 in. deep and 2 ft. wide producing about 60 percent ground cover.

Variables in rice straw preparation that affected evenness of aerial distribution were moisture of straw at baling, chopped length of straw, straw bale weight and length of time bales were stored. Moist, long lengths (>10 in.) of straw, baled at high compression and stored for several months, creates flakes of straw that do not break apart in the air. Another factor to consider in using straw is that it be certified weed free in the county where it is applied.

To provide more information for this presentation, findings from the Haymen fire (Denver, Colorado) and Darby fire (Sonora, California) are included.

[3] Unpublished data on file, Mendocino National Forest, Willows, CA.

Methods

There are several steps that need to be considered in conducting aerial mulching. Some of the steps are unit delineation and location, an aviation safety plan, staging area location and operations.

Treatment Units

On the Trough fire, burned firelines, burned brush channels and upland slopes were marked with red panel flagging (approximately 2 ft wide and 4 ft long) for treatment. Average size of units was about 10 ac Plastic markers were held down with steel staples and rocks. Flagging was used to outline the treatment area. Unit corners were located with a GPS unit.

On large fires, treatment unit corners can be determined from the BAER burn intensity map. With either unit designation method (field or map), units need to be plotted on a topographic map to aid the pilot in locating the treatment areas.

Aviation Safety Plan

For Forest Service operations, a helicopter manager and a helibase manager (if a large operation) are needed to write an aviation safety plan and determine a suitable helicopter area of operations. The aviation safety plan covers such items as project objective and description, helicopter and pilot certification, radio frequencies, flight following, ground support, environmental and aerial hazards and load calculations.

Staging Area

Factors to consider in locating a staging area are space for hay truck access, straw bale storage, helicopter work area, flight paths and support vehicles. Another important factor to determine is if noxious weeds are located on the staging area and to determine how not to spread the seeds to treatment areas.

Normally on the staging area, there is a fire engine or water truck to wet down the area to reduce blowing dust. There should be enough cargo nets to have empty nets being loaded while the helicopter is flying out loaded nets to the treatment areas. Square nets that work well have flat straps and are 12 x 12 ft, 15 x 15 ft or 20 x 20 ft. A variety of net sizes are useful if using small and large bales. Straw bales are placed on the net with the side edge up to facilitate the removal of the baling twine.

Operations

A hand crew with small bales or a loading "squeeze" for the heavy large bales can do net loading. Hand loading is very fast and tiresome. A 15 to 20 person crew is needed to keep up with the helicopter. Crew rotation on a daily basis would keep the crew fresh and less prone to accidents.

For firelines, four 12 x 12 ft cargo nets were each loaded with five small (65 lb) bales each. The nets were attached to a specially made heavy duty four hook carousel developed at West Wind Helicopters, Lincoln, CA. As the pilot flew down the fire line, one net corner was released. When that net was almost empty of straw, another net corner was released. Very accurate placement of straw was made using this technique providing there was no wind or a light constant wind. Wind gusts can cause straw to miss the fire line.

Monitoring of Straw Distribution

On the Trough fire, straw was applied at one ton ac^{-1} with 65 ac treated in 2 days. Since the straw preparation technique was not perfected, the straw came out as clumps. With most of the weight in clumps, ground cover was about 35 to 40 percent. There was concern that the clumps would not provide enough ground cover to protect the soil from erosion. However, after the first rainstorms, it was observed that the straw clumps did not blow away as did the fine material and they were large enough to stop rill formation and catch sheet erosion. In comparison, on an adjacent untreated private land hillside, rill erosion was extensive as seen by streaks oriented up and down the hillside.

After aerial straw mulching was conducted on the Mendocino N.F., the technique was used on the Oregon fire (Weaverville Ranger District, Trinity N.F.). The next fire using aerial straw mulching was the Darby fire (Stanislaus N.F.). With each fire, straw preparation and application was perfected. Results on the Darby fire were excellent. On this fire, straw was applied at two tons $acre^{-1}$ using the large 800 lb bales. Ground cover was 90 to 100 percent and the wind did not blow the straw, due to straw being interlocked and below a burned tree canopy. Janicki and Grant (2001) reported that no erosion occurred on the treated slopes, but the untreated slopes had heavy sheet and rill erosion. On this fire 70 ac were treated in 3 days or 23 ac per day.

Cost of Aerial Straw Mulching

Factors influencing the cost of aerial straw mulching are trucking costs, application rate, acres treated and helicopter/pilot rates. For three California fires, Trough, Oregon and Darby, the costs per acre were $447, $328 and $710 respectively. The latter fire had an application rate double (2 tons ac^{-1}) the other two fires (Clarkin and Dean 2003).

Two 2002 fires in New Mexico, Rodeo and Borrego, had costs per acre of $290 and $650. The Rodeo fire used large bales (800 lb.) at an application rate of ½ t ac^{-1} and treated almost 11,000 ac. The Borrego fire treated 155 ac at just over 1 t ac^{-1} but had high costs due to long flight turn around times (3 min.) and hand unloading of trucks and hand loading of nets.

The Mollie fire in Utah had the highest cost per acre of about $950. These high costs resulted from having very light small bales, small nets, great elevation difference between the staging and treatment areas, resulting in long turn around times (7 to 9 min. per load). The project manager thought if they had a sufficient number and larger sized nets the cost per acre would be between $700 and $800.

In general, one would expect that an application rate of 1 to 1¼ t ac^{-1} would average about $500.

Results and Discussion

Aerial straw mulching was impressive in that many acres can be treated in a short time period at a reasonable cost compared to hand application. Also, remote and steep areas can be treated, and spreading of straw by air is safer than by hand where crew members can slip and fall on the slick straw. The following are other findings:

1. Aerial application can treat the steeper more erosive slopes,
   and is much faster than hand application.

2. Wind to 15 mph is beneficial to disperse straw on large units,
   but a detriment on narrow fire lines.

3. Mulch gives ground cover >70 percent if applied at 1.25 t ac^{-1}.

4. Mulch greatly reduced sheet and rill erosion.
   Straw may move with the wind or gravity on steep slopes.
   Straw clumps are a benefit, as they do not blow in the wind.

5. Bale straw at low (<15 percent) moisture content to reduce creating flakes.
   Bale straw loosely compacted for better dispersal.
   Weight of small bales about 65 lbs.; large bales about 800 lbs.

6. Chopped straw should have stems 5 to 8 in. long to aid dispersal.

7. Use freshly baled straw as older straw compacts and molds.

8. Keep straw dry during storage, transport and at staging area.

9. On small fires, for quality control, straw can be obtained from one supplier.
   On the Rodeo fire, straw was purchased from numerous vendors.
   For quality control, straw bales were hauled to a staging area,
   chopped and re-baled before being aerially dispersed.

10. Aerial straw mulching usually costs in the range of $350 to 800 per ac.
    Cost variables are:
    Price of straw–about $3.50 per bale.
    Trucking costs–$2.00 mi^{-1} on trips longer than 300 mi.
    $3.00 mi^{-1} for shorter trips.
    Type II restricted helicopter rates–$1,250 to $1,350 hr^{-1}.

11. Method of loading nets–hand (contract or inmate crew) or
    machine (front end loader or squeeze).

12. Weather delays–flight restrictions on wind speed and visibility.

13. Bale size–larger bales are more efficient to use than small bales.

14. Flight time–turn around time normally 1 to 3 min.

15. Aerial straw mulching is a very worthwhile treatment.

Aerial Hydromulch

Methods

Ryan Becker at San Dimas Technological and Development Center contacted the Forest rehabilitation team leader Faust about finding a test site for Erickson Air-Crane (Central Point, Oregon) to try a new chemical (LACCOSET, Sequoia Pacific Research, Draper, Utah) formulation in hydromulch. Two areas, comprising six acres, on the Trough fire were located to determine if the LACCOSET, which uses less water, would 1) provide 90 percent ground cover, 2) treat steep stream banks and 3) would last during a hot summer.

Treatment was in March 2002, when the seedbed was germinating and the brush was sprouting. Hydromulch formulation was a bonded fiber matrix blended with water and paper fiber. Application rate on a per acre basis was 2,000 gal of water, 500 lb mulch including chemical, and 50 lb cereal barley. Water was supplied by water trucks. Treatment time was one day.

On the Haymen fire, 1,560 ac were treated. This high quantity operation required five mixing tanks and 3,500 gal of water per acre. Forty ac ft. of water was drafted from a river. Fifty-one acres were treated per day.

On the Trough fire, each hydromulch swath was about 40 ft wide and 700 ft long. With two drops along one side of a stream swath width, with overlap, was about 70 ft This equates to one acre of treatment. The layer covering the soil was about 1/16 in. thick.

The helicopter pilot was able to maneuver the ship down the winding channels. On the Haymen fire, hydromulch treatment was on ridge tops and channels. Another alternative would have been to use straw mulch on the ridges and hydromulch on the steep stream banks.

Results

Effects on Vegetation

Within the Trough hydromulch area, a rectangular piece of panel marker 2 x 3 ft was used to make an untreated plot. Two months after application, the Forest botanist counted ten more species than in the adjacent treated area. In the treated area, sprouted forbs were covered with hydromulch and most perished.

One foot square plots were established to measure plant growth and to evaluate the hydromulch. After application, barley seeds were observed in the plots (10 to 13 seeds) but they did not germinate as no rain occurred after treatment.

In November, after a dry, hot summer, it rained 6 inces in 3 days. At the end of November, seeded barley to 3 inch tall was observed growing in the cracks of the mulch and bare spots created by animal hooves, whereas solid areas of mulch only had a few blades of grass growing through. Since seeded grass was not growing in either open or covered areas, it is assumed that the seed desiccated during the summer.

Effects on Soil Erosion

Bare loose soil on the 70 percent slope stream banks was held in place by the hydromulch. Ground cover was in excess of 95 percent. After the 6 in. of rain, ground cover decreased to 90 percent in one area, 40 percent in another area and 80 percent on another stream bank.

The hydromulch formulation (not LACCOSET) used on the Haymen fire reportedly disintegrated with only ½ in. of rain. (Pers. comm.). This shows that various formulations have varying degrees of longevity to sunlight and precipitation.

A couple of spots on the Trough fire evaluation site were tested for soil hydrophobicity. Surprisingly, burned chamise leaf needles under the hydromulch appeared to still be hydrophobic, whereas burned chamise needles on the bare area were not hydrophobic. However, the hydromulch still readily accepted water when wetted with water from a bottle.

Surface erosion was observed on the slope above the treatment as linear deposits of soil and ash (waves) about ½ inch high. When these waves hit the hydromulch they ended. No surface erosion occurred on or under the hydromulch.

One rill from above the treatment area entered the hydromulch and dissipated. Another rill that was enlarging as it entered the hydromulch decreased in size as it moved through the treatment area to the channel.

Cost of hydromulching for the Trough and Haymen fire was between $2,500 and $3,000 ac^{-1}. Mobilization was an additional $20,000 to $80,000. High treatment cost can be justified when important downstream values such as domestic water supplies, structures, or threatened or endangered fish are threatened by erosion.

Summary

The Trough fire presented an opportunity to evaluate aerial application of straw and hydromulch. Spreading of straw by air was quick to cover large acreages consisting of remote and steep sites. Even though ground cover percent was not as high as hand spreading, use of helicopters gave adequate ground cover to impede erosion at a cost similar to hand crews. The main advantage of aerial straw mulching is that steep slopes can be treated much more safely and quickly than using a hand crew.

Spreading of hydromulch by air also worked very well. The hydromulch provided at least 90 percent ground cover and stuck to very steep stream banks. Cost of this application is higher than use of straw, but where downstream values are high and there is a need for cover on steep slopes to stop erosion, this method is excellent.

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Pre- and Postfire Distribution of Soil Water Repellency in a Steep Chaparral Watershed[1]

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Abstract

The development and nature of water repellent soils and their spatial distribution on the landscape are not well understood relative to evaluating hillslope response to fire. Soil water repellency is particularly common in chaparral communities, due in part to the coarse-textured soils, and the high resin content of the organic litter. Objectives of this study were 1) to investigate pre- and post-fire distribution of soil water repellency on the landscape, and 2) to determine if species composition affected changes in soil water repellency. A prescribed burn was conducted on a 1.28 ha mixed chaparral study site located in the San Dimas Experimental Forest. We sampled pre- and post-fire soil water repellency using the water drop penetration time method at 105 sites. At each site, measurements were taken at the surface, 2 cm, and 4 cm depths within a 15 x 15 cm square plot. Thirty-eight percent of the pre-fire soil surface exhibited moderate to high repellency. At 7 days post-fire, moderate to high repellency in the surface soil increased to 66 percent. After 76 days, post-burn surface soil water repellency returned to near pre-fire values. Variability in water drop penetration time among replicates within a given 15 x 15 cm plot was as large as the variability seen among sites over the whole watershed. At the 2 cm depth, 7 day post-fire moderate and high repellency increased 14 percent. Beneath chamise, soils exhibiting high repellency increased by 17 percent after the burn, and by 38 percent under ceanothus, whereas high repellency beneath sugar bush decreased by 23 percent. Greater understanding of the pre- and post-fire spatial distribution of water repellency is necessary for land management decisions in steep chaparral watersheds.

Introduction

Wildfire can increase soil water repellency across a landscape. The spatial distribution of repellency can be highly variable for several reasons: non-uniform fire temperature and duration, depth and moisture content of the litter layer, litter type and composition, varying plant species, live and dead fuel densities and moisture contents, and differences in soil moisture and soil texture (DeBano 1981, Robichaud 1996). Therefore, most fires create a mosaic of low-, moderate-, and high-severity burn areas (Robichaud and Miller 1999). For example, dry south-facing slopes may burn at high severity, but moist north-facing slopes and drainage areas may burn at lower severities (DeBano and others 1998).

In a water repellent soil, water will not readily penetrate and infiltrate into the soil, but will “ball up” and remain on the surface (Adam 1963, DeBano and others 1967). Soil water repellency is particularly common in chaparral communities, due in part to the coarse-textured soils, and the high resin content of the organic litter (DeBano 1981). Coarse-textured soils generally exhibit more intensive soil water

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repellency because there is less particle surface area to be coated by hydrophobic substances than would be in a finer textured soil (DeBano 1981). Fire volatilizes the existing organic water-repellent materials, which then can move vertically down the soil profile and recondense on mineral soil particles producing a more continuous water-repellent layer (DeBano and others 1967). Nevertheless, live roots, old root channels, and soil cracks can serve as penetrating points for preferential flow of water through the water repellent layers thus providing water to the deeper rooted chaparral species (Meyer 1993).

Because prescribed fires do not burn uniformly across a broad landscape, or even at smaller scales in steep watersheds, the spatial patterns of soil water repellency should not be expected to be uniform (Robichaud and Miller 1999). This work addresses prescribed pre- and post-fire distribution of soil water repellency in a steep chaparral watershed. Specific objectives were to 1) determine pre- and post-fire spatial and temporal variability of water repellency in steep chaparral soils, and 2) investigate how different plant species affect the distribution of soil water repellency on the landscape.

Fire can exacerbate the amount of soil water repellency across the land surface, reducing infiltration, and, in turn, increase overland flow (Shakesby and others 2000). But a mosaic distribution of water repellency accompanied by preferential flow pathways may lower the intensity of the hydrologic response and resulting erosion (Meyer 1993, Ritsema and Dekker 1994). Thus, it is critical that land managers understand pre- and post-fire spatial distribution of water repellency in steep chaparral watersheds.

Materials and Methods

The 1.28 ha study watershed (34° 12’45” N, 117° 45’ 30” W) is located in the San Dimas Experimental Forest (SDEF) in the foothills of the San Gabriel Mountains of southern California, approximately 45 km northeast of Los Angeles, California. The climate is Mediterranean, with hot, dry summers, and cool, wet winters. Mean annual precipitation is 678 mm (Dunn and others 1988). Topography is rough with precipitous canyons and steep slopes (Ryan 1991). Bedrock in the area has been subjected to intense heat and pressure resulting in a high degree of alteration, faulting, folding, and fracturing, resulting in rocks that are poorly consolidated and very unstable (Sinclair 1953). Soils are coarse-loamy, mixed, thermic Typic Xerorthents (Hubbert and others 2006) and are low in organic matter content (Ryan 1991). Common chaparral species were chamise (*Adenostoma fasciculatum* Hook & Arn.**)**, hoaryleaf ceanothus (*Ceanothus crassifolius* Torr.), sugar bush (*Rhus ovata* S. Watson), bigberry manzanita (*Arctostaphylos glauca* Lindl.), scrub oak (*Quercus berberidifolia* Liebm.), black sage (*Salvia mellifera* E. Greene), and wild buckwheat (*Eriogonum fasciculatum* Benth.). Chaparral stand age was 41 yr, with the watershed last burning during a 1960 wildfire that consumed 88 percent of the 7,727 ha SDEF.

The study watershed was burned on May 15, 2001, ignited by flaming gel applied by a helicopter. Residence time of the prescribed burn ranged from 1 to 3 hr, consuming approximately 90 percent of the 1.28 ha study watershed. Burn severity was estimated by measuring the minimum diameter of chamise branches remaining post-fire. Burn intensity (heating) was estimated qualitatively from the color of ash and degree of litter consumption (Miller 2001).

There were 105 sampling plots within the 1.28 ha watershed. Fifty-four of these sampling plots were randomly spaced along eight transects that crossed the watershed in an east-west direction. The remaining 51 plots were selected randomly between transects. Pre-fire cover by species, litter composition and litter depth were identified at each sampling site. Pre- and post-fire soil water repellency was determined by noting the water drop penetration time (WDPT) (Krammes and DeBano 1965) in dry surface soil at each sampling point. Pre-fire litter and post-fire ash were carefully removed to expose the soil mineral surface. Twenty drops from an eyedropper were applied at the mineral soil surface within a 15 x 15 cm area. Another 20 measurements were taken at the 2 cm depth and 10 measurements at the 4 cm depth. The time of water drop penetration was determined when the droplet changed from convex to flat, and infiltrated. We have modified existing soil water repellency indexes (DeBano 1981, Dekker and Ritsema 2000) to give us the following classification scheme: 0 to 1 s–not repellent, 1 to 5 s–very low repellency, 5 to 30 s–low repellency, 30 to 180 s–moderate repellency, and >180 s–high repellency. This classification scheme is in agreement with Robichaud (1996), except that the time for "no repellency" is 0 to 1 s instead of 0 to 5 s.

The smooth surfaces in *figures 1, 2*, and *3* were produced by the nonparametric smoothing routine (loess) located within the S-plus statistical package (Cleveland and others 1992, S-PLUS 2001). The technique is based on locally weighted regression methods. It fits a polynomial surface by weighted least squares with the weights picked such that nearby data points have the most influence.

Results and Discussion

Burn severity (effects of the fire) was low to moderate throughout the watershed, with litter depth and moisture content contributing to the landscape distribution of soil heating. Thirty-eight percent of the pre-fire soil surface exhibited moderate to extreme repellency (>30 s), 22 percent low repellency (5 to 30 s), and 40 percent very low to no repellency (<5 s) (*fig. 1b*). In the surface spatial plot (*fig. 1a*), the variability of surface water repellency extended from the lower to upper watershed. Krammes and DeBano (1965) suggested that at least 60 percent of pre-fire soils on SDEF could be classified as water repellent. In the pre-fire soil landscape, continuous layers of water repellent soil are generally not formed because rodent, worm, insect, and root activity create macropores for water infiltration (DeBano 1981, Spittler 1996). We observed almost no pre-fire water repellency at 4 cm depth (*fig. 3b*). We believe this was partly due to higher pre-fire soil moisture contents (median >0.12 cm^3 cm^{-3}) at 4 cm depth and the effect of time since the 1960 wildfire.

At 7-d post-fire, moderate to high repellency in the surface soil had increased to 66 percent, with very low and no repellency soils dropping 18 percent (*fig. 1a*). The increase in surface water repellency was clearly seen in the post-fire 7-d spatial plot (*fig. 1a*), but variability still remained high throughout the watershed (*fig. 1d*). The amount and distribution of chaparral fuel that collapses during fire and smolders on the ground can cause pronounced spatial variation in total surface heating (Odion and Davis 2000). Post-fire 7-d increases in surface water repellency observed in our experiments resulted from release of hydrophobic substances contained in litter and plant material during combustion (DeBano and others 1970).

At 76-d post-burn, repellency patterns appeared to be returning to pre-fire spatial patterns (*fig. 1e*) and the histogram showed surface soil water repellency returning to near pre-fire values (*fig. 1f*). In most cases, soil moisture plays an important role in the decrease of water repellency (Robichaud 1996, Doerr and Thomas 2000). However, no precipitation occurred during the post-fire 76-d time period and median soil moisture contents remained below 0.5 cm^3 cm^{-3}. It is unknown if fog drip resulted in any decrease in surface repellency during this time period. We believe the breakdown of water repellency (*fig. 1*) was due to movement of the unstable surface soils on the very steep slopes (>55 percent), wind erosion, and perturbation of the soil by plants and fauna. It was unlikely that decomposition and displacement of water repellent substances by microorganisms (Crockford and others 1991) or by fine roots occurred, possibly because of low soil moisture conditions. At the 2 cm depth, moderate and high repellency increased from 33 percent pre-fire to 47 percent 7 day post-fire (*fig. 1b*).

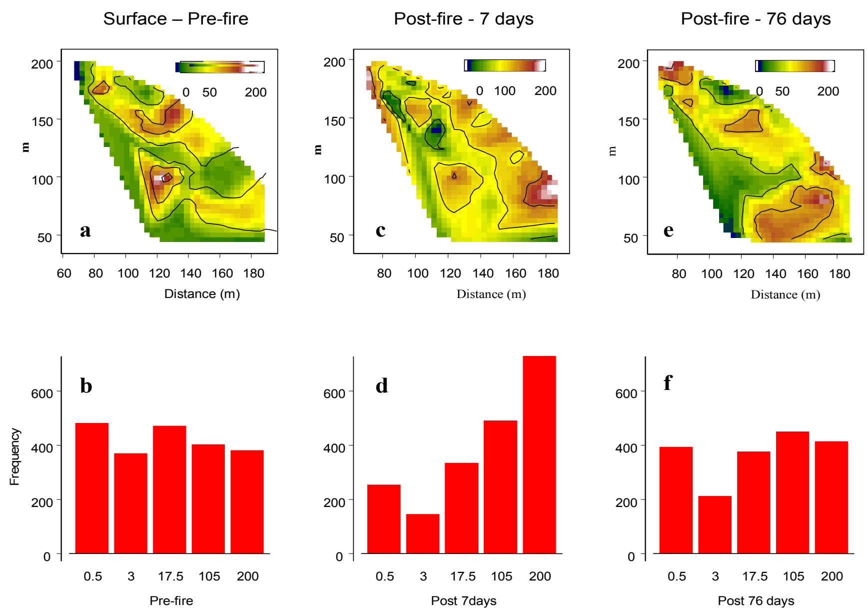

Figure 1—Spatial plots and corresponding histograms describing pre-fire and post-fire surface soil water repellency. Water repellency times were grouped into 5 cells. The x-axis groups (0.5 s, 3 s, 17.5 s, 105 s, and 200 s) represent the repellency index: 0 to 1 s–not repellent, 1 to 5 s—very low repellency, 5 to 30 s–low repellency, 30 to 180 s–moderate repellency, and >180 s–high repellency. The heights of the bars are the total number of cases (at all locations) that fell in the corresponding group.

DeBano and others (1970) noted that increases in repellency were due to volatilization and recondensation of organics moving downward along a temperature gradient, with repellency intensified at temperatures of 175 to 200°C. However, temperatures in the surface soil may not have been high enough to volatize the organics. At 7 day post-fire, “moderate to high” soil repellency at the 4 cm depth increased to 26 percent (*fig. 3d*). Although the fire was of low to moderate intensity, it appears that the increase in water repellency at depth was due in part to volatilized organics translocated downward and soil drying. After 76 day, both the 2 and 4 cm depths had not returned to pre-fire values as seen at the soil surface and moderate to high repellency continued to increase beyond post 7 d (*fig. 2f, 3f*). We believe this was a function of drying soils and the absence of surface soil erosion factors of wind and gravity. The continued increase in repellency may have been due to further soil drying by evapotranspiration, and seasonal soil temperature change at the 4 cm depth that re-intensified latent soil water repellency. As reported by Dekker and others (1998), even slight drying at 25°C can contribute to an increase in soil water repellency at the 2.5 to 5 cm depth.

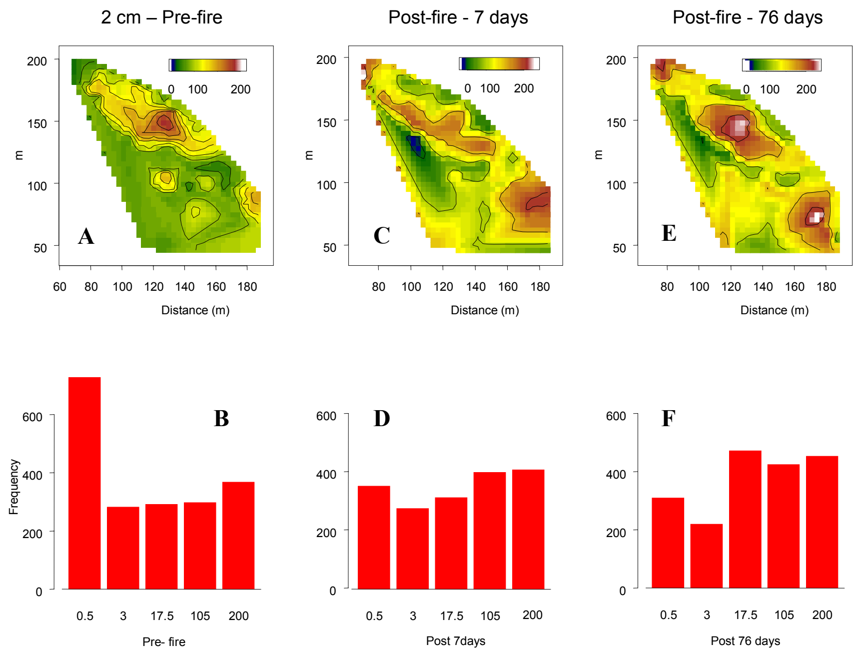

Figure 2—Spatial plots and corresponding histograms describing pre- and post-fire soil water repellency at 2 cm depth.

Variability in WDPT among replicates within a 15 x 15 cm plot was as large as the variability among sites over the study watershed (*table 1*). At the soil surface, approximately 50 percent of the overall variability in times of water drop penetration was due to within site variation. The percent of total variation due to within site variability seemed to decrease at lower depths (approximately 40 percent at 2 cm and 30 percent at 4 cm). Variability was greatest at the surface and decreased with depth (*table 1*). At the 4 cm depth, lower within site variability values were partially due to higher soil moisture contents as compared to dryer moisture contents at the surface and 2 cm depth. Type and species composition of litter played a role in the post-fire variability of soil water repellency (*fig. 4*), as reported by Doerr and others (1998).

Beneath chamise, soils exhibiting high repellency increased by 17 percent 7 days after the burn, and under ceanothus by 38 percent, whereas high repellency beneath sugar bush decreased by 23 percent (*fig. 4*). In the study watershed, the depth of chamise litter averaged only 19 mm (Hubbert and others 2006), but still exhibited >35 percent water repellency 7-d post-fire (*fig. 4*). DeBano and others (1970) have noted that leaf materials of chamise contain large amounts of hydrophobic substances. It appeared that ceanothus litter (average depth=55 mm) also contained large amounts of hydrophobic substances that were released during fire, but were not released during intervening periods absent of fire. The low (<4 percent) water repellency beneath ceanothus indicated that very little hydrophobic substances were leached out of the leaves. The decrease in high repellency beneath sugar bush (average litter depth=62 mm) may have been due to higher fire-induced surface soil temperatures or lower leaf contents of hydrophobic substances. We observed an overall increase in repellency under all species categories post fire. Since the majority of chaparral species are resprouters, the enhanced post-fire water repellency may discourage seed emergence and development, thus preventing competition for water.

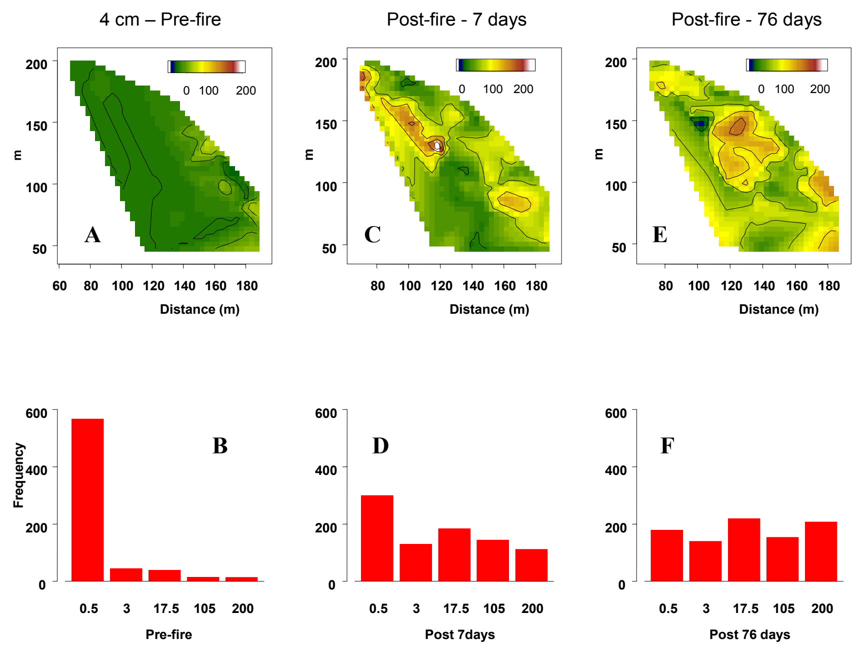

Figure 3—Spatial plots and corresponding histograms describing pre- and post-fire soil water repellency at 4 cm depth.

Table 1—*Percent variability of water drop penetration time measured between replicates and between sites.*

	Pct between sites	Pct within 15 x 15 cm site
Surface pre-fire	58.2	41.8
Surface 7-d post-fire	47.5	52.5
Surface 76-d post-fire	51.0	49.1
2 cm pre-fire	66.6	33.4
2 cm 7-d post-fire	54.6	45.4
2 cm 76-d post-fire	57.9	42.1
4 cm pre-fire	74.8	25.2
4 cm 7-d post-fire	65.1	34.9
4 cm 76-d post-fire	60.9	39.1

Water repellent layers may be beneficial to some chaparral species. Burrow holes, tension cracks, old root channels, and live roots can serve as penetrating points for preferential flow of water when ponding occurs due to repellency and may help contribute to the high within and between site variability of soil water repellency exhibited at the soil surface. These macropores provide conduits for infiltration to lower soil horizons, thus bypassing water-repellent layers near the surface (Imerson and others 1992). High rates of hillslope runoff are prevented from occurring, and water storage at greater depths in the soil is enhanced, permitting its later use in transpiration (Scott and Lesch 1997, McDonald 1981). Deep water allows fast post-fire recovery of deeply rooted chaparral resprouters, which are not affected directly by the water repellent layer, because their root systems remain intact, both above and below it (Meyer 1993).

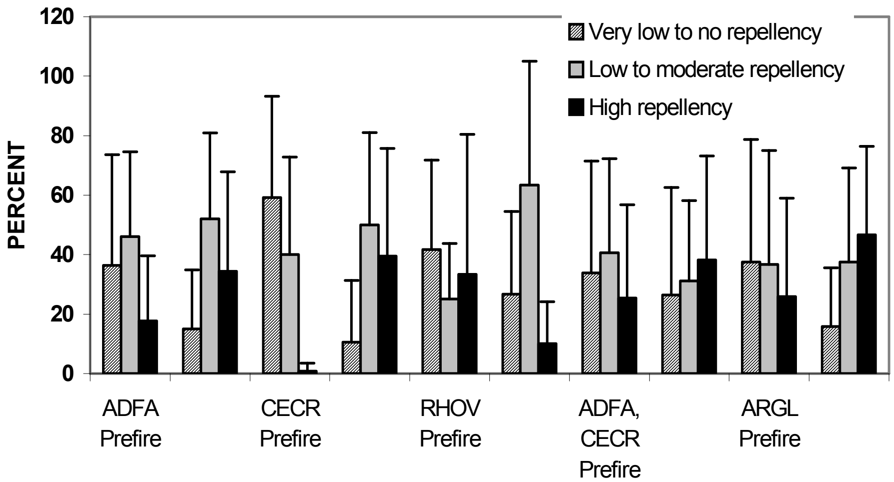

Figure 4—Percent of surface soil water repellency in relation to species litter composition pre-fire and 7 day post-fire. ADFA=chamise, CECR=ceanothus, RHOV=sugar bush, ARGL=manzanita. Error bars represent one standard deviation from the mean. Very low to no repellency <5 s, low to moderate repellency 5 to 180 s, and high repellency >180 s.

Conclusions

At 7 days post-fire, moderate to high repellency in the surface soil had increased by 38 percent from 28 (pre-fire) to 66 percent. After 76-d post-burn, surface soil water repellency returned to near pre-fire values, but increases in repellency at the 2 and 4 cm depths remained. The variability of surface and 2 cm depth water repellency across the watershed was high pre-fire, 7-d post-fire, and 76-d post-fire. Variability in WDPT among replicates within a given 30 cm^2 plot was as large as the variability seen among plots over the study watershed. At 2 cm depth, 7-d post-fire moderate and high repellency increased 14 percent, suggesting volatilization and recondensation of organics moving downward on a temperature gradient. The 38 percent increase of high soil repellency beneath ceanothus suggested that large amounts of hydrophobic substances from leaf materials are released during fire.

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Effects of Three Mulch Treatments on Initial Postfire Erosion in North-Central Arizona[1]

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Abstract
Mulching after wildfires is a common treatment designed to protect bare ground from raindrop impact and reduce subsequent erosion. We tested the effectiveness of three mulching methods on the Indian Fire near Prescott, Arizona, USA. The first method felled all fire-killed trees, chipped the logs and limbs, and spread the chips across the hillslope with a mobile self-feeding chipper. The second treatment spread compressed, tackified straw pellets that expand when wetted and release a soil flocculant. The third treatment was rice straw applied at 4.5 Mg ha^{-1} (2 tons ac^{-1}). Each treatment was applied to a small catchment with a silt fence sediment trap at the mouth. Sediment yield from an untreated (control) catchment was also measured. The treatments were tested by three erosion-causing summer rain events. The chipping treatment and the pellets reduced sediment yield by 80 to 100 percent compared to the control in the first two storms. In the third event, a multi-day storm followed by an intense thunderstorm, the pellets and straw reduced sediment yield 42 and 81 percent, respectively. The effectiveness of the chip treatment could not be completely assessed because of partial failure of the sediment fence. Vegetation cover was low on all sites; ground cover from pellets decreased more than did straw or chips by mid-October, probably accounting for the lower effectiveness in reducing erosion compared to straw.

Introduction

Applying mulch to protect bare ground is regarded as one of the most effective methods for reducing post-fire erosion (Bautista and others 1996, Miles and others 1989, Robichaud and others 2000). Rice or cereal straw is most commonly applied by hand to hillslopes above high-value assets such as roads, streams or reservoirs. Hand-application makes this method relatively labor-intensive and expensive to employ (Miles and others 1989, Robichaud and others 2000).

There is a great deal of interest in developing mulch methods that can be applied more easily or use on-site materials. We initiated a project to use two relatively new mulch treatments – on-site whole tree chipping and compressed straw pellets – on a 2002 wildfire site in north-central Arizona. This paper evaluates the initial erosion-control effectiveness of these treatments compared to a standard method—hand-applied rice straw—and an untreated control.

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Methods

The May 2002 Indian Fire burned approximately 550 ha (1,340 ac) of ponderosa pine (*Pinus ponderosa*) forest with scattered oaks (*Quercus emoryi*) on the Prescott National Forest, just southwest of Prescott, Arizona, USA. A large portion of the fire area, including the location used for this project, was rated as high burn severity, with complete or nearly complete consumption of tree foliage and total removal of ground cover.

The study site was located approximately 4 miles south of Prescott on an east-facing slope at an elevation of 1,800 m (5,900 ft) in the Indian Creek drainage. Soils on the site are classified as lithic ustorthents, derived from granitic parent material. Average annual rainfall in nearby Prescott is 487 mm (19.2 in). Different mulch treatments were applied to small (0.34 to 0.48 ha [0.83 to 1.20 ac]) zero-order watersheds with similar slope geometry situated adjacent to each other. Slope angles increased from less than 10 percent at the bottom to 30 to 40 percent in the upper portions of each watershed.

Each of the three mulch treatments was applied to one small watershed, with a fourth watershed left untreated as a control. A 10 to 15 m wide silt fence was installed at the base of each watershed, where slope angle was less than 10 percent, to serve as a sediment catchment basin, following the method of Robichaud and Brown (2002). A second silt fence was installed below the first to serve as a back-up in case of sediment overflow or fence failure. Silt fence material (supplied by GeoTK LLC, Vancouver, WA, USA) was supported on standard steel fence posts (T-posts) spaced approximately 1.5 m apart. The fences were approximately 1.3 m tall. After installing the fences, the floor of each basin was covered with a layer of construction marking chalk. This facilitated finding the bottom of the basin when digging out sediment for measurement. Mulch treatments were applied as equipment and materials became available. Silt fences were installed as near as possible to the time treatments were applied.

Two tipping-bucket recording rain gauges (Onset Computer Corp., Bourne, MA, USA) were installed on the site; each tip recorded 0.25 mm (0.01 in.) of rain. Data were periodically downloaded from the rain gauges to office computers for analysis via an Onset HOBO Shuttle™ and BoxCar Pro™ software. Rain gauge output was used to calculate total precipitation as well as 10-, 30- and 60-minute maximum rainfall intensities for each storm event.

The "chips" treatment consisted of felling all killed (pine) and top-killed (oak) trees in the catchment with a track-mounted feller-buncher, followed by chipping all logs smaller than 35 cm with a tracked whole-tree chipper. The discharge chute of the chipper could be rotated, allowing a certain degree of control over the distribution of the chips. Most chips produced were 1 to 2 cm wide by 2 to 5 cm long and about 1 cm thick, but they ranged in size up to 10 to 15 cm across and several cm thick. Depth of the chips varied considerably, from a few cm to several dm in isolated areas. Cover was originally nearly 100 percent. Chips were applied in mid-July 2002. The upper 10 to15 m of the watershed contained an archaeological site (largely rocks) that had to be excluded from treatment; this affected less than 10 percent of total watershed area.

The "pellets" treatment was applied on July 16. The pellets (supplied by Pelletized Straw, LLC, Manteno, IL, USA) were made of a highly compressed, pasteurized straw product bound with "Silt Stop," a proprietary polyacrylamidefamily linear polymer soil flocculant/tackifier. Pellets were 1.9 cm diameter and chopped to an average of 0.6 cm in thickness to prevent rolling after application; they are designed to expand 4-fold or more upon wetting and to be held in place by the tackifier. Pellets were hand-spread to produce 50 percent cover when dry in an effort to provide 100 percent cover upon wetting and expansion. Silt fences on the pellet catchment were installed August 1. One significant rain event occurred after treatment but before installation of the silt fences.

Rice straw ("straw" treatment) was spread the week of August 5 at a rate of 4.5 Mg ha^{-1} (2 T ac^{-1}), based on findings of Edwards and others (1995) that an application rate of 4 Mg ha^{-1} was optimum for minimizing erosion. Fences were installed the following week.

After each precipitation event that resulted in significant erosion in the catchments, we removed and weighed collected sediment from behind the silt fences. Stratification of the sediment was visually assessed, and the proportion of each class of sediment–fine, wet, ashy sediment near the fence to coarse, sandy, dry material upstream–was estimated. The sediment was then subsampled for moisture content, with the number of samples taken from each class based on the proportion of the total sediment estimated to be in the class. Wet sediment was weighed on an electronic balance to the nearest 0.1 kg and then discarded downslope of the edges of the silt fences. After sediment was removed to the original contour of the basin, a new layer of chalk was applied in preparation for the next event. Moisture subsamples were returned to the lab, weighed, and dried to constant weight in a forced air oven at 105°C. The moisture content of the subsamples and the contributing area of each catchment were used to convert the measured wet weight of sediment to mass of dry sediment per ha.

Ground cover by straw pellets, pellet remnants, rock, woody debris, ash, and bare soil was visually estimated in nine randomly placed 1 m^2 quadrats in the pellet catchment on August 1, after a rainstorm when the pellets had expanded and their cover was presumably maximal. Ground cover was estimated in all catchments on August 15, 2002 and again on October 28 by placing 1 m^2 quadrats every 5 m along three 50 m transects. Transects were roughly parallel and ran upslope in the middle and on each side of the catchment. Percentage cover was visually estimated in categories that included bare ground, mulch materials, rock, litter, live vegetation, woody debris, and so forth. Casual observation of the state of the mulches was also made at each site visit.

Results

Precipitation prior to installation of the first silt fences in July was light and did not cause sediment movement. Typical monsoon thunderstorms occurred on July 24 and August 4, 2002 (*fig. 1*). Each of these storms produced approximately 10 mm of rain (average of the two rain gauges) and measurable erosion (*table 1*). The maximum 10-minute rainfall intensity of each storm corresponded to a recurrence frequency of 1 year (Bonnin and others 2004); intensities calculated for 30 and 60 minutes had similar recurrence frequency (data not shown).

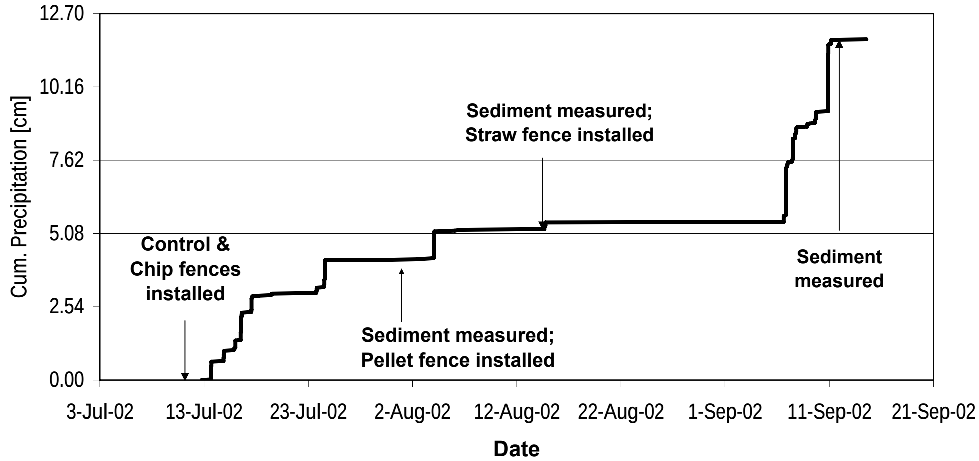

Figure 1—Cumulative precipitation (cm) and dates of silt fence installations and clean-outs at the Indian Fire study site near Prescott, AZ.

An extended rain event from September 6–9 yielded 38 mm of precipitation. We arrived at the site September 10 to find considerable deposition in all sediment basins and standing water accumulated behind several of the fences due to fine silt plugging the fencing material. That evening a thunderstorm produced 23 mm of rain in approximately 30 minutes. The maximum 10 minute rainfall intensity corresponded to a 10 yr recurrence frequency (Bonnin and others 2004) (*table 1*), but the 30 and 60 minute intensities were lower (2 yr recurrence; data not shown).

Following the late-July storm, at which time only the chip and control fences were installed, the chip catchment yielded 0.42 Mg ha^{-1} of sediment and the control 6.4 Mg ha^{-1} (*table 1*). While the pellet fence had not yet been installed, the treatment was inspected during the visit. The pellets were observed to have absorbed rainwater, expanded and released the soil flocculant, which was highly visible. Cover estimates of the pellets/flocculant ranged from a low of 60 percent in one plot, which had 20 percent rock and 5 percent litter cover, to a high of 95 percent in another plot. The remaining seven plots measured each had 85 or 90 percent pellet cover and 0 to 10 percent bare ground.

After the second rain event on August 4, at which time the pellet fences had been installed, we collected 10.8 Mg ha^{-1} of sediment from the control catchment, zero from the chip treatment, and 2.2 Mg ha^{-1} from the pellet treatment (*table 1*). Ground cover of the mulch materials averaged 81, 62, and 75 percent for the chips, pellets and straw, respectively, on August 15 (*table 2*), which was shortly after the second storm.

Because we were unable to clean the basins and measure the sediment (a multi-day task) between the two September storms, sediment yields from the two events were combined. Pressure from the runoff and sediment during the September 10 thunderstorm caused partial failure of the wood chip silt fence by tearing the material in the upper fence and collapsing a portion of the backup fence, allowing some sediment overflow and loss downstream. The combined rain events yielded a total of 48.4 Mg ha^{-1} of sediment from the control catchment, more than 15.5 Mg ha^{-1} from the chips treatment, 28.2 Mg ha^{-1} from the pellet treatment, and 9.1 Mg ha^{-1} from the straw catchment (*table 1*).

Table 1—*Dry weight of sediment and precipitation information for the Indian Fire site, Prescott, AZ, summer 2002. The September 14 value for the Chips treatment is an underestimate because some sediment was lost when the silt fences partially failed. Precipitation intensity recurrence interval was determined from Bonnin and others (2004).*

Cleanout Date:	**7/30/2002**	**8/14/2002**	**9/14/2002**
Treatment Watershed	Sediment yield in Mg ha^{-1} (T ac^{-1})		
Control	6.4 (2.8)	10.8 (4.8)	48.4 (21.6)
Chips	0.42 (0.18)	0 (0)	>15.5 (>6.9)
Pellets	N/A	2.2 (0.96)	28.2 (12.6)
Rice Straw	N/A	N/A	9.1 (4.1)
Total storm precipitation mm (in)	9.2 (0.36)	9.1 (0.36)	61.0 (2.4)
Maximum 10-min intensity mm hr^{-1} (in hr^{-1})	22.9 (0.9)	38.1 (1.5)	117.3 (4.62)
Intensity recurrence interval (years)	1	1	10

Table 2—*Average percent ground cover of mulches and bare ground on two sampling dates in 2002.*

Sampling Date:	**8/15/2002**		**10/28/2002**	
	Ground cover (pct)			
Treatment Watershed	Mulch	Bare	Mulch	Bare
Control	--	58	--	38
Chips	81	5	58	3
Pellets	62	9	27	25
Rice Straw	75	13	51	25

Discussion

Summer precipitation in the Prescott area is dominated by monsoon thunderstorms of variable frequency, beginning early- to mid-summer. While often of limited areal extent, these storms can be of considerable intensity, capable of causing significant postfire erosion. Three such erosion-causing storms occurred at the study site after the Indian Fire in 2002.

The wood chip treatment appeared very effective in reducing erosion during the late July and early August storms, with sediment yields 93 and 100 percent less, respectively, than in the control catchment. The storm intensities were typical of what could be expected every year in this area. Unfortunately the straw treatment was not applied in time for these two storms, so no comparison of the chips to the standard postfire mulch application can be made. The continuous and thick layer of cover provided by the wood chips undoubtedly accounted for the ground protection afforded during these relatively mild summer storms.

After the early August storm, the pellets were observed to have further dispersed, and flocculant was no longer obvious on the surface of the soil. Straw from the pellets had dispersed to fine chaff widely distributed across the ground surface, which would seem unlikely to have much impact in reducing erosion.

Nonetheless treatment effectiveness appeared very good, with sediment yield 80 percent less than the control. This decrease could be a result of the presence of soil flocculant in addition to extra ground cover provided by the chopped straw. For this second mild storm, the pellets were only slightly less protective than the wood chips.

The more intense September storm series provided the first test of the rice straw treatment. Sediment yield was 81 percent less than from the control catchment, confirming earlier studies that have shown good effectiveness for rice straw (Bautista and others 1996, Miles and others 1989), compared to a 42 percent reduction for the pellets. Although data are incomplete for the chips catchment, the treatment produced more sediment than the straw. Because this was the first storm after application of the rice straw, ground coverage may have been more even than in the other two treatments, which had been affected by earlier rains, at least during the first of the two September storms. We did note that the big event caused significant movement of straw, with substantial exposure of bare ground and clumping of straw, and rills formed between the clumps (compare cover on October 28 to August 15 in *table 2*).

Similarly, substantial movement of the wood chips was observed in September, especially on the upper, steeper portion of the catchment, and accumulation of chips atop the sediment in the basin was noted before the September 10 thunderstorm caused the fence failure. This indicates that wood chips are susceptible to being "floated" off a hillslope under conditions of sufficient overland flow. Lower chip cover in October (*table 2*) confirms the loss of material.

The two alternative mulch materials protected the ground effectively during the "typical" thunderstorms earlier in the summer, but they did not appear to perform as well as the freshly-applied standard treatment, straw, during the more intense late-summer storms. Thus there would seem to be no reason to use either instead of straw. The September 10 thunderstorm was a 10-year event in terms of its 10-minute rainfall intensity, however; thus not every postfire site would be subject to those conditions, and the use of straw pellets or wood chips could be justified if they had some other advantage over straw.

Both treatments differ from straw in that they cannot introduce weed seeds onto a burn site (the chopped straw in the pellets is pasteurized). Mulched areas can have greater abundance of nonnative species (Kruse and others 2004). In areas where rice straw, which contains few terrestrial weed seeds, is not available, these treatments could have an ecological advantage over mulching with ordinary cereal or pasture straw, which often contain weed seeds, despite lower expected erosion reduction effectiveness during intense thunderstorms. Also, the rapid decrease in cover of the pellet treatment (*table 2*) could make it preferable for use in areas where recovery of sensitive plant species after fire is a concern.

The effectiveness of these treatments will be monitored for additional years in this study. Wood chips would be expected to persist on the hillslope longer than either pellets or straw, and relative erosion control effectiveness of treatments may change in the future. The first year after fire is generally the most significant in terms of erosion, however, when soil protection is most needed (Robichaud and others 2000). The impact of mulches on vegetation recovery may also vary through time. Use of straw mulch after fire is increasing, particularly aerial application (Faust, this volume). It remains to be seen whether wood chips or straw pellets have substantial merit as postfire mulches that would recommend their use in place of straw.

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The Effects of Fire on Soil Hydrologic Properties and Sediment Fluxes in Chaparral Steeplands, Southern California[1]

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Abstract

Fire is a major disturbance event in southwestern ecosystems. A prescribed burn in the San Dimas Experimental Forest provided an opportunity to quantify the effects of fire on soil hydrologic properties and sediment fluxes in chaparral-covered steeplands. Located in the San Gabriel Mountains about 50 km northeast of Los Angeles, a 1.28 ha watershed was instrumented with hillslope sediment collectors and a debris basin. Sediment fluxes were measured for seven years prior to the fire and through the first post-fire winter. Soil samples taken just before and after the fire were analyzed for bulk density, texture, and moisture content. Soil non-wettability was determined pre- and post-fire at the same 105 locations using the water drop penetration test. Post-fire soils were denser, coarser, and much drier than prior to the burn. Although extremely variable, non-wettability generally increased after the fire. Post-fire hillslope erosion was twice as great as pre-burn levels during the dry season and increased by 10-fold in the wet season, despite a record drought year. Post-fire sediment yield was 20 times greater than the unburned annual average. We believe the fire-induced changes in soil properties coupled with reduced ground cover caused an increase in surface runoff that accounts for the large increases in post-fire sediment fluxes.

Introduction

Fire is a major disturbance event in semiarid southwestern ecosystems. Fire incinerates vegetation, alters soil properties, and renders the landscape susceptible to the forces of gravity, running water, and wind. Under these conditions, accelerated post-fire erosion is inevitable. This accelerated post-fire erosion is accentuated in the southern California mountains because of steep topography, non-cohesive soils, and intense rainfall events (Rice 1974). Often these factors combine to produce debris flows with great destructive power (Wells 1987).

At the wildland/urban interface (WUI) in southern California, where the fringes of urban development impinge on adjacent steep mountain fronts, accelerated post-fire erosion can seriously harm human communities. Lives are threatened, property is jeopardized, and infrastructure (roads, bridges, pipelines, utility lines) is placed at risk. Uncertainty about the magnitude of destruction associated with accelerated post-fire erosion stems from our limited ability to predict specific post-fire watershed responses. Unfortunately, prediction, usually in the form of risk assessment and planning that involves numerical modeling, can only derive from a sufficient understanding of the erosion problem and the quantification of fire effects and erosion processes.

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A prescribed burn in the San Dimas Experimental Forest provided an opportunity to document and quantify the effects of fire on soil hydrologic properties and sediment fluxes within a small watershed unit in a semiarid, chaparral-covered, steepland ecosystem. Results of this research could serve as a benchmark against which to test existing predictive models of post-fire erosion for the southern California area.

Study Site

Located in the San Gabriel Mountains about 50 km northeast of Los Angeles, the San Dimas Experimental Forest has been a wildland research site for hydrology and ecology for nearly seventy years (*fig. 1*). In conjunction with an ongoing fuel reduction operation, we selected a 1.28 ha chaparral-covered study watershed that was part of a 4 km^2 prescribed burn project. Situated at an elevation of about 1,000 m on a southerly aspect, hillslope angles in the study watershed average 31^o (60 percent) and stream channel gradients average 21^o (35 percent). The region experiences a Mediterranean climate regime, with cool, wet winters and hot, dry summers. Precipitation, falling almost exclusively as rain between the months of November and March, averages about 700 mm annually (Dunn and others 1988). Crystalline metamorphic and intrusive igneous bedrock produces non-cohesive sandy loam soils (coarse-loamy, mixed, thermic Typic Xerorthents) that support a mixed chaparral vegetation assemblage dominated by ceanothus (*Ceanothus crassifolius*), chamise (*Adenostoma fasciculatum*), and scrub oak (*Quercus berberidifolia*) (Dunn and others 1988).

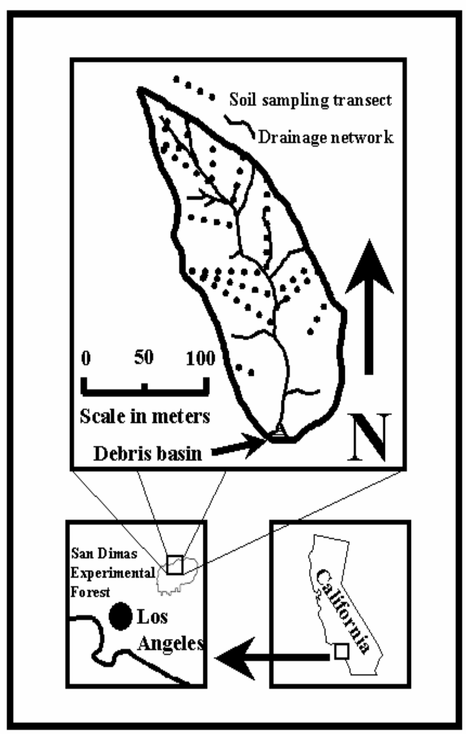

Figure 1–Location map of the study watershed showing the soil sampling transects.

Methods

To determine soil properties, we chose eight transects that crossed the watershed in a chevron pattern (*fig. 1*). Six to eight sample locations were randomly chosen along each transect, resulting in a total of 54 sampling sites. The soil characteristics were measured at the same locations at a depth of 0 to 5 cm both before and after the fire. Particle size distribution was determined for the mineral soil by the pipette method using samples that were sieved to remove coarse fragments greater than 2 mm in diameter (Gee and Bauder 1986). Percent gravel was determined by weighing the portion remaining in a 2 mm sieve. Soil bulk density was determined by the core method (Blake and Hartge 1986). Subsequent determinations of the volume and mass of rock fragments greater than 2 mm were subtracted from the aggregate samples to yield a corrected bulk density (Vincent and Chadwick 1994). Porosity was calculated from bulk density assuming a particle density of 2.65 g cm^{-3} (Danielson and Sutherland 1986). Water content measurements were made gravimetrically on soil samples taken at 0 to 3 cm depth (Gardner 1986).

An average duff and litter depth was determined using the 4-corners and 1 mid-point of a 50 cm x 50 cm square at the 54 locations throughout the watershed. Litter and duff reduction was determined using steel pins (20 cm in length) that were installed flush with the duff surface at 105 sampling points. The distance of the head of the pin protruding above the mineral soil after fire was measured to determine duff and litter consumption. Soil non-wettability was determined by the water drop penetration test (DeBano 1981) at the same 105 sampling points.

We measured hillslope sediment fluxes in 30 cm aperture collector traps on unbounded plots (Wells and Wohlgemuth 1982). Seventy-five traps scattered throughout the watershed quantified both hillslope erosion and sediment delivery to the stream channels. Hillslope sediment fluxes are measured as the air-dried mass of collected debris per unit width of slope contour (kg m^{-1}). We measured watershed sediment yield in a debris basin behind an earthen dam. Repeated sag tape surveys at monumented cross sections recorded the deposition of sediment flushed out of the small watershed (Ray and Megahan 1978). Sediment fluxes were measured for seven years prior to the fire and through the first post-fire winter.

Results and Discussion

Prescribed Burning

The prescribed burning occurred on May 15, 2001, and the fire burned for 1 to 3 hours in the target watershed. Burn severity was estimated qualitatively from post-fire fuel size diameter, color of ash, and degree of litter and duff consumption (Wells and others 1979). The fire consumed the chaparral vegetation in a period of less than 30 minutes, leaving only charred skeletal remains. Residual chamise fuel tip diameters were less than 6 mm on the site, indicating a uniform burn with light severity in the aboveground foliage (Wells and others 1979). However, ground fuels continued smoldering for several hours, consuming considerable amounts of litter and duff, indicative of a fire of moderate to high severity. Thus, a burn that appears to be of light severity in the vegetative canopy may result in a burn of greater severity at the soil surface.

Soil Properties

Soil properties from before and after the fire are shown in *figure 2*. Results of particle size analyses indicate an increase in the silt fraction at the expense of both sand and clay. This suggests that sand-sized aggregates are breaking down, perhaps as organic material is incinerated. Heating to 250° C can decrease soil organic matter content by 30 percent (Giovannini and others 2001). The 39 percent reduction in the clay fraction coupled with the increase in silt indicates that there was some aggregation of fines into silt-sized particles (Ulery and Graham 1993). Soil heating to 250° C can be sufficient to aggregate clay particles to silt and sand sizes (Giovannini and others 2001). The increase in percent gravel may be related to the post-fire release of greater than 2 mm material from the litter and duff layers, which then becomes intermixed in the disturbed surface soil.

The fire reduced the median litter depth across the watershed from 22 cm to 3 cm. In many cases the litter and duff layers were totally consumed in the smoldering ground fire. Areas where litter remained after the fire consisted of an amalgamation of ash, char, and loose mineral soil. The patterns of duff consumption reflected the species composition of the pre-fire vegetation.

Bulk density increased by 27 percent after the fire, leading to a similar decrease in calculated porosity. This suggests that the combustion of low-density organic material was not confined to the litter and duff layer, but extended into the mineral soil as well. Not surprisingly, soil moisture was reduced by 69 percent after the burn, attesting to the heat and residence time of the ground fire. Soil non-wettability is described in detail in an accompanying paper (Hubbert and others, this volume).

Sediment Fluxes

Results from the hillslope collector traps are displayed as cumulative totals (*fig. 3*). The number, order, and magnitude of storm events that produced more then 5 cm of rain are shown on the same graph. Activity levels or erosion rates correspond to the slope of the line graph. Prior to the fire, sediment flux was fairly constant, with slightly greater wet season erosion than dry season erosion (*fig. 3*). However, post-fire hillslope erosion was twice as great as pre-burn levels during the dry season, roughly equivalent to the pre-fire wet season rate. Moreover, during the initial post-fire wet season, erosion increased by 10-fold compared to pre-fire wet season levels, despite a record drought year. These results agree with previous studies of post-fire hillslope erosion in southern California chaparral ecosystems (Wohlgemuth and others 1998). During and immediately after the fire, erosion increases on these steep hillsides as surface organic barriers within the litter and duff layers are consumed, liberating the sediment trapped behind them. Erosion rates remain elevated, as the burned landscape with altered soil properties is sensitive to even minor disturbances (animal traffic, micro-earthquakes, etc.). With the onset of winter rains, a second flush of post-fire erosion occurs. This wet season erosion is enhanced by the lack of protective vegetation and organic litter, and the presence of non-wettable soils that can produce extensive overland flow.

Results from the debris basin surveys, along with the rain events, are displayed in *figure 4*. Prior to the fire, sediment yield was variable and did not always correspond to storm rainfall (*fig. 4*). However, first-year post-fire sediment yield was 20 times greater than the unburned annual average and 200 times greater than the annual median, despite a record drought year. This is similar to results from larger watersheds from the same general area (Rowe and others 1954).

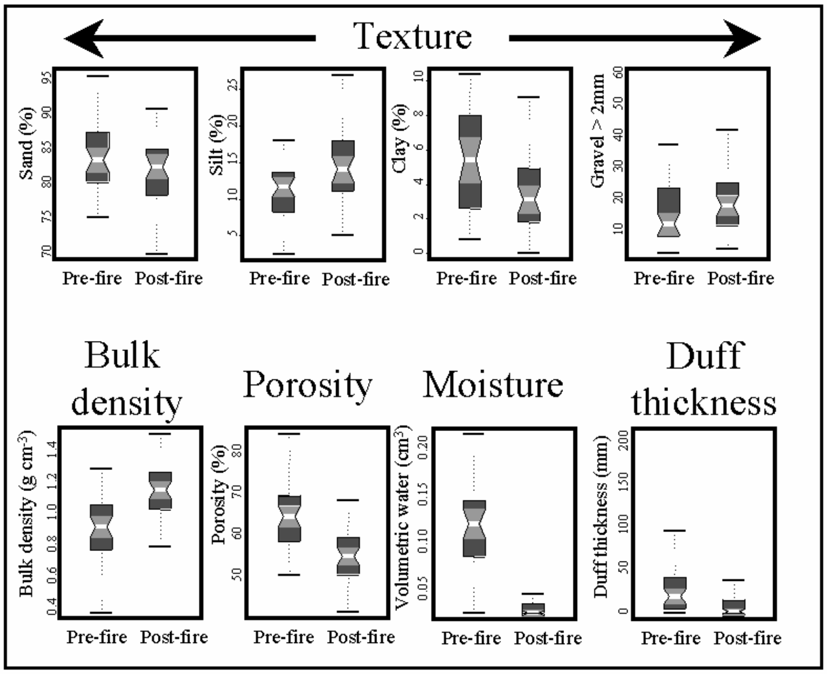

Figure 2–Selected soil properties from before and after the fire. White bar in middle of the box represents the median, with the notch representing the approximate 95 percent confidence band. Solid body of the box indicates the interquartile range. Lines outside the box show the range.

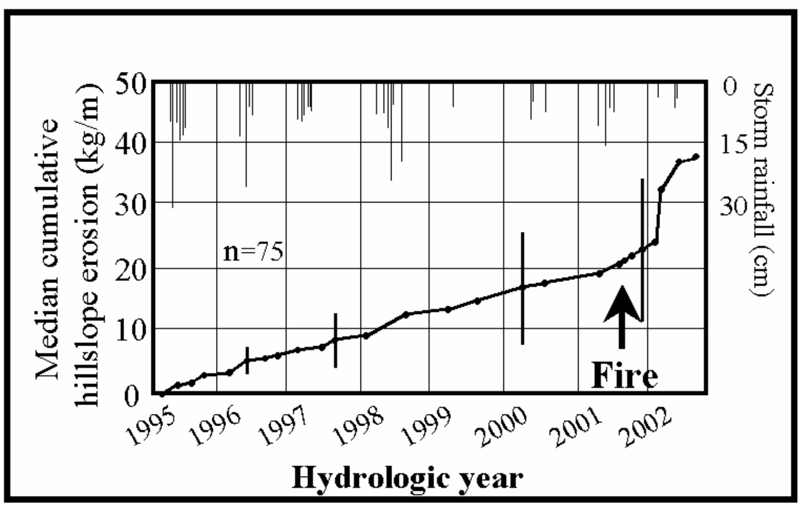

Figure 3–Cumulative median hillslope erosion and storm rainfall over time. The heavy vertical lines show the interquartile range of selected data points.

Sediment yield is a spatially integrated measure of erosion across a watershed unit. Although sediment yield is driven exclusively by surface runoff in the stream channels, it also reflects sediment delivery from the hillsides, the amount of sediment previously stored in the channels, and the time since the last channel-flushing event. The prescribed fire occurred in the midst of a four yr drought, culminating in the first post-fire winter as the driest season in 70 yr of record (Dunn and others 1988). Thus, while there was ample sediment stored in the stream channels that was subsequently mobilized by the initial post-fire storm events, the results were undoubtedly muted compared to the response that would have been experienced with normal rainfall patterns.

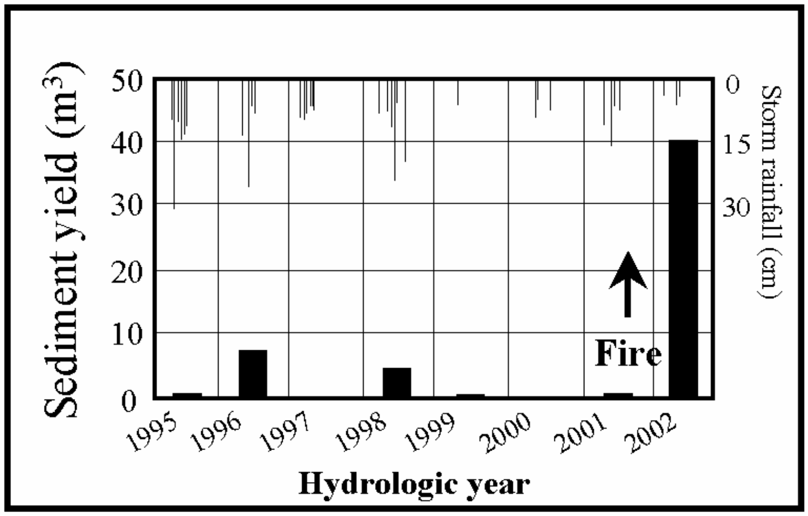

Figure 4–Sediment yield and storm rainfall over time for the 1.28 ha watershed.

Conclusions

Fire is a major disturbance event in chaparral ecosystems. Fire alters soil physical characteristics, rendering them denser, coarser, and much drier. The litter and duff layer is greatly reduced, and in many areas may be totally consumed. Although the sediment fluxes in the study watershed increased by an order of magnitude after the prescribed burn, the response was no doubt subdued compared to a wildfire followed by average rainfall. We believe the fire-induced changes in soil properties coupled with reduced ground cover caused an increase in surface runoff that accounts for the large increases in post-fire sediment fluxes.

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Geographic Variation in Mixed-Conifer Forest Fire Regimes in California[1]

R. Matthew Beaty[2] and Alan H. Taylor[3]

Abstract

This paper reviews recent research from California on geographic variability in mixed conifer (MC) forest fire regimes. MC forests are typically described as having experienced primarily frequent, low to moderate severity burns prior to fire suppression that created a mosaic of vegetation patches with variable structure. Research throughout California generally supports this view, but recent research demonstrates that fire regimes and vegetation patterns in the MC zone were more complex and displayed significant spatial and temporal variability at landscape and regional scales. At the landscape scale, patterns of fire severity and fire return intervals varied with forest composition and environmental setting (e.g., fire frequency generally decreases from south to north facing slopes). There were also apparent regional differences in some fire regime parameters (e.g., season of burn, fire severity) and not in others (e.g., fire return interval). The degree of geographic variation in MC forest fire regimes suggests that researchers should be cautious when extrapolating relationships between fire regimes and forest structure that are observed in one area to other locations.

Introduction

The 1996 Sierra Nevada Ecosystem Project (SNEP) identified the lack of empirical data on the spatial and temporal variability of fire regime parameters as a key knowledge gap in understanding the dynamics of Sierran ecosystems (Skinner and Chang 1996). Geographical variation in disturbance regimes is known to contribute to regionally distinct vegetation patterns in several widespread forest types (e.g., Spies and Franklin 1989, Veblen and others 1992, Shinneman and Baker 1997, Taylor and Skinner 1998). Yet few studies from forests in California quantify spatial and temporal variability in disturbance regimes, identify the factors that control them, or describe how this variation influences vegetation patterns and dynamics. Determining how fire regime characteristics vary is important for understanding the role of fire in the long-term dynamics of forested ecosystems and is essential for developing strategies to manage and restore fire-prone landscapes. This paper focuses on recent research that begins to address these issues for mixed conifer forests in California. We begin by summarizing the literature on the fire history and ecology of mixed conifer forests and then review the results of recent research from the central Sierra Nevada, southern Cascades, and Klamath Mountains that addresses landscape and regional scale variation in mixed conifer forest fire regimes. We conclude with some of the implications of this recent research for the management of mixed conifer forests.

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Mixed conifer forests

Distribution and characteristics

Mixed conifer forests cover 1.6 million ha (Franklin and Fites-Kaufman 1996) of the mid-montane zone (900 to 2,200 m) in the Klamath, Sierra Nevada, Cascade, and Transverse Ranges of California (Barbour and Minnich 2000). Mixed conifer forests are compositionally and structurally diverse and may be dominated by up to six conifer species including ponderosa pine (*Pinus ponderosa* Dougl.), Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirb.) Franco), incense cedar (*Calocedrus decurrens* Torr.), sugar pine (*Pinus lambertiana* Dougl.), Jeffrey pine (*Pinus jeffreyi* Grev. & Balf. in A. Murr), and white fir (*Abies concolor* Gord. and Glend.). Species composition varies depending on site conditions, latitude, and stand history (Parker 1995, Barbour and Minnich 2000). Patches of montane chaparral often interrupt tree cover in mixed conifer forests and occupy sites that have experienced severe fire or are too poor to support trees (Wilken 1967, Weatherspoon 1987, Rundel and others 1988, Bolsinger 1989). Mixed conifer forests have undergone dramatic compositional and structural changes due to land-use changes (e.g., logging, livestock grazing, fire-suppression management) associated with EuroAmerican settlement.

Ecology and fire regimes

Much of our understanding of the role of fire in mixed conifer forests comes from stand level research conducted in the southern Sierras (Skinner and Chang 1996). These studies suggest that mixed conifer community dynamics prior to EuroAmerican settlement were controlled by feedbacks between vegetation structure, fuels production, and the frequency and severity of burns (e.g., Bonnicksen and Stone 1981, 1982). In these forests, low intensity surface fires occurred frequently (e.g., median fire return interval [MFI]=3 to 20 yr) and created a fine-grained mosaic of multi-aged stands (Kilgore and Taylor 1979, Swetnam 1993, Skinner and Chang 1996). The spatial pattern of burning was controlled by the time required for fuels to build up in a previously burned patch (e.g., van Wagtendonk 1995, Miller and Urban 1999, Minnich and others 2000), and the forest pattern resulting from this process has been described as a shifting mosaic steady-state (e.g., Bormann and Likens 1979) at the landscape scale (Bonnicksen and Stone 1981, 1982).

There is considerable spatial variability in forest composition and structure at landscape and regional scales in the mixed conifer zone that may not have resulted from feedbacks between fire regimes and forest structure. The relationship between fire and forest structure described for mixed conifer forests in the southern Sierras is based primarily on stand level data. At landscape and regional scales, fire regimes may also have been strongly influenced by topographic variation (e.g., Taylor and Skinner 1998, Taylor 2000, Beaty and Taylor 2001), extreme weather (e.g. Miller and Urban 1999, Bekker and Taylor 2001) or climate variability (e.g., Swetnam 1993). As a result, interactions between fire regimes and forest structure may have been more complex and less stable than previously described, and a non-equilibrium explanation (e.g., Botkin 1990, Sprugel 1991) of mixed conifer forest dynamics may be more appropriate. The remainder of this paper illustrates this point by describing some of the ways fire regime parameters varied historically at landscape and regional scales in relation to variation in environmental factors.

Variation within the mixed conifer zone

Landscape scale variation

Researchers are beginning to recognize and quantify the influence that topography has on disturbance regimes at the landscape scale (e.g., Harmon 1982, Foster and Boose 1992, Kulakowski and Veblen 2002). In mixed conifer forests, several studies have demonstrated that fire return intervals (FRI) varied with slope aspect and elevation (Kilgore and Taylor 1979, Caprio and Swetnam 1995, Fites-Kaufman 1997, Taylor 2000, Beaty and Taylor 2001, Bekker and Taylor 2001). There was a general pattern of lengthening fire return intervals from south to north facing slopes and from low to high elevations. This spatial variation in FRI may have been due to factors related to slope aspect and elevation that affect species composition and the flammability of fuels. In the Sierra Nevada and southern Cascades, species distribution and abundance patterns at the landscape scale are strongly controlled by elevation and slope aspect (Parker 1994, Parker 1995, Bekker and Taylor 2001). Generally, there is increased representation of fir versus pine along a gradient from south to north aspects and from low to high elevations. This variation in species composition has implications for landscape scale patterns of burning. Forest litter from long-needled species (i.e. ponderosa pine, sugar pine) is less dense than that from short-needled species (i.e. Douglas-fir, white fir, red fir) (e.g., Albini 1976, van Wagtendonk 1998), and fire intensity and spread are greater in lower density fuel beds (Albini 1976, Rothermel 1983, Fonda and others 1998). Additionally, on south facing slopes fuel moisture is lower for a longer period each year than on north facing slopes increasing the likelihood of an ignition developing into a fire (Agee 1993). Fires may also burn again sooner in pine dominated forests since the production of fine fuel is greater in pine versus fir dominated mixed conifer forests (e.g., Agee and others 1978, Stohlgren 1988).

Variation in fire severity has been identified as an important source of structural diversity in forested landscapes. In mixed conifer forests, fires have been described as being primarily low and moderate severity events (e.g., Kilgore and Taylor 1979, Bonnicksen and Stone 1982, Skinner and Chang 1996). Recent research from the southern Cascades (Beaty and Taylor 2001, Bekker and Taylor 2001), Klamath Mountains (Taylor and Skinner 1998), and central Sierra Nevada (Nagel 2002) demonstrates that high severity burns were an intrinsic part of the fire regime and that landscape scale fire severity patterns were also strongly influenced by topography. In these studies, the authors used fire scar dendrochronology, stand structural analysis, and repeat historical aerial photography to reconstruct cumulative patterns of fire severity. They found a gradient of increasing fire severity with increasing topographic position, and fire severity was generally high on upper slopes, low on lower slopes and intermediate on middle slopes (Taylor and Skinner 1998, Beaty and Taylor 2001). Higher fire induced tree mortality at mid and upper slope positions may have occurred due to higher fire line intensities at these positions. Higher fire line intensities are driven by the presence of fuels dried by greater exposure to wind and solar radiation and by the preheating of fuels that occurs when a fire moves up a slope (e.g., Rothermel 1983). This topographically controlled fire severity gradient indicates that interactions between topography and fire behavior can promote highly variable structures at landscape scales in mixed conifer forests.

Topography also influenced the timing of fire occurrence by promoting discontinuous fuels and limiting fire spread. In a study of a 2,325 ha mixed conifer forest in the Klamath Mountains, Taylor and Skinner (2003) found that patterns of fire occurrence were strongly influenced by riparian areas, changes in slope aspect, and rock outcrops that controlled fire spread. Fire occurrence groups had similar fire return intervals, but fires tended to occur in different years (Taylor and Skinner 2003). Differences between groups were not related to other factors such as species composition or differences in environmental parameters. The authors concluded that topographically controlled fire breaks act as filters that reduce the spread of fires, except in extremely dry years.

The previous examples emphasize the important influence of topography at the landscape scale on fire regime characteristics in forests on highly contrasting terrain. It is important to note that topographic controls may have been weaker on less contrasting terrain. For example, on flat terrain burn severity may have been more strongly associated with the strength and direction of prevailing winds than with variation in topography. Such differences may also have had important influences on compositional and structural patterns within mixed conifer forests at the landscape scale.

Regional scale variation

Within the mixed conifer zone, some fire regime parameters varied, while others were similar across stands. The range of values for fire return interval estimates was similar for the pre-fire suppression period in mixed conifer forests in the southern Cascades (Taylor 2000, Beaty and Taylor 2001, Bekker and Taylor 2001, Norman and Taylor 2002), the Klamath Mountains (Taylor and Skinner 1998, Taylor and Skinner 2003), the Sierra Nevada (Kilgore and Taylor 1979, Caprio and Swetnam 1995, Fites-Kaufman 1997), the San Bernardino range (McBride and Lavin 1976), and the Sierra San Pedro Mártir in Baja California (Burk unpublished in Savage 1997, Stephens and others 2003). These studies all indicate that frequent fire was important in the dynamics of mixed conifer forests, yet they also suggest that variation in fire return intervals may not have been as important as variation in other fire regime parameters in explaining compositional and structural differences across mixed conifer stands.

Fire history research throughout California suggests there may have been considerable variability in season of burn within mixed conifer forests. The season of fire occurrence has been reconstructed in dendroecological studies based on the position of fire scars in annual growth rings (e.g., Caprio and Swetnam 1995). In the Sierra San Pedro Mártir in Baja California nearly all fires in mixed conifer forests burned during the growing season (Stephens and others 2003), while in the central Sierra Nevada, 50 percent of the fires occurred near the end of the growing season and only 30 percent occurred in the dormant season (Caprio and Swetnam 1995). Fire seasonality in the Klamath Mountains (e.g., Taylor and Skinner 2003) and southern Cascades (e.g., Beaty and Taylor 2001, Norman and Taylor 2002) was different than that reported for the central Sierra Nevada. In these more northern forests, fires occurred mainly (75 to 90 percent) in the dormant season. These geographical differences may have been related to differences in the onset of summer drought along the Sierra Nevada-Cascade axis (e.g., Major 1977, Parker 1994, Skinner 2002). Drought may have occurred earlier in southern mixed conifer forests than in northern forests and this may have influenced the length of time fuels were dry enough to burn each year. The dominant season of burning has a strong influence on a species' response to fire (e.g., Kauffman and Martin 1989, Kauffman 1990), and regional variation in fire season may lead to distinct vegetation responses to fire.

There may also have been a regional gradient in fire severity between northern and southern mixed conifer forests. In the southern Cascades (Beaty and Taylor 2001, Bekker and Taylor 2001), Klamaths (Taylor and Skinner 1998), and central Sierra Nevada (Nagel 2002), topographically controlled (see previous section), high severity burns that generated coarse-grain patterns of even aged forest stands were an intrinsic part of mixed conifer forest fire regimes. This contrasts with typical descriptions of fires in mixed conifer forests as having been primarily low and moderate in severity (Kilgore and Taylor 1979, Bonnicksen and Stone 1982, Skinner and Chang 1996). Few landscape scale studies, however, reconstruct fire severity patterns. Thus, additional research is needed to evaluate whether high severity fires were indeed uncommon in southern mixed conifer forests.

Conclusions

Understanding spatial and temporal variability in fire regime parameters is essential for developing strategies for managing and restoring fire-prone landscapes (Skinner and Chang 1996). Recent research from mixed conifer forests in California demonstrates that there was significant variability in fire regime parameters at both landscape and regional scales. At landscape scales, patterns of fire occurrence, severity and timing were influenced by topography and climatic variability. At regional scales, some fire regime parameters were similar (e.g., fire return interval), while other parameters may have varied substantially (e.g., season of burn, fire severity). Variable disturbance regimes in mixed conifer forests highlight the importance of incorporating "site-specific" knowledge of past fire and vegetation dynamics into management plans and suggest that researchers should be cautious when extrapolating relationships between fire regimes and forest structure that are observed in one area to other locations.

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Response of Selected Plants to Fire on White Sands Missile Range, New Mexico[1]

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Abstract

Little was known about the ecology, impacts, effects, and history of fire related to many plants and communities within White Sands Missile Range. I began by identifying the known aspects and the gaps in knowledge for White Sands Missile Range. I analyzed existing data available for the Installation taken from the Integrated Training and Area Management (ITAM) program for 1988 to 1999. Burn plots were identified at 34 sites with fires occurring sometime within that 11 yr span. Selected plant species were analyzed to identify the response to fire including change in frequency, cover, and structure. Analysis of data indicated varied responses to fire and identified a need for long term monitoring to account for natural variability.

Introduction

Fire is a major factor influencing the ecology, evolution, and biogeography of many vegetation communities (Humphrey 1974, Ford and McPherson 1996). In the Southwest, semi-desert grasslands and shrublands have evolved with fires caused by lightning strikes (Pyne 1982, Betancourt and others 1990). Fires have maintained grasslands by reducing invading shrubs (Valentine 1971). The impact these fires have on the ecosystem depends not only on current biological and physical environment but also on past land use patterns (Ford and McPherson 1996). Fires impact communities by affecting species diversity, persistence, opportunistic invading species, insects, diseases, and herbivores. Plant species diversity often increases after fires and some communities are dependent on fire to maintain their structure (Jacoby 1998). Savannas often change to a woodier community when fire is suppressed (Jacoby 1998).

The effects of fire on semi-desert grasslands and shrub invasion have been debated. Buffington and Herbel (1965) did not consider fire a main factor, though others have (Bock and Bock 1988, Pyne 1992, Ford and McPherson 1996). Many have stated that a combination of factors including fire suppression, competition, drought, grazing, rodents, and rabbits have played a significant role in the impact of fires on these communities (Branscomb 1956, Humphrey 1958). A change in fire frequency and intensity has occurred in the past 80 yr and has been correlated to grazing (Baisan and Swetnam 1997). Grazing impacts areas by reducing the fine fuel composition. Factors associated with grazing are an increase in rodent activity, increased erosion, increased seed dispersal by livestock, increased seed source of trees and shrubs, and reduced competition from perennial grasses (National Wildfire Coordinating Group 1984).

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Little was known about the fire ecology of many plants and communities within White Sands Missile Range (WSMR) and the deserts of the southwestern United States. This research identified known aspects and gaps in knowledge for White Sands Missile Range.

Study Area

White Sands Missile Range covers over 828,800 ha (2,210,117 ac) in south central New Mexico. Terrain includes steep, rocky, mountains; steep to moderate footslopes, level to rolling grasslands, dunes, lava flows, and salt flats. Vegetation communities include Chihuahuan desert scrub, Chihuahuan desert grasslands, montane shrublands, juniper savannah, pinyon-juniper woodlands, and ponderosa pine woodlands. The maximum elevation is 2,783 m at Salinas Peak. Rainfall typically occurs during the summer (July, August and September) as short, intense, localized storm events that account for 55 to 64 percent of the total annual precipitation (Barlow and others 1983). The intensity of rains generally results in massive runoff and very little infiltration especially in steep areas and areas with exposed bedrock. Average precipitation varies from 200 mm in the basin to 400 mm in the mountains.

The major fire season for southern New Mexico occurs from May to early July when temperatures are high and relative humidity is low (Kaufman and others 1998). Records kept by the Lincoln National Forest from 1964 to 1994 indicate that 60 percent of the fires started by lightning occurred during April to June and 28 percent occurred during July to September (Kaufman and others 1998). Barrows (1978) identified high starts in June (60 percent), July (18 percent), and May (17 percent). Although the highest rate of lightning ignition occurs at the end of July, the relative humidity and fuel moistures are high enough to limit fire activity. Most fires (60 percent) that have been documented on WSMR were human caused (201), 20 percent (84) were unknown causes, and 20 percent (75) were from lightning strikes. This is a typical Southwestern fire occurrence pattern (Kaib and others 1996, Kaufman and others 1998). July had the most fires reported with 84, followed by June with 75, and May with 60. Other months are high for the area in comparison to Kaib and others (1996) and Kaufman and others (1998) and are dominated by human caused fires.

Methods

Vegetation transect data were obtained from the Integrated Training Area Management (ITAM) program at WSMR. These data included pre and post-fire data for 34 sites. Data were collected by belt transects and Daubenmire quadrats. Belt transects were 100 m long and 2 m wide. Sites were read generally once a year with the number and height class of each species documented. Ten Daubenmire quadrats were placed at 10 m increments along each belt transect. These 34 sites were identified by WSMR ITAM personnel as having had fires sometime during 1988 to 1999. In some cases the date of the fire was known, but on most plots only presence of a fire in the past was recorded.

Daubenmire quadrats were analyzed in an analysis of variance (ANOVA) comparing species cover within the same plots over time. Sites with pre and postfire data were analyzed to determine if cover changed significantly. Significance was determined by the 90 percent confidence interval. Daubenmire quadrats were added in 1996, and only 21 sites had sufficient data to analyze. Belt transects were analyzed by comparing prefire mean density with postfire mean density.

Description of analysis is based on change in frequency, cover, or structure. A positive response was identified by one of these factors increasing. A negative response was based on a decrease. Structure was identified by categorical class and discussion assumed an increase in size structure was a positive response.

Results and Discussion

Analysis of Daubenmire quadrats (*table 1*) and belt transects (*table 2*) produced varied responses of plants to fire. Some trends were found, but confounding factors such as differing methodologies, plot surveys, and localized differences in precipitation make it difficult to form major conclusions.

Table 1—*Yearly mean cover (standard error) of Daubenmire quadrats containing each species with results of analysis of variance including significant F values for Land Condition Trend Analysis plots on White Sands Missile Range, New Mexico. Each plot consisted of 10 quadrats. Fires occurred between 1996 and 1997.*

No. of plots	Species	1996	1997	1998	F	P
6	blue grama (*Bouteloua gracilis*)	33.4 (12.7)	56.5 (14.5)	55.1 (12.9)	6.016	0.028
3	hairy grama (*Bouteloua hirsuta*)	36.7 (24.0)	a	52.21 (7.2)	0.50	0.53
1	black grama (*Bouteloua eriopoda*)	36.9 (34.7)	73.4 (12.8)	a	13.6	0.001
1	Snakeweed (*Gutierrezia sarothrea*)	7.5 (6.2)	51.8 (21.0)	a	32.78	<0.001
1	alkali sacaton (*Sporobolus airoides*)	37.8 (32.5)	a	20.8 (15.8)	3.962	0.055
1	Tobosa (*Pleuraphis mutica*)	22.3 (7.7)	41.8 (6.3)	a	5.672	0.035
1	Wolftail (*Lycurus setosus*)	18.3 (3.3)	a	36.7 (6.3)	3.484	0.089
1	Wolftail (*Lycurus setosus*)	26.5 (5.1)	50.5 (7.2)	a	7.796	0.023
1	Creosotebush (*Larrea tridentata*)	33.4 (4.6)	a	17.2 (5.7)	4.896	0.058

[a] indicates no data available.

The grama grasses were positively impacted by fire. Blue grama increased significantly on all six plots after fire. This is consistent with other findings that blue grama is positively impacted by fire (Wright and Bailey 1980, Ahlstrand 1982). Black grama increased significantly on the only plot analyzed. This contradicts some previous studies indicating a negative response to burning (Buffington and Herbel 1965). The impact of fire on black grama depends on various burn and climatic conditions, and as with many plant species, generalizations regarding impacts must be made with care. Hairy grama was identified as having a positive response on two plots and a negative response on one plot. Location variation may be the reason for the differences observed, as well as the timing of the burns involved. The two positive response plots were burned in March and burned under much cooler conditions then the one negative response. Additionally, data for the negative response were from 2 to 4 yr post-fire. There may have been an initial increase during the first 2 yr and then a decline. Other studies have documented hairy grama having variable responses to fire based on drought or wet years (Wright and Bailey 1980, Ahlstrand 1982). Unfortunately there were no precipitation data with specificity for this site.

Table 2—*List of plant species and number of plots with positive or negative responses on plots after analysis of belt transects on White Sands Missile Range, New Mexico. Response was defined as an increase (positive) or decrease (negative) in frequency within a plot.*

Common name/Scientific name	Belt transect		
	Response		Change in structure (shrub height)
	Positive	Negative	
Apache plume (*Fallugia paradoxa*)	2		
banana yucca (*Yucca Bacata*)		5	
creosotebush (*Larrea tridentata*)		1	1
desert rose (*Rosa stellata*)		1	
plains prickly pear (*Opuntia phaeacantha*)	3	2	
false indigobush (*Dalea formosa*)	2	2	
four-wing saltbush (*Atriplex canescens*)	3	2	5
mesquite (*P. glandulosa*)		2	
New Mexico agave		1	
pineapple cactus (*Neollydia intertexta*)	2		
purple prickly pear (*Opuntia macrocentra*)		4	
sand sage (*Artemisia filifolia*)		1	
skunkbush (*Rhus trilobata*)	1	2	
soaptree yucca (*Yucca elata*)	2	4	4
sotol (*Dasylirion wheelerii*)	1	1	
tarbush (*Flourensia cernua*)	1	1	2
winterfat (*Krascheninnikovia lanata*)		2	

Alkali sacaton has been identified as being positively impacted by fire (Wright and Bailey 1980), but the one plot analyzed here had a negative response. This site is located near the eastern boundary of WSMR. Sacaton recovers in 2 to 4 yr to pre-fire conditions (Wright and Bailey 1980), and the plot surveys analyzed here provided only 2 yr post fire data. This negative response may be the short term impact. Timing of the fire may also be a factor, as burns in spring or fall can be more detrimental to this species than burns during the natural regime of May through September (Cox and Morton 1986).

Tobosa increased significantly at one site based on cover in a one year period after fire. This is similar to findings from Britton and Steuter (1983), who found that, depending on the precipitation, tabosa can revegetate quickly and biomass can increase 2 to 3 times.

Creosotebush decreased at one site. This corresponds with the response observed by Brown and Minnich (1986) in the Sonoran Desert, where recovery of creosotebush was quick and density was similar to pre-fire years in the second post-fire year. On three sites, there was neither a positive nor negative response. On one site, the structure of the creosotebush community changed as larger plants (1 to 2 m) were found in higher frequencies after the fire.

Apache plume responded positively to fires on 2 plots. Banana yucca (5 plots), redjoint prickly pear (2 plots), and pineapple cactus (2 plots) were negatively impacted by the fires. However, Worthington and Corral (1987) identified no mortality of banana yucca after the fire in the Franklin Mountains.

Many species had varied impacts, with a combination of positive, neutral, or negative response being associated with different plots. These include plains prickly pear, false indigobush, mesquite, New Mexico agave, sotol, tarbush, and four-wing saltbush. The variation of impact observed among these species may be associated with the type of fire, season of fire, fire behavior, location, and vegetative communities.

One of the difficulties in using this data set is that species are not identified as resprouting or sprouting from seed. In some cases, species such as soaptree yucca were identified as having a change in structure. This change in structure is important in the fire ecology of the system. If the change in structure is due to burning of the above surface portion of the plant and resprouting, the ecological impact is different than if that structure change includes sprouting of new propagules. This is an important distinction when trying to support fire adaptation in plants.

Fire plays a major role in many of the vegetation communities that occur on White Sands Missile Range. These communities on WSMR are structurally different from similar communities in the southwest because of the accumulation of fine fuels and high densities of closed canopy piñon woodlands. The impact of fire suppression on these fuel loads on WSMR cannot be determined, because comprehensive historical fire suppression documentation is not available. Historical reports that do exist indicate that many fires were actively attacked, but the degree and success in which those fires were observed, located, and controlled is unknown.

Analysis of these long term data suggests that sampling immediately prior to the fire and 1 year post fire does not provide an adequate identification of plant species trend. Variability was high on many plots prior to being burned. Impacts observed based on these types of analyses may reflect the variability of the system rather than the impact of the fire. I recommend that plots be sampled several years prior to fires, although management often does not have this luxury. Incorporating other long term sampling data may provide the manager information to identify the long term trends in relation to fires.

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A Neutral Model for Low-Severity Fire Regimes[1]

Don McKenzie[2] and Amy E. Hessl[3]

Abstract

Climate, topography, fuel loadings, and human activities all affect spatial and temporal patterns of fire occurrence. Because fire occurrence is a stochastic process, an understanding of baseline variability is necessary in order to identify constraints on surface fire regimes. With a suitable null, or neutral, model, characteristics of natural fire regimes estimated from fire history data can be compared to a "null hypothesis." We generated random landscapes of fire-scarred trees via a point process with sequential spatial inhibition. Random ignition points, fire sizes, and fire years were drawn from uniform and exponential family probability distributions. For this paper we focused on two sets of statistics commonly computed in fire history studies. Composite fire records and Weibull median probability intervals (WMPIs) were calculated at multiple spatial scales for random subsets of each landscape, and parameters of the Weibull distribution were estimated for each simulated "fire history" and tested for significance. We compared results from simulations to fire-history data from a watershed in eastern Washington. Strong nonlinear relationships were evident between and area sampled and WMPIs for a range of fire sizes for both real and simulated data. Patterns of significance of Weibull "shape" parameters were distinctly different between real and simulated landscapes. The clear patterns on neutral landscapes suggest that deviations from them in empirical data represent real constraints on fire regimes (e.g., topography, fuels) rather than sampling artifacts. Neutral models show promise for investigating low-severity fire regimes to separate intrinsic properties of stochastic processes from the effects of climate, fuel loadings, topography, and management.

Introduction

Fire-history reconstructions provide the empirical basis for fine- and coarse-scale modeling of fire regimes and for informed management and restoration of ecosystems (Landres and others 1999, Schmoldt and others 1999, Swetnam and others 1999, McKenzie and others 2004). Reconstructions use different methods depending on objectives, the nature of the fire regimes being studied, and the spatial and temporal scales of analysis (Clark 1990, Agee 1993, Heyerdahl and others 1995, Lynch and others 2003, Prichard 2003). Although estimates of fire frequency from different methods are often combined for modeling and management, they are not always equivalent (Heyerdahl and others 1995, McKenzie and others 2000, Li 2002).

In dry low-elevation forest ecosystems of the western United States, living trees provide a long-term record of low-severity fire from fire scars (Agee 1993, Swetnam 1993, Everett and others 2000, Veblen and others 2000, Heyerdahl and others 2001). Existing reconstructions are of varying quality; some were developed from very

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small samples; others were not crossdated against a master chronology (Madany and others 1982, Heyerdahl and others 1995). Crossdating errors can substantially change estimates of the statistical properties of fire regimes (McKenzie and others 2000). In addition, fire-history reconstructions are sensitive to the area sampled, because too small an area may miss fires entirely, whereas too large an area may record separate fire events as one (Agee 1993, Heyerdahl and others 1995, Baker and Ehle 2001). Research is needed to quantify how fire regime statistics change across spatial scales and how these changes may differ in different ecosystems (Falk and Swetnam 2004).

Neutral models

In formal experiments, in which a null hypothesis is tested against an alternative, “controls” and “treatments” are ideally identical except for the treatment effect. If this effect is statistically significant, the null hypothesis is rejected. “Neutral models” in community and landscape ecology generalize the concept of a null hypothesis to an array of patterns and processes that capture relevant details of a system but eliminate constraints or mechanisms of interest (Caswell 1976, Kimura 1983, Gardner and O’Neill 1991, Hubble 2001). When a complex stochastic process (e.g., landscape disturbance or community assembly) is being studied, identification of the appropriate neutral model is difficult, and much controversy exists in the literature (e.g., Weiher and Keddy 1999).

Neutral models have been used in landscape ecology to study species co-occurrence (Milne 1992, Palmer 1992), formation of ecotones (Milne and others 1996), metapopulation models and conservation (With 1997), and connectivity and disturbance spread (Green 1994, Keitt and others 1997). In this paper, we present a neutral model of low-severity fire regimes – those for which the fire history is preserved via fire scars on living trees. The model was designed to represent stochastic properties of fire regimes in that fire sizes, fire-free intervals, and fire locations are considered to be random variables. We examine two statistical properties of the neutral model: how estimates of fire frequency change with changing spatial scale, and how temporal trends in the hazard of burning change with area and number of trees sampled. We compare results to the same statistical properties on a real landscape, and make qualitative inferences about the usefulness of pursuing this method for identifying constraints on fire regimes.

Methods

Fire history data

The fire-history data are from a dry forest ecosystem in eastern Washington, USA. Everett and others (2000) produced a detailed, spatially explicit dataset of fire history data from 17,700 fire-scarred trees collected in five study sites (3,116 to 12,747 ha) extending from the Okanogan-Wenatchee National Forest in central Washington to the Colville National Forest in northeastern Washington (*fig. 1*). These study sites occupy a 500 km northeast to southwest gradient across the Okanogan Highlands and down the east side of the Cascade Range.

Fire-history sites are located within forests dominated by ponderosa pine (*Pinus ponderosa* Dougl. ex Loud.). These forests in Washington typically occur between 600 and 1,200 m elevation and change into Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), grand fir (*Abies grandis* [Dougl.] Lindl.), western larch (*Larix* *occidentalis* Nutt.) and lodgepole pine (*Pinus contorta* Dougl. ex Loud.) at higher elevations and grassland or sagebrush (*Artemisia tridentata* Nutt.) at lower (*fig. 2*).

This spatially distributed network of geo-referenced, cross-dated fire scar chronologies is ideal for spatial and temporal analysis of regional surface-fire history. To date, chronologies have been developed for five study sites (*fig. 1*). Within each site, all fire-scarred trees were mapped, and a spatially stratified random sample of high-quality trees (with a large number of scars) was collected.

For this paper, we focus on the 11,088-ha Swauk watershed (inset in *fig. 1*), with a total of 7,048 fire scars on 671 recorder trees. The composite fire interval for the watershed (counting only fires that scarred >10 percent of the trees) is 8 yr, and the Weibull median probability interval (WMPI) for these same fires is 7 yr (Hessl and others 2004). The frequency of occurrence of a fire somewhere in the watershed (possibly fewer than 10 percent of trees scarred), however, was just under 2 yr.

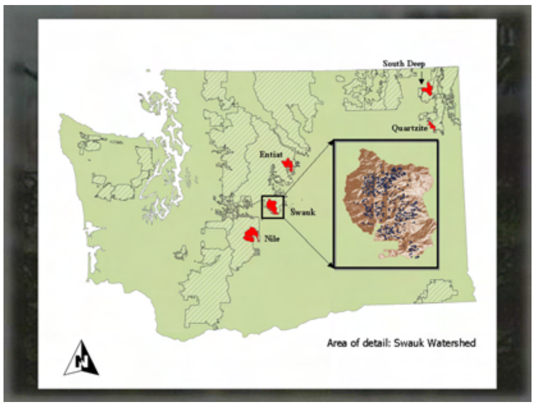

Figure 1–Location of fire-history study in eastern Washington

Figure 2–Study locations in ponderosa pine forests.

Simulation model

The “neutral” landscape consisted of a square grid with unitless X and Y coordinates, in emulation of a watershed. We randomly located 200 points to represent trees that could record fires over the course of the simulations. Sequential spatial inhibition (Ripley 1987) was used to ensure that no two trees overlapped. A 300 yr fire history was simulated for two watershed-scale mean fire frequencies (MFRIs): 2 and 5 yr. These numbers represent the average interval between the occurrence of fire somewhere in the watershed and correspond to the 2 yr value computed from the Swauk watershed. Time steps (years until the next fire) were drawn from an exponential distribution, with mean=MFRI and rounded to the nearest integer, until their sum was >300. At each time step, a circular fire was simulated with its center randomly located in the watershed. Mean fire sizes were defined within a range of proportions of the total watershed area, between 0.1 and 0.4, with steps of 0.05. These proportions were chosen to approximate proportional sizes of composite fire records in real watersheds. For each round of simulations, the size of each fire was drawn from a gamma distribution whose mean was the mean fire size.

Composite fire records

The coordinates of each recorder tree were noted and a data matrix created (300 rows=years, 200 columns=trees) to hold the "fire history" of the simulated watershed. We then simulated fire-history reconstructions within the watershed, using a "neutral" method whereby one tree was selected as a starting location. Using this tree as a center, we computed composite fire intervals (CFIs) for search radii from 0.1 to 0.6 proportions of the watershed (at intervals of 0.01–50 total) and fit each time series of fire-free intervals to the two-parameter Weibull distribution. For each search radius, 20 replicate CFIs were computed (*fig. 3*).

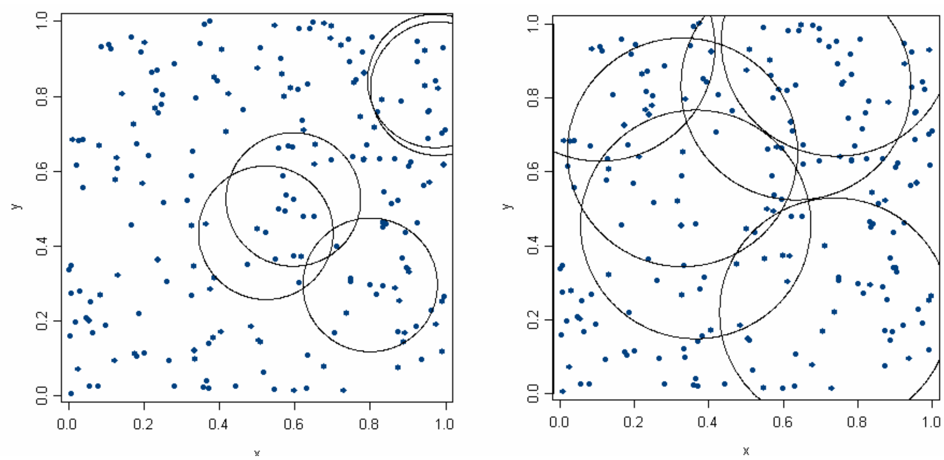

Figure 3–Simulated areas for fire-history reconstructions with search radii equal to 0.1 and 0.3 of the watershed.

Two features of the Weibull distribution make it well-suited to modeling both high- and low-severity fire regimes (Clark 1989, Johnson and Gutsell 1994, Grissino-Meyer 1999): the Weibull median probability interval (WMPI) is a robust measure of central tendency, and the shape parameter and associated hazard function allow changes in the hazard of burning over time to be identified (*figs. 4 and 5*). For each of the composite fire records (20 replicates x 50 search radii), we computed the WMPI and the Weibull shape parameter, the latter as a surrogate for the slope of the hazard function (Clark 1989, Johnson and Gutsell 1994). For each search radius, the mean of 20 replicates was stored for WMPI and the shape parameter. We applied the same iterative process to fire history for the Swauk watershed, but restricted sample years to the period 1651 to 1900. By 1650, most trees had recorded one fire. Before 1900, fire exclusion had not drastically changed fire frequency (*fig. 6*, Hessl and others 2004). We created a set of composite fire records (20 replicates x 50 search radii) similar to the simulated watershed, using a 250 yr fire record.

We compared simulated to empirical results qualitatively for changes in WMPI and shape parameters with changing spatial scale. The exploratory analysis presented here was designed to provide insight as to whether the neutral model of fire regimes is a useful concept that can be applied to inferences about the constraints (e.g., fuels, topography, land use) on low-severity fire regimes. Formal statistical tests and extrapolation to other geographic areas will be the subject of future research.

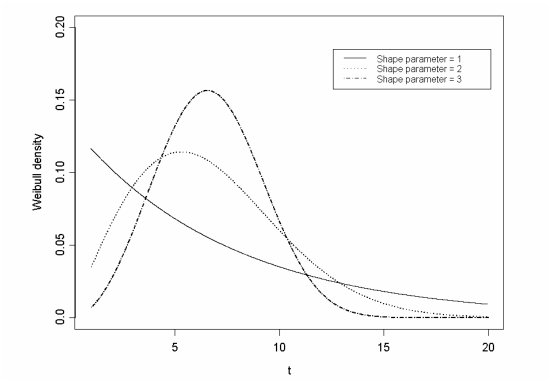

Figure 4–Shape of the Weibull density for different values of the shape parameter, with scale parameter held constant..

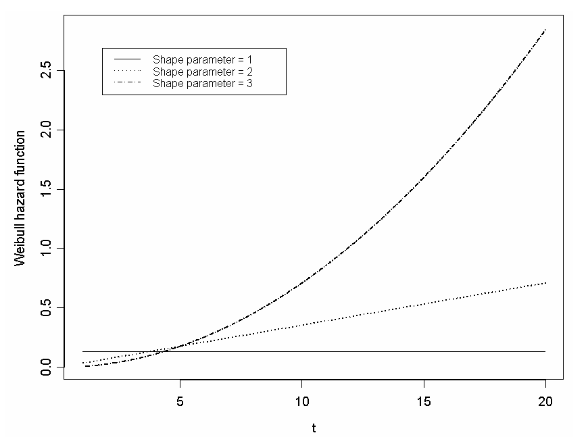

Figure 5–Shape of the Weibull hazard function for different values of the shape parameter, with scale parameter held constant..

Results and Discussion

Scaling relationships for the estimated fire-free intervals (WMPI) between simulated (neutral) and real watersheds are clearly similar (*figs. 7 and 8*). In both cases a log-linear model best predicted the relationship between the search radius, equivalent to sample area in a fire-history study, and WMPI, with R2 of 0.96 and 0.89, for simulated and real data, respectively. When the watershed-scale mean FRI was 2 yr in simulated data, point-level WMPI ranged from 9 yr, for simulated mean fire sizes of 0.15 of the watershed area, to 5 yr for a simulated size of 0.35. For watershed-scale means of 5 yr, point-level WMPI ranged between 23 yr (0.15 of watershed) down to 12 yr (0.35 of watershed). For the Swauk watershed, mean point-level WMPI was 13 yr (*fig. 8*).

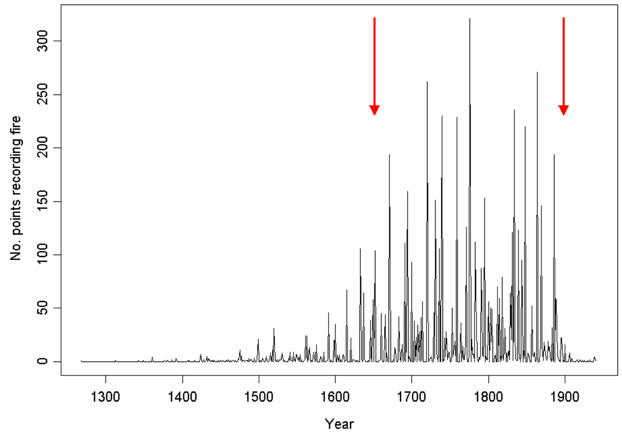

Figure 6–Fire history record in the Swauk watershed. Arrows represent the beginning and end of the period used in this paper.

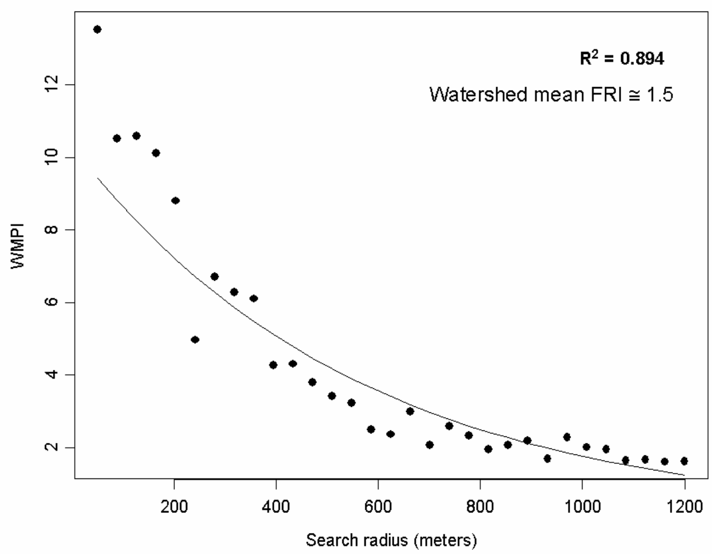

Figure 7–Relationship between area sampled and WMPI for simulated data.

Results from the neutral model suggest that non-linear scaling laws are an intrinsic property of the spatial relationships among points subject to this particular stochastic process (low-severity fire) (Falk and Swetnam 2004). The log-linear relationship holds for a variety of mean fire sizes and fire frequencies. In the composite fire record for the Swauk watershed, there are no local influences that invalidate the scaling relationship, despite the variety of fire sizes and the relatively complex, dissected topography (*fig. 1*). Given that fire regimes vary over seemingly homogeneous landscapes (e.g., Baker 1989) and that topography has been shown to provide strong constraints on fire regimes (e.g., Taylor and Skinner 2003), more research is needed in a range of low-severity fire regimes before a global scaling law can be said to exist.

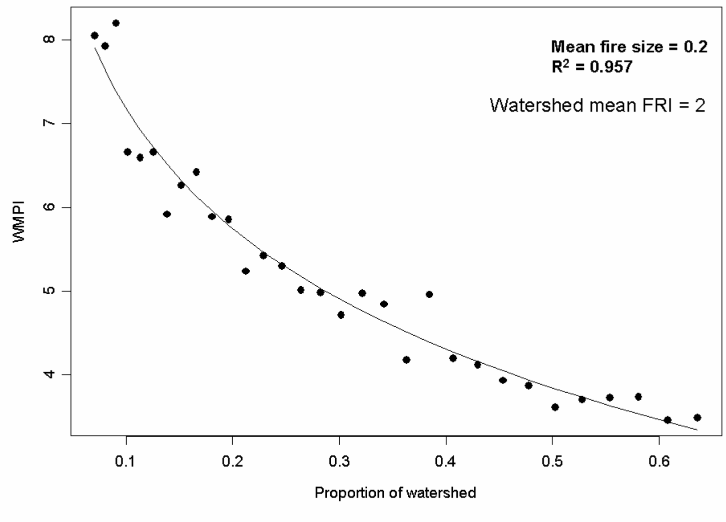

Figure 8–Relationship between area sampled and WMPI for the Swauk watershed.

Estimates of the Weibull shape parameter differed substantially between simulated and real watersheds, both in mean value and trends over increasing search radii. For simulated watersheds, mean estimates were between 1.10 and 1.45 for a variety of mean fire sizes (0.1 to 0.5 of watershed) and search radii (0.15 to 0.45) (*fig. 9*). Search radii below 0.15 contained too few fires to estimate shape parameters successfully. In contrast, shape parameters estimates in the Swauk watershed ranged between 1.5 and 2.5, with a sharp descent between the point level and a search radius of 200 m (*fig. 10*). Initial bootstrap estimates of significance for the shape parameters suggested that for a 95 percent confidence interval for both simulated and real watersheds to exclude 1.0 (corresponding to a flat hazard function), shape parameters should have a minimum between 1.3 and 1.6. A more refined bootstrap algorithm for these distributions, to more precisely compute p-values, is under development.

Changes in shape parameter are surrogates for changes in the slope of the hazard function (Clark 1989, Johnson and Gutsell 1994, *fig. 5*). The neutral model, shape parameters gradually increased with the search radius (*fig. 9*), though changes were slight and probably not statistically significant. However, similar behavior for a variety of simulated fire sizes suggests that this gradual increase may be an intrinsic

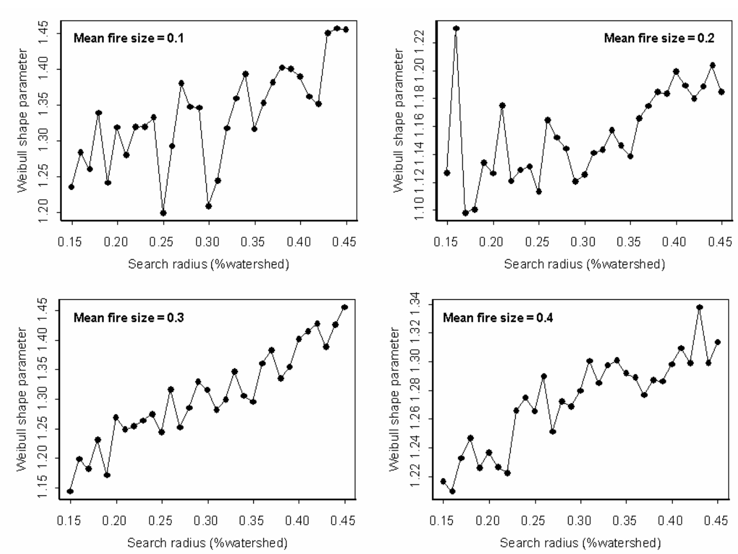

Figure 9–Trends in the Weibull shape parameter with increasing search radii on simulated landscapes with different mean fire sizes.

property of the stochastic process being modeled, and needs to be distinguished from any processes of interest when the neutral model is compared with real data.

In contrast, in the Swauk watershed the shape parameters drop very quickly with increasing spatial scale, up to 240 m search radius, then begin a gradual ascent similar to those in the neutral model, although values are higher (approximately 1.5 to 1.8 in Swauk vs. a maximum of 1.45 for neutral model). We infer that a constraint on fires in rapid succession is operating on the Swauk landscape up to, on average, an area of 18 ha (240^2 * PI / 10^4), and that the increasing hazard function over time at this scale is a reflection of it. Beyond this search radius, the shape parameter behaves very similarly to that from the neutral model. In the Swauk and similar landscapes, composite fire records from study areas of different sizes, for example 10 ha vs. 40 ha, could therefore yield very different interpretations about fire history. Baker (1989) observed that different statistical models were appropriate for different-sized study areas in a landscape with high-severity fire. Key processes controlling fire frequency may change, or at least appear different, when data are collected and analyzed at different spatial scales.

An obvious candidate for the small-scale constraint on historical fire frequency in the Swauk watershed is buildup of fine fuels after fire. Time is required for a spatially continuous fuelbed, and therefore sufficient potential fire spread for trees to record fire, to accrue. When viewed at scales much greater than 18 ha, this constraint on fire occurrence is no longer observed. The 18 ha threshold would appear to be a surrogate for a modal, or characteristic, fire size, in that it defines the spatial extent over which the constraint of fuel buildup can be observed. When the search radius becomes much greater than typical fire size, it would be expected to include multiple fires, burning different patches over time, and confounding the fuel constraint.

The threshold for change in behavior of the hazard function would clearly change with changes in characteristic fire size, but only on landscapes on which a mechanism exists that controls fire occurrence (not neutral landscapes). Constraints on fire size, then, will be another factor controlling fire frequency. Other results from this exercise—the log-linear models of WMPI as a function of search radius (and fire size)—indicate that on both neutral landscapes and the Swauk watershed, scaling relationships among all factors are clearly defined. If proven to be robust on a variety of landscapes, particularly those on which topography provides strong constraints on fire spread, these scaling relationships could be useful for predicting characteristics of fire regimes for which extensive, spatially explicit data are not available.

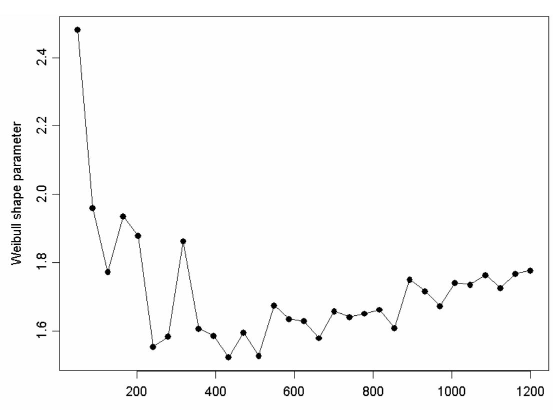

Figure 10–Trends in the Weibull shape parameter in the Swauk watershed with increasing search radii.

Conclusions

Neutral models hold promise for understand complex behavior in low-severity fire regimes. Fire is a stochastic process, of which each fire-history record is just one of many potential realizations (Lertzman and others 1998). Because fire-history reconstructions are made with finite resources, each can discover only an incomplete sample of even the one realization that it observes (Baker and Ehle 2001). A template is needed for intrinsic statistical properties of fire regimes, against which specific realizations can be compared to distinguish their individual properties (for example, constraints on fire size and frequency).

In this paper, we begin to explore the ways in which neutral models can help distinguish these individual properties of real landscapes from the "neutral" background. More research is needed as to what to include in the neutral background. For example, many fire-history reconstructions develop collector's curves to determine the first point in time at which a representative sample of the total population of recorder trees has recorded at least one fire (Hessl and others 2004). Should a neutral model incorporate an increasing population of trees over time? Similarly, many researchers include only significant fire events (greater than a certain percentage of recorder trees scarred) in models of the association between fire occurrence and fuel or climatic constraints (Swetnam and Betancourt 1990, Grissino-Meyer 1995, Heyerdahl and others 2001, Hessl and others 2004). How would neutral model behavior differ with these additional attributes? Finally, more comparisons with data from real landscapes are needed, both to test the usefulness of the concept and to identify consistent patterns, if any, in scaling relationships and departures from "neutrality" in real fire regimes.

Acknowledgments

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A Comparison of Postburn Woodpecker Foraging Use of White Fir (*Abies concolor*) and Jeffrey Pine (*Pinus jeffreyi*)[1]

Kerry L. Farris[2] and Steve Zack[3]

Abstract

We examined the temporal patterns of the structural decay, insect infestation and woodpecker foraging patterns on white-fir and yellow pine following a prescribed burn in Lassen National Park, CA. Our objectives were to: 1) describe how pine and fir differ in their decay patterns and insect activity, and 2) determine how these differences reflect woodpecker foraging habitat quality. Preliminary results indicate that these two tree species differed in several aspects of structural decomposition, insect use and subsequent woodpecker foraging intensity. White fir tended to decay more quickly and was used more intensively by both wood-boring beetles and foraging woodpeckers during the first 1 to 2 yr following the fire. In contrast, Jeffrey pine was not initially used as intensively, but continued to provide foraging resources for both insects and woodpeckers throughout the entire study period (4 yr). These results suggest that prescribed burning may help to restore ecological interactions between insects, woodpeckers, and snag decomposition critical for snag-dependant wildlife species.

Introduction

Increasingly, forests are being managed using prescribed burning to reduce fuels and restore fire as an ecosystem process where it has been excluded for decades. Monitoring the short and long-term effects of prescribed burning on the biota is necessary to assess the biological outcomes and to refine prescriptions and management goals. Comprehensive monitoring programs are in place for fuels and forest structure and composition, for example, but specific information regarding the potential response of various wildlife species and specific habitat parameters is still limited.

The response of woodpeckers to prescribed burns is of particular interest because they excavate cavities in snags that ultimately serve as nesting habitat for a wide variety of other vertebrate species (Bull and others 1997). Woodpeckers are often the most conspicuous vertebrate responding to forest fires as they prey on the large influx of beetles whose larvae feed on, pupate in, and emerge from fire-damaged trees. Additionally, woodpeckers foraging on decaying snags may influence the capacity of those snags to be excavated in the future (Farris and others 2004). Understanding the nature of woodpecker response to prescribed burns is important to forest managers.

Little is known about how woodpeckers utilize different tree species after fire or how beetle infestation and woodpecker foraging interact to affect snag decay and

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woodpecker nesting habitat. In this paper we present preliminary results of a long-term study to investigate temporal dynamics of insect infestation, woodpecker foraging, and structural decomposition of Jeffrey pine and white fir. Our objectives are to describe how pine and fir differ in their decay patterns and associated insect activity and to determine how these differences affect woodpecker foraging patterns. A secondary objective is to gain a better understanding of these interactions following spring (growing season) prescribed fires.

Materials and Methods

This study was conducted in Roadside Prescribed Burn in Lassen Volcanic National Park (LAVO), which lies in the southern Cascade Range in northeastern California (lat 40°33' N, long 121°30' W). The 47 ha burn, implemented in June of 1999, ranged in elevation from 1,725 to 1,798 m (5,660 to 5,900 ft) and was one of the few spring (growing season) prescribed burns in the park. Tree species consisted primarily of Jeffrey pine (*Pinus jeffreyi*) and white fir (*Abies concolor*), with occasional incense cedar (*Calocedrus decurrens*) (Parker 1991). Resident woodpeckers of interest include the Black-backed (*Picoides arcticus*), Hairy *(P. villosus*), and White-headed (*P. albovartus*)

Immediately following the burn we established a sampling grid of 60 points across the entire area. At each grid point the nearest Jeffrey pine and white fir were selected for annual sampling, making a total of 120 sample trees. In order to be considered for our samples, all trees had to be at least 10 cm in diameter and had to exhibit bole char but did not have to be visibly dead immediately following the fire. To assess the structural and biological changes between tree species and years, we recorded the following variables at each of 120 sample trees once per year from 1999 until 2002: status (live, dead, broken, or fallen), diameter at breast height, bole char, evidence of bark and wood boring beetle activity, and woodpecker foraging sign.

To eliminate ambiguity in determining tree death, we classified a tree as dead when all the needles had died. Evidence of beetle activity was classified as belonging to either bark or wood boring beetles, based on the size and shape of the emergence holes left on the bole of the tree. Woodpecker foraging activity was classified into two categories: "scaling," which is a superficial flaking of successive bark layers in search of primary bark beetles; and "excavating," which creates distinct holes that typically penetrate the bark and outer sapwood as the birds search for wood boring beetles. Scaling was quantified by estimating the proportion of the bole in which bark was removed by woodpeckers. The amount of excavating was quantified using binoculars, scanning from the base of the tree to the tip and counting all visible foraging excavations on a randomly selected azimuth, which remained constant throughout the study period. The surface area surveyed was calculated and divided into the number of observed foraging "hits," resulting in a standardized index of foraging hits per square meter of tree surface. This index was used to compare woodpecker foraging activity between tree species and years. The index was calculated annually as the cumulative number of foraging excavations per m^2.

The relative amount of wood decay in each snag was quantified using an IML Resistograph® (Dunster 2000). The Resistograph® is designed to detect decay and defects in trees and wooden structures. The instrument works by inserting a fine drill (approximately 3 mm in diameter) into the wood at a constant rate and recording the amount of resistance imposed by the structural condition of cell walls. We recorded

Resistograph® measurements on each of the dead trees in 3 yr after the burn in 2002. Measurements were recorded at a constant height of 1.3 m and on three separate azimuths (120°, 240°, and 360 °). These three measures were averaged to obtain a single estimate of wood decay for each snag.

We calculated yearly fall rates following Landram and others (2002), where the number of snags falling within a year is divided by the total number standing at the start of the year. We used repeated measures ANOVA (Zar 1990) to evaluate differences in woodpecker scaling and foraging intensity between tree species and years. A nonparametric Mann Whitney U Test (Zar 1990) was used to assess differences in wood quality between Jeffrey pine and white fir. All statistical tests were conducted using SPSS Version 11.0.1 (SPSS 2001).

Results

Structural Changes

Size class distribution was similar between our sampled Jeffrey pine and white fir (*table 1*). Mean diameters of all trees did not significantly differ between species ($t=-0.609$; $p=0.544$). Both mortality patterns and fall rates varied between the two species during the first three years following the fire (*fig. 1*). Sixty-eight percent of the Jeffrey pine trees died within the first year and this number increased steadily to 78 percent of the original population by year three. In contrast, 63 percent of the white fir died within the first year, but no new snags were added to this group in subsequent years (*fig. 1a*). Surviving trees were significantly larger than those that died ($t=4.319$; $p=0.000$; and *table 2*).

Table 1—*Size class distributions of sample white fir and Jeffrey pine within the Roadside Prescribed Burn, Lassen Volcanic National Park, CA.*

Diameter class	*Abies concolor*	*Pinus jeffreyi*
10-15cm	4	6
16-28cm	15	24
29-60cm	34	21
>60cm	7	9

Table 2—*Mean diameters of white fir and Jeffrey pine by condition within the Roadside Prescribed Burn, Lassen Volcanic National Park, CA.*

Species	Condition	Mean	N	SD
Abies concolor	Dead	34.50	38	15.26
	Live	40.95	22	18.38
Pinus jeffreyi	Dead	28.60	48	12.93
	Live	83.00	12	34.22

Fall rates of snags also differed by species and year (*fig. 1b*). After the first year following the fire, fall rates for both species was zero. However, after 2 years, white fir fell at a rate of 5.3 percent per year, while the rate for Jeffrey pine remained the same. Following the third year fall rates for white fir climbed to 11.1 percent while Jeffrey pine increased to 8.3 percent (*fig. 1b*). Diameters of standing snags were similar to those that fell ($F=0.725$, $p=0.397$). Three year old Jeffrey pine snags had significantly greater wood densities than the same age white fir (Z=-2.046, p=0.041). As a reference point, wood densities for both live Jeffrey pine and white fir average approximately 24.0. In contrast, mean wood densities for fir snags averaged 10.3 (± 2.26 SE; n=32) while pine snags averaged 15.5 (± 1.70 SE; n=37).

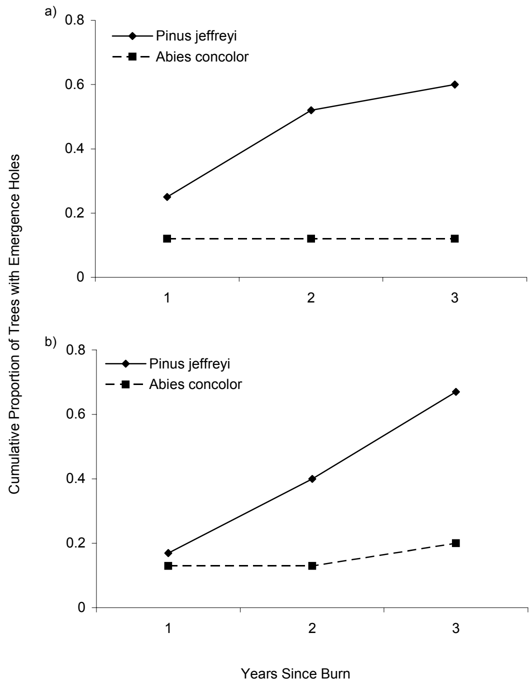

Figure 1—Recruitment (a) and fall (b) rates of Jeffrey pine and white fir following a spring prescribed burn in Lassen Volcanic National Park, CA.

Beetle and Woodpecker Use

Primary bark beetles from the family Scolytidae (insects which kill trees during the process of their reproduction) and secondary wood borers from the families Buprestidae and Cerambycidae (insects requiring dead trees to reproduce) were active in both tree species after the burn. However, the pattern of beetle activity differed between tree species. Sixty-eight percent of the pines were infested with bark beetles, but only 25 percent were infested with wood boring beetles (n=60). Conversely, only 32 percent of the fir contained evidence of bark beetle use, while 75 percent were used by wood boring beetles (n=60). Primary bark beetles emerged from the Jeffrey pine with greater cumulative frequencies each year. In contrast, cumulative emergence frequencies in fir remained the same throughout the study period. Secondary wood boring beetles showed a similar emergence patterns by increasing annually in pines but not in fir (*fig. 2*).

Woodpecker scaling activity did not vary between species or years (F=0.165; p=0.848). In contrast, woodpecker excavation activity differed appreciably between tree species and across sampling years (F=3.21; p<0.04). In general, white fir was utilized intensively by foraging woodpeckers during the first year after fire, but use declined dramatically thereafter. Mean cumulative foraging intensity on Jeffrey pine was relatively low during the first year but gradually increased over the study period until levels surpassed those initially seen in white fir (*fig. 3*).

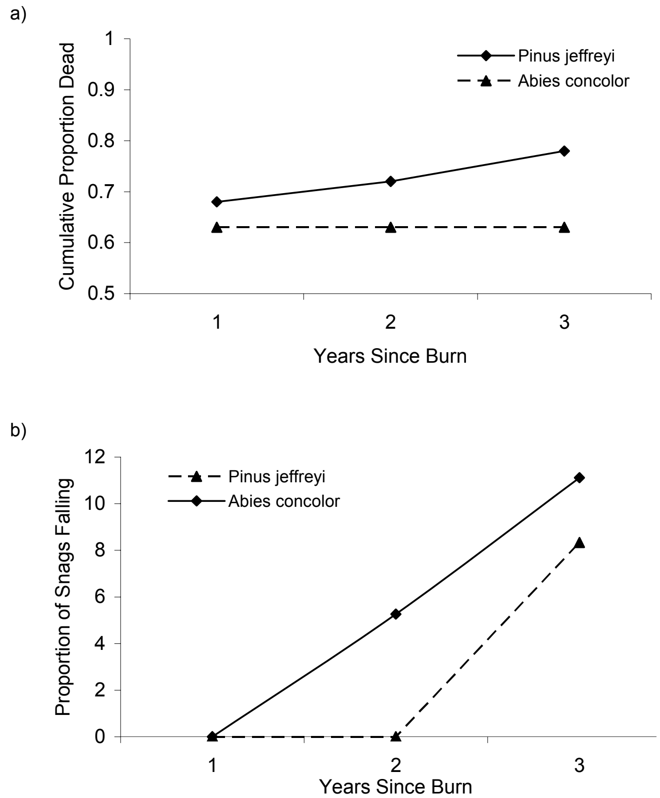

Figure 2—Emergence patterns of primary bark beetles (a) and secondary wood boring beetles (b) in Jeffrey pine and white fir following a spring prescribed fire in Lassen Volcanic National Park, CA.

Discussion

Our preliminary findings highlight potentially important differences in tree mortality, beetle colonization, woodpecker foraging, and snag decomposition between white fir and Jeffrey pine following early season fires in Lassen Volcanic National Park. All white fir mortality in our sample population occurred during the first year after the fire, with resultant snags being heavily infested by wood boring beetles. In contrast, Jeffrey pine mortality occurred throughout the study period as trees were attacked by bark beetles and subsequently colonized by wood boring beetles. Thus, white fir provided an immediate but short-lived post-burn foraging resource for woodpeckers, while Jeffrey pine provided a more prolonged foraging reserve as new pine snags were recruited into the population each year. Three years after fire, white fir snags were significantly more decayed and fell at faster rates than Jeffrey pine.

Differential morality and insect colonization of fir and pine provided different foraging opportunities for woodpeckers. In snags containing bark beetles, woodpeckers foraged by scaling away bark; in snags infested by wood boring beetles, woodpeckers foraged by excavating through bark and sapwood. Scaling activity was relatively low in both pine and fir during the first year and did not increase significantly in either species during subsequent years. Successful scaling is likely restricted to the first year of bark beetle infestation because beetles typically produce an adult brood within one year of colonization (Furniss and Carolyn 1977).

Excavation activity of wood boring beetles was common on white fir only during the first year, suggesting a complete exploitation of prey. In contrast, initial excavation activity was lower in Jeffrey pine but gradually increased each year as wood boring beetle activity increased.

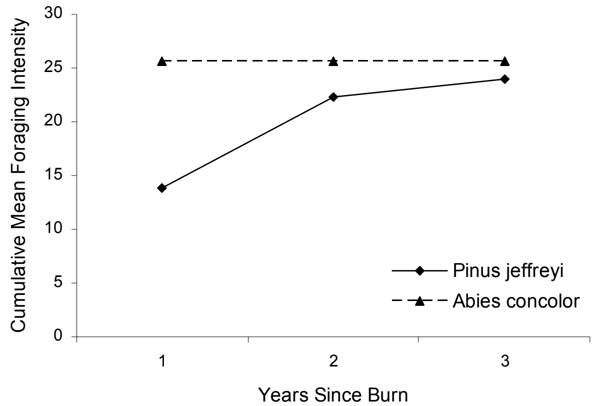

Figure 3—Mean cumulative woodpecker foraging intensities in Jeffrey pine and white fir in the first three years following a spring prescribed burn in Lassen Volcanic National Park, CA.

The observed patterns of tree mortality and insect colonization likely reflect the differential susceptibility of pine and fir to fire-caused damage. A positive correlation has been found between the amount of fire-cause tree damage, subsequent tree mortality, and beetle colonization in conifers (Agee 1993, Kelsey and Joseph 2003, Wallin and others 2003). In general, white fir is particularly susceptible to fire damage due to its relatively thin bark (Agee 1993, Laacke 1990). Trees that were weakened by fire but not immediately killed are typically attacked by phloem consuming bark beetles (McCullough and others 1998, Wallin and others 2003). Trees directly killed or severely damaged by fire are typically colonized by xylem consuming beetles (Furniss and Carolin 1977, Kelsey and Joseph 2003, McCullough and others 1998). In this study, white fir was more frequently colonized by wood boring beetles and woodpeckers excavated intensively in the snags outer sapwood.

Post-fire beetle infestation and ensuing woodpecker foraging patterns may have important long-term influences on subsequent decay in conifers and may help explain observed differences in decay patterns between Jeffrey pine and white fir in this study. Bark beetles and woodpeckers are both known carriers of fungi associated with wood decomposition (Farris and others 2004, Whitney and Cobb 1972, Paine and others 1997). Furthermore, the reproductive and foraging activity of beetles creates galleries and fragments the wood which can create microhabitats conducive to fungal invasion (Rayner and Boddy 1988). Foraging woodpeckers that respond to these beetle infestations can further modify the bark and underlying wood while they search for prey. Together, these processes may alter the local structure and microclimate of sapwood and lead to further fungal inoculation and subsequent decomposition (Rayner and Boddy 1988). Previous research has documented associations between foraging woodpeckers and greater wood decay in both hardwoods and conifers (Conner and others 1994, Farris and others 2004). In this study, a greater proportion of white fir snags were affected by both wood boring beetles and foraging woodpeckers during the first year following fire. Both high wood borer levels, followed by intense woodpecker foraging may have influenced the greater wood decomposition and subsequent attrition of this species.

Our sample population of white fir snags decayed more rapidly and fell at higher rates than the Jeffrey pine snags (*fig. 1b*). These results differ from snag attrition patterns reported elsewhere in the region, but may be difficult to compare. In both burned and unburned study areas, Jeffrey pine consistently fell at faster rates than white fir (Landram and others 2004, Morrison and Raphael 1993, Raphael and White 1984). However, the study by Raphael and White (1984) was conducted in a relatively older burn (7 to 11 yr), and none of the studies mentioned provide any information about the local bark and wood boring beetle activity, so comparisons are difficult. We suspect burn intensity and subsequent tree damage are the primary drivers of decay patterns observed in this study, but don't have adequate data at this time to evaluate these potential affects. Additionally, several studies have documented the influence of tree size on fall rates (Landram and Laudenslayer 2002, Raphael and White 1984, Morrison and Raphael 1993), with smaller trees having greater rates than their larger counterparts. We didn't detect a difference in diameters between trees that fell and those that remained standing, but small sample sizes could have precluded effective analysis.

In summary, our initial monitoring results suggest that differences in response of white fir and Jeffrey pine to an early season prescribed fire has important implications for post-fire woodpecker habitat. White fir provided a short pulse of high quality habitat for beetles and woodpeckers during the first year, while Jeffrey pine provided a more continuous supply of primary and secondary bark beetle prey for woodpeckers during the first 3 yr following the burn. However, more research in different burning and habitat conditions is needed to improve our understanding of post-fire habitat dynamics and validate the results of this case study. Our continued monitoring efforts over the next several years will help reveal how cavity generation patterns are related to species composition and early post-fire beetle and woodpecker utilization. We also plan to examine the influence of other variables such as fire severity, snag size and age on beetle infestation and subsequent woodpecker use and decay. These results contribute to our limited knowledge of wildlife habitat dynamics following spring prescribed burning in pine-fir forests of the southern Cascades. Prescribed burning may help to restore ecological interactions between insects, woodpeckers, and snag decomposition critical for snag dependant wildlife species.

Acknowledgments

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Developing a Multiscale Fire Treatment Strategy for Species Habitat Management[1]

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Abstract

Reintroducing fire to manage vegetation and fuel may have poorly understood consequences for wildlife. Prescribed burning can reduce down wood and snags that provide critical habitat and mechanical thinning designed to reduce fire hazards may alter forest structures that are preferred by some species. Moreover, fine scale fuel treatments may alter wildlife and habitat dynamics within the larger landscape. In this paper, we provide a process-based heuristic for understanding varied wildlife responses to fire at multiple scales that integrates fire behavior, vegetation dynamics and long-term habitat resilience.

Introduction

An increasing number of large wildland fires have prompted calls for changing how fire and fuel management is practiced in many forests of the West (GAO 1999, Covington 2000). Implementing alternative forest practices successfully may depend on how well wildlife concerns are integrated with prescribed burning and thinning. Historically, wildlife protections have provided logistic hurdles and legal constraints on management, but wildlife often are poorly integrated into proposed fuel and fire management alternatives. This may reflect our rudimentary understanding of the effects of fire regimes and fuel management on wildlife (Tiedemann and others 2000). Mechanical understory thinning or prescribed fire that is designed to reduce the chance of canopy fires or make fire easier to suppress may negatively affect species that require multi-layered canopies, high tree densities, or an abundance of down wood and snags. Strategies that fail to acknowledge the heterogeneous role of disturbance in generating habitat may place wildlife of concern at further risk. In forests that have been altered by fire suppression, successional changes in structure and composition may have favored some wildlife, but an increased hazard of stand-ending fire may erode the potential for long-term population persistence. Managers often must weigh the risk of uncharacteristic fire severity in the absence of fuel treatment against the complex effects of prescribed habitat alteration.

More is known about the habitat attributes that species prefer than how habitat dynamics affect reproductive success, adult mortality, or foraging opportunities that sustain populations over time. For example, tree-cavity users require a continuous source of snags, yet fire regimes differ in terms of how and when snags are created and how potential snags develop through time. Disturbance regimes may also differ in their effects on reproduction, mortality and foraging opportunities. As a second example, slow-moving species such as mollusks or amphibians may be particularly vulnerable to severe fire because they may recolonize burned areas slowly if

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extirpated. Places that are least likely to burn severely or frequently because of wildfire behavior and topography may provide disturbance-based refugia for improved conservation. Such process-based management requires a sophisticated knowledge of wildlife life history attributes and habitat dynamics. This understanding includes how population and vegetation dynamics interact within the larger landscape because habitat context (i.e., neighborhood effects) and habitat configuration are important (George and Zack 2001, Crist and others in press). In this paper, we integrate wildlife life history attributes with two distinct types of vegetation dynamics that are related to differences in fire behavior. We identify wildlife response groups that reflect population responses to these fire effects.

Wildlife Response Models

To integrate vegetation dynamics with wildlife responses to fire, we adapted a classification developed to describe plant responses to fire (Rowe 1983). In doing so, we expand on wildlife responses to fire that have been proposed by others (Oliver and others 1998, Hoff and Smith 2000). Use of this classification conceptually links wildlife responses to compositional responses of vegetation to fire. In restructuring the concept for wildlife habitat resilience, however, the response of vegetation structure to fire is also important. In our models, we address habitat structure by distinguishing between local and landscape fire effects. In the hierarchical classification below, the first group, *site recolonization*, typically results from radical changes in habitat structure and composition that occur in forest systems that experience stand-ending fire. The second group, *site persistence,* results from stand-modifying fire. Species responses are further distinguished within each of these two groups.

Model 1: Site Recolonization

Stand-ending fire profoundly changes wildlife habitat. Severe fire replaces living trees of the pre-fire forest with large numbers of snags and early successional habitat. Wildlife must quickly adjust to this change in habitat, and some species are well-adapted to take advantage of the post fire conditions. Over a half century ago, G. Evelyn Hutchinson referred to post-fire colonizers as “fugitive species” because they regularly move into recently burned areas in response to the availability of new habitat (Hutchinson 1951). In disturbance prone forests, however, all succession-dependent species (both early and late) can be considered fugitive when forest patches are dynamic in time and space.

In an idealized model of patch dynamics, a stand-ending fire regime provides a shifting steady-state mosaic in the larger landscape (Bormann and Likens 1979). At any given location, successional changes in forest structure and composition predominantly reflect the time since the last fire (Oliver 1981). Wildlife responses to these changes in habitat can be classified according to when habitat becomes optimal *(fig. 1)*. *Invaders* require recently burned areas for demographic success. Productivity quickly rises from unsustainable to optimal values in the years following fire. One of the best examples of this population response is the black backed woodpecker (*Picoides arcticus*) that thrives on the larvae of wood-boring bark beetles associated with recently burned areas (Hutto 1995). Over subsequent years and decades, habitats become suitable for different species according to species’ life history needs and the course of vegetation change (Thomas 1994, Oliver et al. 1998). *Avoiders* are species that thrive in association with old trees, an abundance of down wood, and structurally complex forests. An example of an Avoider is the winter wren (*Troglodytes troglodytes*) that is typically associated with late seral forests (McGarigal and McComb 1995).

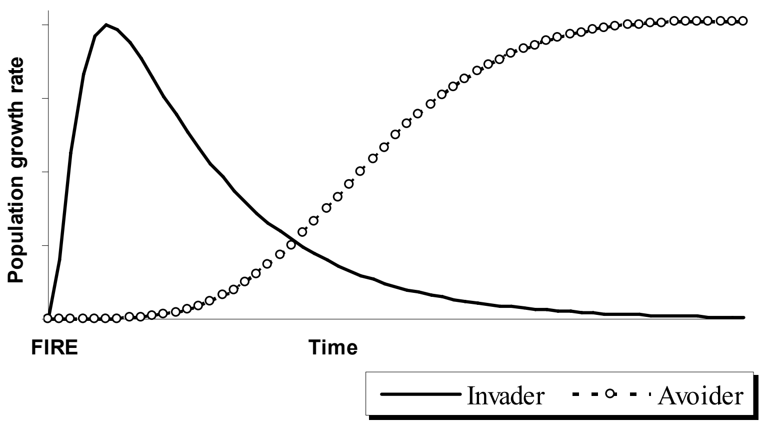

Figure 1—Wildlife responses of site recolonization for highly mobile species.

The recolonization potential of highly mobile bird and mammal species differs from that of less mobile species. Patch configuration may be especially important for colonizers that are relatively immobile or sensitive to edge effects. Colonizers may be sensitive to the distance between patches and the availability of migration corridors (Wiens and others 1985, Noss 1991). Edge effects may increase predation risk and provide limited effective habitat for maintaining wildlife populations. Only in a simplified patch dynamics model can disturbance, succession, and recolonization result in a predictable chronosequence of wildlife at a given site.

Within a landscape, an idealized flow of fugitive species among successional patches consists of a metapopulation-like dynamic (Pulliam 1988, Thomas 1994). Patches that provide optimal habitat act as demographic sources, while patches of sub-optimal quality are demographic sinks. The condition of habitat at all sites is ephemeral, however, and the continuity of habitat over time is evident at the scale of the entire landscape, rather than the individual patch. The eventual loss of optimal habitat is to be expected within a given patch because of successional change and the inevitable stand-ending fire.

This shifting-steady state model of wildlife recolonization provides a useful heuristic if the landscape is homogenous and all areas have an equal probability of burning, but this is rarely the case. Historically, fire frequency varied across mountainous landscapes, often in concert with changes in vegetation (Camp and others 1997). While the steady-state mosaic model incorporates sources and sinks that are constantly shifting, a model that includes relatively fire-free refugia includes somewhat stable source and sink areas. These refugia burn less often than the landscape as a whole, and are most likely to provide old trees, snags, and down wood. Such geographically stable habitats may function as persistent sources of emigrants and provide key habitat for the preservation of wildlife in a landscape over time (Crist and others in press).

Model 2: Site persistence

In contrast to the recolonization response of wildlife described above, some wildlife populations persist at a site through repeated fires. This is largely possible because fire effects on habitat are less severe than those of stand-ending fire. A fire regime of low to moderate severity does not result in radical changes in forest structure and is more likely to burn with variable intensity. That historical fires burned with low intensity in some forests is evidenced by fire scar analysis of individual trees that survived repeated fires. Reconstructed stand structures indicate that trees were typically clumped rather than regularly distributed. This structural complexity would have resulted in variable patterns of surface fuel accumulation and burn intensity. The fine scale heterogeneity of fire severity may have included unburned or lightly burned areas within the fire perimeter (Baker and Ehle 2001). Spotty areas of high fire intensity may have been associated with concentrations of shrubs, thickets of young trees, or down wood. In non-severe fire weather, such fuel structures may have resulted in intense, but passive crown fires. In the past, a sustained regime of patchy fire may have contributed to the resilience of these forests, although the way in which fire results in fine scale heterogeneity is poorly understood (Miller and Urban 1999). Some degree of fine scale variability in fire severity is common in recent wildfires (Lertzman and others 1998) and prescribed fires (Kauffman and Martin 1989). The creation of a burn mosaic is often an explicit goal of prescribed burning for wildlife management (Brownlie and Engstrom 2001).

Relatively high fire frequency and low fire severity generates an assemblage of habitat elements that is distinct from that of patch dynamics. A stand-modifying fire regime may result in aggregations of fire-resistant trees (Bonnicksen and Stone 1982), but down wood and snags may be lost unless unburned areas are present. Dry exposed wood is highly vulnerable to ignition from fire brands from adjacent trees and may readily combust from radiant heat. Moreover, snags that result from a fire may be less durable than those caused by other factors (Morrison and Raphael 1993). A loss of snags and down wood will likely reduce habitat for those species that depend on them, but a regime of frequent, low intensity fire perpetuates live tree structure for centuries. From the perspective of wildlife that thrive in these forests, a low-intensity fire regime provides a second type of refugia—one of chronic disturbance. Unlike refugia that burned less often than the surrounding landscape, these refugia provide relative protection from stand-ending fire that would require renewed succession and recolonization.

Wildlife populations may respond positively or negatively to stand-modifying fire, or they may show no notable response *(fig. 2)*. What distinguishes this group from the recolonizer responses is that a substantial number of individuals persist at a site following disturbance. The demographic success of *Endurers* is reduced after fire, but the population is able to recover without external subsidy from unburned populations. *Exploiters* show increased success in response to the fire, but habitat requirements are only temporarily improved over background levels. *Resisters* are not affected by fire. This latter response may be typical of generalist species or it may occur because fires are of sufficiently low intensity to not significantly modify critical habitat attributes. These disparate responses of wildlife to fire are linked to species' life history attributes and to subtle fire characteristics. Nuances in the heterogeneity of burn severity and season of individual fires may affect the vegetation response to fire and the subsequent quality of habitat for wildlife.

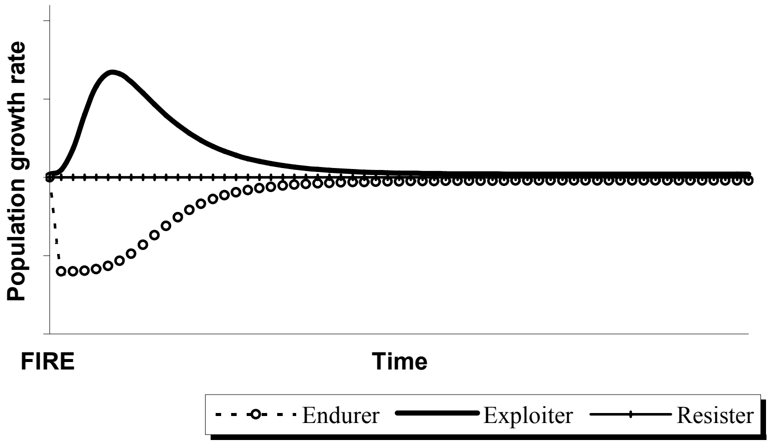

Figure 2—Wildlife responses of site persistence.

Integrating fire and wildlife management

Fire management strategies can integrate wildlife concerns better by being sensitive to how wildlife responds to vegetation dynamics. Management of systems that provide wildlife resilience through frequent low severity fire may fail when fire exclusion forces species to contend with uncharacteristically severe, stand-ending fire. Many species may be poorly adapted for recolonization. Moreover, requisite habitat structures may not develop in the absence of frequent fire or specifically-tailored fire surrogate treatments. In other areas, habitats that were sustained by stand-ending fire may be degraded at the landscape scale if fire is replaced by selective thinning to make stands artificially fire resilient. Sensitivity to the effects of different disturbance regimes is important for maintaining quality habitat over time.

Landscapes that are managed for site recolonization by wildlife require fuel treatment strategies that are consistent with wildlife adaptations. Strategies might include the following: 1) build redundancy of seral classes into the landscape, 2) ensure migration access among patches, 3) identify and maintain refugia that naturally burn infrequently and assess their importance for maintaining rare species, and 4) maintain disturbance size, edge, interior and openings within a desired range of variability. In forests, that historically experienced stand-ending fire, 20th century fire exclusion may have reduced the areas in mid age classes—thereby placing future recolonization at risk as stand-ending fire eliminates old forest habitat. Managers might weigh the relative effects to wildlife of severe wildfire, stand ending prescribed fire, and emulative silviculture. Stand-ending wildfires are difficult to control, however, and extensive high severity fire is rarely prescribed outside high-elevation wilderness areas because of social and ecological constraints.

In contrast to recolonization-based management of habitat, ensuring site persistence may require fine scale fuel treatments when habitat dynamics have significantly changed from historical conditions. Management strategies may include the following: 1) restore fine-scale forest structure to increase the heterogeneity of fire severity, 2) burn during non-extreme weather for patchy effects (e.g., nocturnal or cool season burning), and 3) individually target old trees, snags, logs (e.g., raking, spot thinning, covering logs with soil, variable drip torch burn strategies). This fine-scale surgical approach to habitat management may be difficult because it involves high cost and effort depending on the degree that current conditions depart from the habitat-sustaining dynamic. Our understanding of the subtle effects of fine scale fire dynamics on wildlife processes is more limited than it is for the radical effects of stand-ending fire, and this is especially true for rare species that typically constrain management the most.

Conclusion

In conclusion, the ecological effects of fire on wildlife are complex, but an improved integration of fire and fuel management with wildlife will be possible when habitat dynamics are more fully understood. An assessment of how historical habitat dynamics functioned may provide insight into how prescribed fire and fire surrogate treatments can be designed to better accommodate wildlife at coarse and fine scales. In this paper, we have provided a conceptual model that pairs two different habitat-generating processes with two fundamental wildlife responses to fire. We believe that this provides a useful heuristic for broad scale planning. Desired future conditions for wildlife and forests should incorporate these key scale-specific processes while being sensitive to social and ecological constraints.

Acknowledgments

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Short-Term Effects of Wildfires on Fishes in the Southwestern United States, 2002: Management Implications[1]

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Abstract
Summer 2002 was a season of markedly increased wildfire in the southwestern United States. Four fires affected landscapes that encompassed watersheds and streams containing fishes. Streams affected in three of the four fires were sampled for multiple factors, including fishes, to delineate the impact of fires on aquatic ecosystems in the -Southwest. All fishes were lost in one stream, Ponil Creek, affected by the Ponil Complex fire. In two streams, Rio Medio and West Fork of the Gila River, 60 to 80 percent reductions in fish populations occurred following combinations of ash and flood flows. In 2002, information on fires effects on fishes was dramatically increased for mostly native, non-salmonid species of fishes. Results of these short-term studies suggest that the impacts of fire on fishes in lower order montane streams are extensive and negative. Because of the listed status of many (70 percent) of southwestern fishes, land and resource managers must be vigilant of opportunities to protect these species following wildfire

Introduction
Prior to the 1990s, there was little information on the effects of wildfire on aquatic ecosystems and their inhabitants. With the Yellowstone fires of 1988, increased effort to study these impacts was initiated. Since that time, information on fire effects on aquatic ecosystems has increased dramatically. Notwithstanding, most of the information is on forested ecosystems in the northern Rockies. In spring, 2002, a workshop was held in Boise, Idaho to address emerging issues, experiences, and theory relative to the effects of fire and its management on aquatic ecosystems. The results of this workshop were published in spring 2003 in *Forest Ecology and Management*. This effort is the most comprehensive to date addressing these issues and should be referenced by all interested in this topic.

In the Southwest, little information is available on the effects of fire on fishes. Propst and others (1992) first discussed the impacts of fire on a native endangered trout, the Gila trout (*Oncorhynchus gilae),* in headwater streams in southwestern New Mexico affected by the Divide Fire, 1989. Soon after, Rinne (1996) reported on the effects of fire on rainbow (*O. mykiss)* and brook (*Salvelinus fontinalis)* trout in three streams affected by the Dude Fire in central Arizona in 1990. Rinne and Neary (1997) summarized and assessed the probable affects of wildfires on streams in the Southwest.

In summer 2002, fires were once again extensive and intensive across the drought-stricken West. In Forest Service Region 3, Arizona-New Mexico, 440 kha

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(1.1 million ac) or about three times the average over the past decade were consumed by wildfire in 2002. We took opportunity to examine the short-term effects of these fires on aquatic ecosystems and their inhabitants, primarily fishes. A primary purpose of our efforts was to expand our data base on fish species other than salmonids. Most of the information that is available on fire effects is on species of salmon and trout (Rieman and others 2003, Dunham and others 2003). In addition, we were interested in determining which factors; species of fish, fire size and intensity across landscapes, changes in stream habitat and hydrological regimes (Rinne 2003a) have the greatest post fire impact on fish survival in the Southwest.

This paper will present only the short-term effects of wildfire on fish populations in three of the four fires. Additional information on habitat, post-fire hydrology (changes in flow), water quality, and fire size and intensity will be presented elsewhere. Most importantly, information were obtained on a dozen, mostly non-salmonid fishes in three separate stream affected by wildfire. Because the southwestern fish fauna is primarily a cypriniform assemblage (Rinne 2003b) several of these fishes are special status (threatened, endangered or sensitive) species. Although these are short term, immediate (< 4 months) post-fire effects, when combined with previous information on both introduced and native trout, they have management implications. Finally, the regionality of fire impacts on fishes in southwestern aquatic ecosystems will be briefly addressed.

The Fires

The three fires sampled that affected watersheds encompassing streams with surface water and fishes extant were the Borrego, Ponil Complex, and Cub Mountain. All were lightening-caused fire ignited during the building of the monsoon season characteristic of the Southwest. The Ponil Fire was contained at over 90,000 ac; the Boreggo and Cub Mountain fires were smaller, both burning about 13,000 acres.

Streams Affected

Middle Ponil Creek was the primary stream draining the fire-impacted area. Sample points were in second and third order stream channels. Rio Medio, affected by the Boreggo Fire, is a second order stream, and the West Fork of the Gila River (Cub Mountain) is a third order stream at points of sample. With exception of Ponil Creek, all had surface water at time of immediate post-fire sampling. Ponil Creek was intermittent and in extreme drought condition during late June sampling and sample locations in Ponil Creek were determined by presence of surface water during initial sampling effort. Limestone and Carizzo creeks were in low flow conditions (approximately 0.1 m^3 sec^{-1}; 1 cfs) at points of sample. Corduroy Creek had reduced, modest (0.3 m^3 $min.^{-1}$; 3 cfs) surface flow at time of initial sampling. Rio Medio had the greatest base flow (17 to 25 m^3 $min.^{-1}$, 10 to 15 cfs), with the West Fork of the Gila being much reduced in flow (<5 m^3 $min.^{-1}$; 5 cfs).

The Fishes

A dozen new species of fishes were sampled in the streams affected by the wildfires in the Region three of the U. S. Forest Service. In Ponil Creek, information was obtained on densities and biomasses of three cypriniform species: creek chub, *Semotilus atromaculatus,* white sucker, *Catostomus commersoni,* and blacknose dace, *Rhinichthys cataractae.* These three species comprised the major portion (98 percent) of the fish assemblage (*table 1*). Rainbow trout (*Oncorchynchus mykiss)* also were present in low numbers.

Table 1—*Total fish numbers at eight sample points, Ponil Creek, June 29 to July 1, 2002. Site 1 was above the fire influence zone, site 4 was affected by a small ash flow, and site 6 was affected by an ash/flood event by time of late June sampling.*

	Fish Species			
Site	**Longnose dace**	**White sucker**	**Creek chub**	**Rainbow trout**
1	15	8	6	18
2	71	10	30	5
3	42	45	291	0
3a	58	142	208	0
4	78	42	164	13
5	159	46	135	0
6	0	0	0	0
7	91	95	138	0
total	514	388	972	36

Only brown trout, *Salmo trutta,* were present in Rio Medio; however, this represented a new species of trout for which fire effects was determined. In the West Fork of the Gila River, data were collected on a half a dozen native species of the Gila River basin: longfin dace, *Agosia chrysogaste*; speckled dace, *Rhinichthys osculus*; Sonora sucker, *Catostomus insignis*; roundtail chub, *Gila robustus*; desert sucker, *Catostomus clarki*; and the threatened spikedace, *Meda fulgida*. This reach of river also contains the threatened loach minnow, *Rhinichthys cobitis*; however, this species has only been collected downstream a kilometer. These downstream reaches were intermittent during regular, annual sampling exercises in May and therefore were not usable in delimiting the effects of wildfire. Spikedace have been collected at tan established, long term sample point (Rinne and others 2005). In summary, in summer 2002, a dozen new species were collected in reaches of six streams that potentially could be affected by post wildfire impacts.

Effects of Wildfires on Fishes, Summer 2002

Ponil Complex Fire, Ponil Creek

Eight sample sites were established in June on mainstream Ponil Creek, which was in extreme drought condition. Only site 1 was positioned in what was determined above the influence of the fire. A single ash/flood event already had occurred on the North Fork of Ponil Creek by the time of initial sampling and had affected the lowermost site, 7. Waters were highly turbid and blackened in pools, and the stream was low in flow, as subsurface recharge had only commenced. Similarly, a much smaller (<1 cms) ash flow event had issued from Horse Canyon and affected the immediate, downstream sample site 4. Water was brown and tannic in appearance at time of sampling in June, but all species, including rainbow trout, were present (*table 1*). The other sites were unaffected at the time of initial June sampling because of location and stream intermittency.

Almost 2,000 individual fishes were collected at the eight sample sites in late June/early July (*table 1*). Creek chub comprised half of total fishes collected. By comparison, longnose dace made up 27 percent, white sucker 20 percent, and rainbow trout only 3 percent of total fishes sampled at the eight areas. Rainbow trout were collected at only three of the eight sample areas, the two uppermost sites and the less open, aspect-shaded, canyon-bound site 4 at Middle Ponil Camp. All cypriniform species were present at all sites except site 7, which was previously (1 week prior) affected by a post-fire flood and ash event. All species including trout were present and abundant at site 4, affected by the minute post-fire flow event. In general, the cypriniform species displayed healthy populations, with young-of-year being present. In contrast, trout were mostly greater than 125 mm total length (TL) and averaged near 180 mm TL.

Sampling on August 7-9 at all eight sites resulted in fish being present at site 1 only. Water quality was poor, and two ash flows on August 7-8 were sampled and affected sites 3 through 8. A flood event (estimated 250 m^3 sec^{-1}; 1,500 cfs) occurred in mid July and affected all sites but 1. Numbers of fishes in a single pool at site 1 were not observed to be different than those estimated in initial sampling. Re-sampling site 1 in its entirety again on 20 October further documented the lack of change in fish populations (*table 2*). Rainbow trout and white sucker were similar in numbers, longnose dace doubled in numbers since June sampling, and only two creek chub were collected. Total numbers of fishes were nearly identical for the two sampling events.

Table 2—*Comparison of fish species abundance and total fish numbers in Ponil Creek, site one, at initial (June) and final autumn (October) sampling, 2002.*

Species	June	October
Site 1		
Rainbow trout	18	13
White sucker	8	5
Creek chub	6	2
Longnose dace	15	29
Totals-Site 1	47	49
Sites 2-8	1910	0

Boreggo Fire, Rio Medio

Three sites within the fire-influence zone were established and sampled in June on the Rio Medio. Only brown trout were present in the stream, which represents a new species of trout potentially impacted by wildfire. Total numbers of trout 50 m^{-1} section of stream were near identical in sites 1 and 2 and almost doubled at the most upstream site, 3 (*table 3*). Sites 2 and 3 were not sampled in early August.

Table 3—*Pre- and post-fire comparison of brown trout densities per 50 m of stream at three sites in Rio Medio. Percent reductions between June and October are in parentheses. Sites 2 and 3 were not sampled in August.*

Site	June	August	October
1	74	33	21(74)
2	77	-	19(75)
3	127	-	18 (86)

Near identical numbers of brown trout were present at sites 1 and 2 during initial, June sampling. By contrast, almost twice that number was estimated to be present at site 3. Brown trout at site 1 were reduced by 50 percent during early August sampling and greater than 70 percent by final, autumn sampling in October. Similarly, estimates of brown trout numbers at sites 2 and 3 were reduced by 70 percent or greater by final sampling. No size specific loss was present in data.

Cub Mountain Fire, West Fork of the Gila River

Two sample sites were established and sampled in early and late July and again in early October. The sites were located on the West Fork of the Gila River immediately above the Cliff Dwellings National Monument. One site was about 200 m below a regular monitoring site on the West Fork (Rinne and others in press), and the other about a half kilometer above that site. Five of the six cypriniform species native to the Gila River basin were most common in initial early July samples (*table 4*). A single brown trout, yellow bullhead (*Amerius natalis*), and smallmouth bass (*Micropterus dolomieui)* were collected in the late July sampling.

The response of native Gila River fish populations is shown in *table 4*. Several ash flows occurred between mid and late July sampling events. A flood event (210 m^3 sec^{-1}, 1,300 cfs) occurred in early September after the ash flows and between the second (31 July) and last or autumn (5 October) sampling. Total fish numbers were reduced 60 to 80 percent at the two sites between July and October.

Table 4—*Total numbers of fishes in 50 m reaches of stream in the West Fork of the Gila River, July and October, 2002.*

Date	Site 1	Site 2
Early July	168	560
Late July	278	481
October	50	118

Discussion

Post-fire impacts on fishes in all streams sampled were apparent. Most dramatic was the loss of all species at all sites in the fire-affected reaches of Ponil creek (*table 2*). The effects were already evident at site 7 during initial sampling in late June. The combined ash, flood event issuing from the North Fork of Ponil Creek a week prior had apparently already been fatal to all fishes (*table 1*). Several were found dead in fine sediments deposited by this flow. By contrast, the small ash flow from Horse Canyon was not fatal to fishes at site 4.

Brown trout were dramatically (>70 percent loss) reduced at all three sample sites in the Rio Medio following the Boreggo Fire. Based on the sample from site 1 in August populations of this salmonid species appeared to gradually decline through the summer. Warnecke and Bayer reported a 90 percent loss of trout in Canyon Creek, also affected by the Rodeo-Chedeski Fire. Similar reductions in native cypriniform fishes were recorded in the West Fork of the Gila River.

Management Implications

Data collected on the short-term effects of fire on the dozen species of fishes in a half dozen streams corroborate previous findings by Propst and others (1992) and Rinne (1996). Because the majority of southwestern native fishes are threatened, endangered or Forest Service sensitive species, managers must be vigilant of opportunities to remove fishes from streams whose watersheds are affected by wildfire. Efforts such as those conducted for Gila trout following the Divide Fire (Propst and others 1992) may be considered a fundamental management approach to address native fish sustainability in the Southwest following wildfires. Because most populations of rare, southwestern fishes are isolated and unique genetically they indeed become evolutionary significant units. As such, they cannot be replaced once lost. Further, the climate and landscapes of the Southwest result in fragmentation of aquatic habitats (Rinne 1995). Such fragmentation precludes natural repatriation that occurs more readily and frequently with primarily salmonid species in the more mesic northern Rockies and Pacific Northwest (Rieman and others 2003, Dunham and others 2003).

Finally, 2002 was a year characterized by an increase in fire activity in Region 3 of the U. S. Forest Service (Arizona and New Mexico). The Southwest is currently in a period of extreme drought. Continued drought combined with the massive outbreak of bark beetles across forested landscapes has resulted in thousands of acres of dead and dying trees in the national forests of the southwestern region. The potential is high for even greater wildfire activity in summer 2003. In parallel, the probability likewise increases that additional streams containing rare and endangered fishes will be impacted by the aftermath of these fires. Land and fishery resources managers must be prepared to take strategic, coordinated, and timely responses to these events as they potentially affect the invaluable, often locally irreplaceable resource, native southwestern fishes.

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Spatial Characteristics of Fire Severity in Relation to Fire Growth in a Rocky Mountain Subalpine Forest[1]

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Abstract

We compared the spatial characteristics of fire severity patches within individual fire "runs" (contiguous polygons burned during a given day) resulting from a 72,000 ha fire in central Idaho in 1994. Our hypothesis was that patch characteristics of four fire severity classes (high, moderate, low, and unburned), as captured by five landscape metrics, would vary with the size of the run (ranging from 1 to 6,200 ha). Our results indicated that high severity patches (i.e., crown fires) were typically larger and more complex in larger runs than in smaller runs. Moderate severity, low severity, and unburned patch characteristics were relatively insensitive to fire run size, with most metrics showing little or no trend.

Introduction

The importance of large, infrequent fire events in shaping subalpine forest pattern and dynamics has received much attention since the Yellowstone Fires of 1988 (Romme and Despain 1989, Turner and Romme 1994). Climate and weather conditions that contribute to such events are rare in these high elevation systems, and most fires remain small. Periodically, however, severe drought and fire weather conditions set the stage for extensive fires that can burn tens of thousand of hectares. These landscape-scale fire events occur infrequently (every 90 to 400 yr) in the northern Rockies but are ecologically significant, because they account for the majority of area burned over time and shape the subalpine forest landscape mosaic for many years (Turner and Romme 1994, Turner and others 2001).

The degree to which infrequent large fires differ qualitatively and quantitatively from smaller, more frequent fires remains an unresolved question in ecology (Turner and others 1997a). Prescribed burns and Wildland Fire Use (WFU) fires may also differ from larger wildfires in terms of spatial characteristics and fire behavior. Fire severity, which is frequently characterized by the relative degree of canopy mortality, has direct effects on many ecological processes, including postfire tree recruitment (Anderson and Romme 1991), successional pathways (Turner and others 1997b, 1999), diversity (Romme and Knight 1982), old-growth and wildlife habitat (Eberhart and Woodard 1987, Kushla and Ripple 1998), erosion (Knight and Wallace 1989, Minshall and others 1989), and subsequent fire spread (Renkin and Despain 1992, Turner and Romme 1994). Large wildfires often burn for long periods across

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extensive areas and encompass a wider range of weather conditions, topography, and vegetation variability than smaller fires.

Remote sensing and GIS have been used widely to map fire growth and extent (see Miller and Yool 2002), but fewer studies have analyzed spatial patterns of fire characteristics such as fire severity (Turner and others 1994). However, the area burned on any given day is usually divided up into a large number of individual patches of varying size, because spread progresses from multiple fronts and/or becomes fragmented. This can confound calculations of patch characteristics, such as size and shape, and landscape pattern complexity. Our objective is to quantify and compare the spatial characteristics of fire severity patches using “fire runs” (individual, contiguous polygons burned during a given day) as the unit of analysis. We examine landscape metrics for different fire severity classes as a function of fire run size and speculate on the relative influence of large and small fires on the landscape mosaic.

Methods

Study Area

The Corral-Blackwell Complex Fire burned approximately 72,000 ha in the Salmon River Mountains of central Idaho in 1994 (*fig. 1*). The fire burned from August 3 through October 6, when it was extinguished by precipitation. Much of the area burned resulted from few days with exceptional fire growth (*fig. 2*). Fire suppression focused primarily on structure protection and confinement and was minimal throughout the burn. Elevation within the fire perimeter ranged from 600 to 2,735 m, but more than 85 percent of the area burned was in subalpine forests above 1,850 m. Affected forests consisted primarily of mixed stands of lodgepole pine (*Pinus contorta* Doug. Ex Loud. var *latifolia*), Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), and subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.). Other species included Douglas-fir (*Pseudotsuga menziesii* var. *glauca* [Beissn.] Franco), grand fir (*Abies grandis* [Dougl.] Lindl.), and whitebark pine (*Pinus albicaulis* Engelm.).

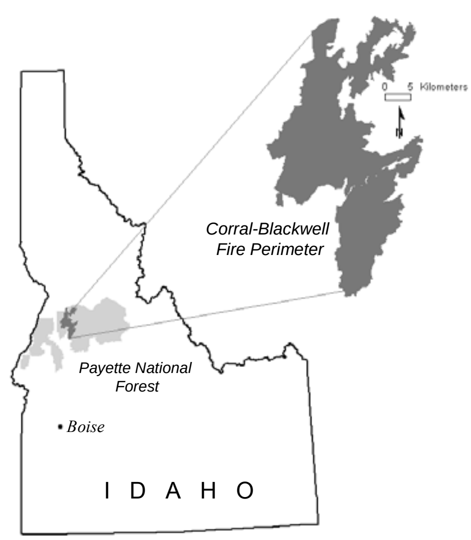

Figure 1—Location and perimeter of the Corral-Blackwell Fire.

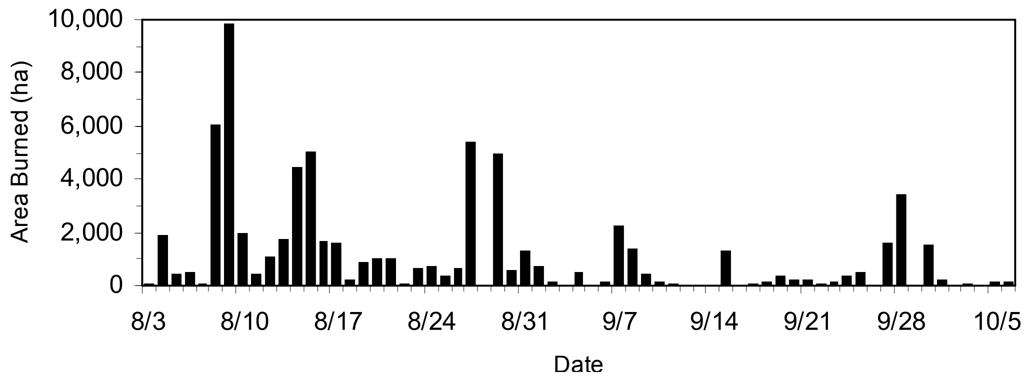

Figure 2—Daily area burned in the Corral-Blackwell Fire.

Fire Severity Mapping

The relative degree of crown damage and canopy mortality caused by a fire is commonly referred to as *fire severity*. Immediately after a fire, crown damage is closely related to the behavior and intensity of the fire, but the degree of canopy mortality can change dramatically over time as burned trees eventually succumb to cambial or crown damage and changes in environmental conditions (Agee 1993, Ryan and Amman 1996). Thus, analyses of the postfire mosaic are dependent on the timing of fire severity mapping. Kafka and others (2001) distinguish between initial and longer-term damage by calling the former "fire impact" and the latter "fire severity." While we focus explicitly on initial patch characteristics related to the behavior and intensity of the fire, we choose to utilize the more widely recognized term fire severity but acknowledge the important distinction between initial and eventual impacts of fire.

For this research, we used a spatial database developed by the U.S. Forest Service after the Corral-Blackwell Fire (Boudreau and Maus 1996, Farris and others 1998, Greer 1994). Fire severity was mapped from Landsat Thematic Mapper Imagery acquired on October 8, 1994. An unsupervised classification was performed to delineate four classes of fire severity based on the relative amount of overstory crown damage (see Boudreau and others 1996, Farris and others 1998 for more detail):

1. High Severity–Complete canopy consumption (e.g., a "crown fire"). Duff layer usually consumed.
2. Moderate Severity–Needles scorched but not consumed on >50 percent of stand. Duff layer usually partially consumed.
3. Low Severity–Little discernable damage to the overstory foliage (<50 percent of stand). Duff consumption light or patchy.
4. Unburned–An unburned area within the fire perimeter.

Imagery was flown within 1 week of active burning, therefore crown damage was primarily due to direct impact rather than longer-term mortality. Based on an assessment of 238 field plots established after the fire and aerial photo interpretation, the map accuracy ranged from 89 percent for the high severity class to a low of 71 percent for the low severity class.

Stratification of Fire Severity by "Run" Size

Using daily fire progression maps created by the U.S. Forest Service with twice-daily infrared flights (Greer 1994), we identified individual runs using GIS (*fig. 3a*). Many runs were composed of patches of varying severity, including unburned areas. Fire severity information from the classified TM image was extracted for each run and compiled into a database for the calculation of landscape metrics (*fig. 3b*). All runs were assigned to one of fourteen rank run size classes based on the standard deviation from mean run size (*table 1*). This approach provided an even and representative gradient of run sizes, but the lack of replication of larger runs made rigorous statistical comparisons difficult.

Table 1–*Ranked run size classes based on standard deviations (STDV) from the mean, and the corresponding area and number of patches (n) within each class. The mean fire run size was 36 hectares.*

Rank run size class	STDV from mean	Area (ha)	N
1	-1[a]	<36	631
2	0-1	36–266	154
3	1-2	266-496	23
4	2-3	496-727	10
5	3-4	727-957	5
6	4-5	957-1,187	2
7	5-6	1,187-1,417	3
8	6-7	1,417-1,647	2
9	8-9	1,877-2,107	2
10	9-10	2,107–2,337	1
11	10-11	2,337–2,567	1
12	14-15	3,257–3,487	1
13	15-16	3,487–3,717	1
14	28-29	6,022-6,252	1

[a] Fire runs less the mean size of 36 ha.

Spatial Analysis and Metric Calculation

Indices have been developed to describe landscape patterns (e.g., McGarigal and Marks 1995), but many provide redundant or ambiguous information. Riitters and others (1995) demonstrated that landscape patterns can be effectively captured using a few metrics. Li and Reynolds (1994) suggested that spatial heterogeneity can be broken down into five major components: 1) number of patch types, 2) proportion of each type, 3) spatial arrangement of patches, 4) patch shape, and 5) contrast between patches. We thus chose five metrics to characterize the composition, size, shape, and interspersion of fire severity patches within individual fire runs:

1. CA percent - Proportion of each fire run area in each severity class
2. MPS - Mean patch size for each fire severity class
3. LPI - Largest Patch Index is the percentage of a fire run area occupied by the largest patch of a given severity class.

4. AWMSI - Area Weighted Mean Shape Index measures patch shape complexity weighted by size. High values are more irregular shapes.
5. IJI -Interspersion and Juxtaposition Index, measures interspersion and adjacency of severity classes IJI ranges from 0 to 100: 0 indicates patches of a class are adjacent to one other class, while 100 indicates patches of a class are equally adjacent to all other classes

We used the Patch Analyst Extension in ARCVIEW GIS (Elkie and others 1999) to calculate the metrics and restricted our analysis to class-based metrics. Additional information is provided for these metrics in McGarigal and Marks (1995).

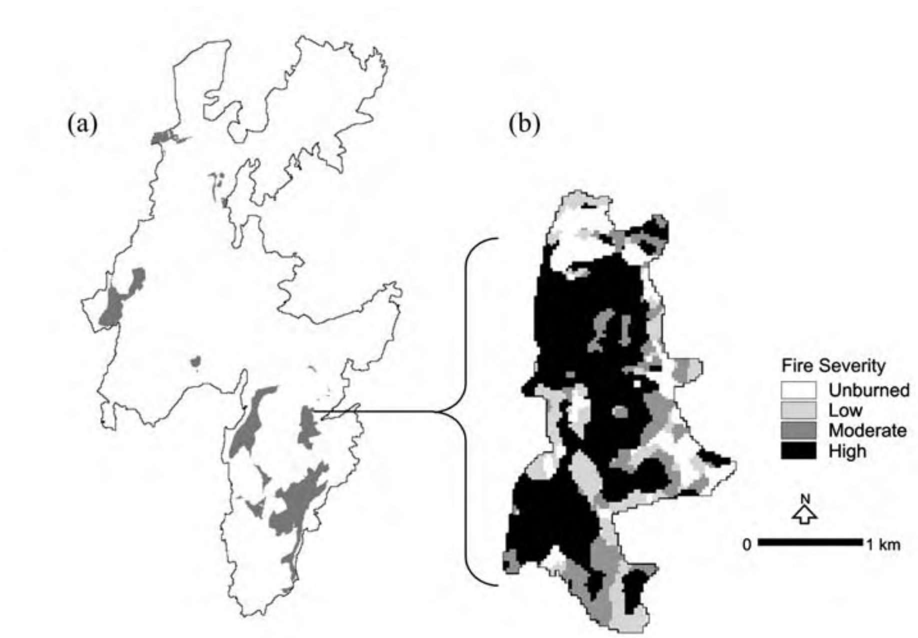

Figure 3—Example of the individual fire runs from the Corral-Blackwell Fire on August 15 (a) and a close up of fire severity patterns within a single run (b).

We calculated metric values for each of four fire severity classes in each fire run., then plotted the mean metric value for each class in fire runs belonging to the same run size class (e.g., a mean patch size was calculated for high severity patches in all fire runs in Size Class 2). The mean values for small runs were based on hundreds of runs while large runs were based on only one to a few (*table 1*). Computed metric values for each severity class were plotted against rank run size. A best fit regression line was fit to the data to provide a descriptive measure of trends.

Results

Although it might be expected that larger patch sizes (MPS) would be associated with larger runs, this was only the case for high severity fire patches (*fig. 4*). Further, as runs became larger, the proportion of the run area (CA percent) composed of high severity fire patches increased and the proportion of area occupied by low severity burns and unburned islands decreased slightly. Smaller runs contained a mix of different severity classes. High severity patches had progressively higher values of AWMSI and LPI for larger run sizes which, when coupled with the results for mean patch size and class proportion, indicates that crown fire patches were typically larger and more complex in larger runs than smaller runs. Moderate severity, low severity, and unburned patch characteristics were relatively insensitive to fire run size, with most metrics showing little or no trend. Moderate severity patches increased in size slightly as run size increased, but the average patch size for all non-high severity patch classes was generally small (<3 ha). Fire runs greater than 1,000 ha were largely composed of large high severity patches with small patches of the other severity classes. The degree of interspersion between patches of different classes generally decreased as run size increased, but the decrease was strongest for high severity and moderate severity burns. High and moderate severity burns were more closely associated with each other spatially as the size of the run increased.

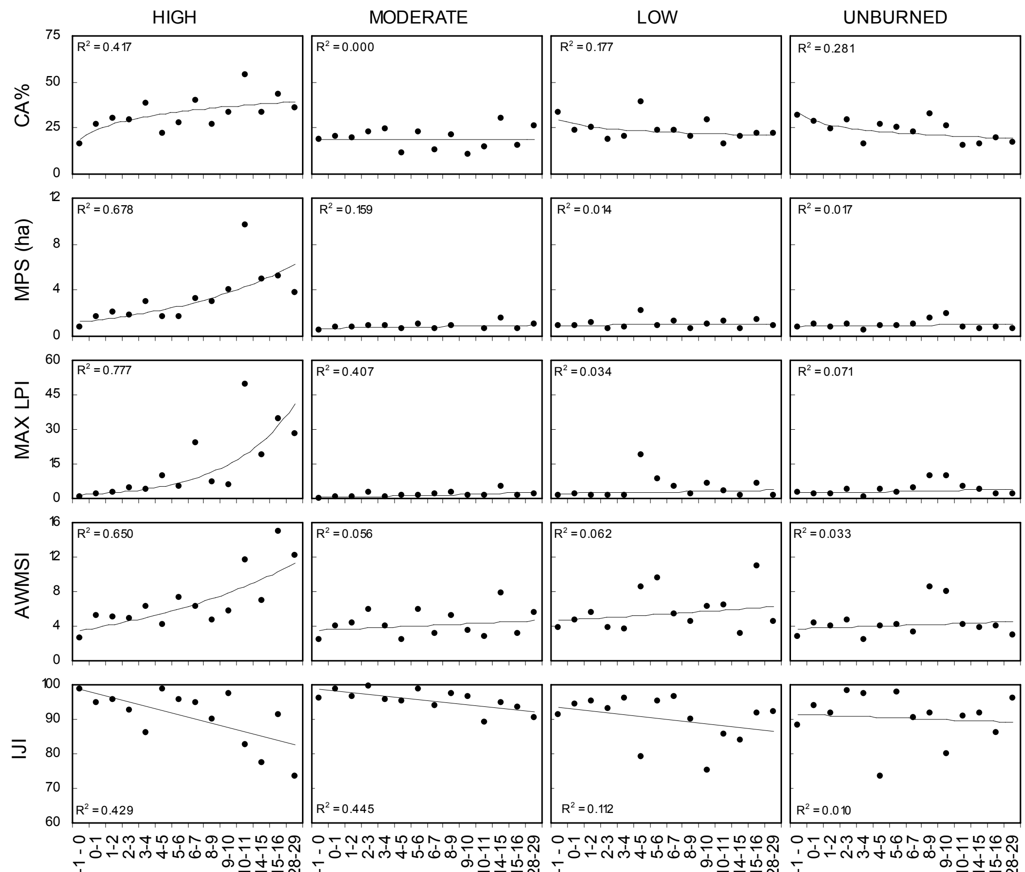

Figure 4—Trends in landscape metric values vs. rank fire run size for unburned, high severity, moderate severity and low severity fire patches. CA percent=proportion of total area; MPS=mean patch size; Max LPI=maximum largest patch index; AWMSI=area weighted mean shape index; IJI=interspersion and juxtaposition index.

Discussion

These results suggest that variability in the size of individual runs during large fires may influence the overall heterogeneity of the postfire mosaic. Large runs burn more area than smaller runs and result in fundamentally different patch characteristics. Larger runs generally had a higher proportion of high severity burns than small runs, and the high severity burn patches were larger and more complex on average. The degree of interspersion varied in larger runs, with a greater association between high and moderate severity burns. Erratic fire behavior during larger runs resulted in frequent transitions between severe surface and crown fires with more crown scorch along large patch edges. The interplay of small and large runs combined to create a great diversity of patch characteristics across the landscape.

Turner and others (1994) found that the proportion of high fire severity burns during the 1988 Yellowstone Fires increased relative to low and moderate severity burns as the gross daily area burned increased. Although exact comparisons are not possible, our results are similar. This is because large runs responded to severe fire weather conditions that promoted crowning, but the role of terrain and vegetation patterns cannot be discounted (Kushla and Ripple 1997, Kafka and others 2001). Turner and others (1994) found that small early season fires in Yellowstone were more heterogeneous than larger, late season fires. Interestingly, we found that larger high severity patches were more complex in shape and corresponded to topographic configurations.

Large fires such as Corral-Blackwell have spectacular high intensity runs with huge canopy gaps, yet they can also produce many low severity patches. The extensive number and area of active flame fronts across the landscape as the fire grew allowed for lower intensity fires to creep around the perimeter during periods of moderate or low weather danger. Patchy surface fires were common along the flanks of huge crown fires during mild weather, suggesting that large, infrequent fire events are likely an important source of low and moderate severity burning in subalpine forests. An average of 24 percent of the burned area across all run sizes was classified as low severity immediately after the burn, although a significant proportion of the area mapped as low severity is likely to experience increased mortality over time due to the high susceptibility of these subalpine conifers to fire damage. All of these variations in disturbance severity (e.g., high vs. low severity) and the timing of effects (e.g., immediate damage vs. longer-term effects) provide a range of different successional pathways and opportunities for subalpine forest species (e.g., Kipfmueller 2003, Kipfmueller and Kupfer 2005).

Our results underscore that infrequent large high severity wildfires with large runs have a disproportionate effect on structuring the post-fire mosaic because they create patches with very unique characteristics. Landscape heterogeneity that results from large, multiple week burns may be difficult to recreate using a series of smaller burns over time, even if they ultimately consume the same amount of acreage. Further, because conditions during prescribed fires and many Wildland Fire Use fires are frequently very different than during large wildfires, the resulting patch mosaic will be very different, which may have implications for stand structure, composition and long-term dynamics. Although large fire runs accounted for the majority of area burned, the large number of smaller fire runs still contributed significantly to burned acreage and created different spatial characteristics in between larger runs. The degree to which these initial postfire patterns persist or converge through time is unknown and will require long-term monitoring to understand.

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Madrean Pine-Oak Forest in Arizona: Past Dynamics, Present Problems[1]

Andrew M. Barton[2]

Abstract
This paper synthesizes research on presettlement dynamics and modern disruption of Madrean pine-oak forests in Arizona. In response to surface fires characteristic of presettlement times, pines were fire resistant, exhibiting high top-survival, whereas oaks were fire resilient, exhibiting lower top-survival but pronounced resprouting. Thus, low-severity fire favors pines, but resprouting allows oaks to rebound during inter-fire periods. Age structures reveal large increases in stand density, especially for oaks, as a result of modern fire suppression, suggesting more open conditions and higher pine:oak ratio during presettlement times. Stands were, in fact, so open that fire-caused thinning rarely stimulated radial growth. Frequent fires also apparently excluded less fire tolerant species, which have invaded some pine-oak sites. In anomalous stand-replacing crown fires, seedling establishment was very low for pines and oaks, but most oaks resprouted. *Pinus leiophylla* also resprouted but at low levels, which might nonetheless be an important source of future pines. These results suggest anomalous high-severity fires can transform Madrean pine-oak forests into more homogenous oak woodlands. This synthesis argues for urgent restoration using a variety of flexible approaches.

Introduction

Prior to Euro-American settlement, lightning fires shaped the forests of the American Southwest (Covington and Moore 1994, Swetnam and Betancourt 1998). The dramatic anthropogenic reduction in fire frequency, beginning in the mid-19th century, has radically changed the structure, composition, and processes of many of these forests (Allen and others 2002, Covington and Moore 1994, Noss and others 1995, Swetnam and others 1999).

Initially, ecologists investigated the direct effects of reduction in fire frequency on species, communities, and ecosystems (e.g., Cooper 1960, Covington and Moore 1994, Marshall 1962). More recently, researchers recognized that exclusion was not only reducing fire frequency but also promoting stand-replacing crown fires (Covington and Moore 1994, Dahm and Geils 1997, Fulé and Covington 1996, Swetnam and others 1999, 2001). For many ecosystems characterized by presettlement surface fire regimes, these high-severity fires are outside the natural range of variation (Moore and others 1999, Swetnam and others 1999), and can cause long-term disruption, even transforming some forests into grasslands or shrublands (Allen and others 2002).

Effective restoration requires documentation of the presettlement and altered fire regimes and an understanding of the natural and altered dynamics of major forest tree species. Although vigorous debate continues on restoration strategies, these bases have been relatively well established for southwestern ponderosa pine ecosystems

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[1]An earlier version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.
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(Allen and others 2002). Knowledge of the fire ecology of Madrean pine-oak systems in the Southwest lags far behind. This is a widespread forest type that harbors tremendous biological diversity (DeBano and others 1995).

My goal is to contribute to an understanding of the fire-driven dynamics of Madrean pine-oak forests in order to promote effective fire management and restoration. I will synthesize my relevant past work (published and unpublished, solo and collaborative) from the Chiricahua Mountains, Arizona. The paper will focus on three themes: 1) presettlement dynamics of pine and oak species in response to frequent, low-severity surface fires, 2) changes in community and ecosystem attributes in response to reduced fire frequency, and 3) changes in community composition in response to anomalous, high-severity crown fires.

Study Site and Fire History

The work reported here was carried out in the Chiricahua Mountains, located in southeastern Arizona in Cochise County (31^{o} 52' N, 109^{o} 15' W), and considered part of the Sierra Madre "archipelago" (DeBano and others 1995). The mountains extend southeast to northwest for about 80 km and rise from about 1,100 to 3,000 m altitude. The climate is semi-arid, with two wet seasons, one in July-September, when more than 50 percent of total precipitation falls, and the second December-March. A pronounced dry season usually occurs between the final winter storms in March or April and the onset of the rainy season in July (Sellers and others 1985).

The Chiricahuas support a wide diversity of desert, Petran, and Madrean communities (Barton 1994, Brown 1982). This study focused on Madrean pine-oak forest (Mexican oak-pine woodland subtype #3, Brown 1982), the most abundant community type between 1,650 and 2,050 m. The major tree species are *Pinus leiophylla* (Chihuahua pine), *P. engelmannii* (Apache pine), *P. arizonica* (Arizona pine; *P. ponderosa* var. *arizonica*), *Quercus hypoleucoides* (silverleaf oak), *Q. arizonica* (Arizona white oak), and *Q. emoryi* (Emory oak).

A three-century fire history of Rhyolite Canyon in the Chiricahua Mountains (Swetnam and others 1989, 1992; see also Barton and others 2001) revealed a fire regime characterized by frequent (every 5 to 15 yr), low-severity surface fires, similar to results for other Madrean pine-oak forests (Fulé and Covington 1996, Swetnam and others 2001). Fires were often synchronous, suggesting extensive fires the length of the canyon. This inferred synchrony was interrupted, possibly as a result of flood events (Swetnam and others 1991), from 1801-1851, when fires ceased in most of middle Rhyolite but continued in lower Rhyolite. Fire frequency declined drastically beginning in the 1870s, probably as a result initially of intensive livestock grazing and then fire suppression (Bahre 1995, Barton and others 2001). Most presettlement fires occurred in late spring to early summer (Swetnam and others 2001), and during unusually dry years (Barton and others 2001).

Methods

This paper synthesizes data from four different sources. Details for each of these can be found in the references provided.

1) Population age structures were constructed for pines and *Quercus hypoleucoides* in lower Rhyolite Canyon (Barton 1999, Barton and others 2001), and for pines and *Pseudotsuga menziesii* in Cave Creek Canyon (see Barton 1994 for site

details). Increment cores were prepared and cross-dated using standard practices. Seedling growth was used to estimate age at coring height where appropriate.

2) To examine the relationship between tree radial growth and fire, ring widths were measured to the nearest 0.001 mm for 77 *P. arizonica* trees from Rhyolite Canyon that germinated before 1860 (Barton and others 2001).

3) I quantified the responses of two pines (*P. leiophylla* and *P. engelmannii*) and four oaks (*Q. hypoleucoides, Q. arizonica, Q. emoryi*, and *Q. rugosa*) to recent low-severity fires (Barton 1999). In each fire, which had occurred during the previous 10 years, data were collected along belt transects on top-kill, resprouting, and seedling establishment. Using this retrospective approach, I calculated stand data for each species for pre-fire, immediate post-fire, and actual sample period.

4) I collected similar data for *P. leiophylla*, *P. engelmannii*, and *Q. hypoleucoides* for two high-severity crown fires. Sampling was carried out in 1999 in the 10,330 ha 1994 Rattlesnake Fire and in 1992 in the 23 ha 1983 Methodist Fire (Barton 2002). Data were collected along belt transects, in areas of complete top-kill, on resprouting, height of sprouts, and number and height of seedlings.

Results and Discussion

Presettlement Madrean pine and oak dynamics

Pines and oaks contrasted in their response to low-severity fires characteristic of the presettlement fire regime. Pines were fire resistant, exhibiting high levels of top-survival, whereas oaks were fire resilient, exhibiting lower top-survival but prolific resprouting (*table 1*). *Pinus leiophylla* also resprouted after top-kill but at much lower levels than for oaks. As a result of these responses, low-severity fire favors pines in the short-term, but resprouting allows oaks to increase relative to pines during inter-fire periods (Barton 1999; see Fulé and others 1996, 1997).

Age structures reveal, similarly, that pine individuals persisted through many fires, whereas *Q. hypoleucoides* stems arose in a narrow window of time after the last series of frequent fires in 1850 to 1870 (*fig. 1*). Although pines survived fire well, their establishment was closely tied to periods of relatively low fire frequency (*fig. 1*). In fact, number of pine stems for a given decade is negatively correlated with number of subsequent fires but not correlated with number of past fires (Barton 1999, Barton and others 2001). These results suggest that, in addition to appropriate rainfall and seed availability, regeneration of these pines requires fire-free periods sufficient to allow growth to a height that provides resistance to subsequent low-severity fires.

These studies taken together (see also Fulé and others 1996, 1997, Rodriguez-Trejo and Fulé 2003), then, suggest a presettlement Madrean pine-oak forest in which occasional pulses of pines arose, many of which persisted in open stands for centuries through frequent surface fires. Oaks also probably persisted for long periods of time, but were forced to resprout repeatedly after top-kill from these fires, which maintained them at relatively small sizes and low cover levels.

Table 1—*Oak and pine top-survival, resprouting, and seedling establishment after low-severity and high-severity fires*[1].

Fire name	Fire severity	Top-survival Oaks	Top-survival Pines	Resprouting Oaks	Resprouting Pines[2]	Seedlings Oaks	Seedlings Pines
		pct (n)	pct (n)	pct (n)	pct (n)	# ha^{-1}	# ha^{-1}
Rhyolite III	Low	53.2 (124)	84.9 (33)	44.8 (58)	0.0 (5)	9680	5900
Rhyolite T	Low	39.6 (265)	61.1 (36)	56.2 (160)	0.0 (8)	1400	950
Methodist 1	Low	11.0 (154)	50.4 (141)	59.1 (137)	11.6 (43)	2140	640
Methodist 2	Low	18.8 (218)	63.7 (55)	43.5 (177)	12.5 (16)	980	480
Animals	Low	30.5 (128)	69.5 (95)	38.2 (89)	6.9 (29)	2690	30
Rattlesnake	High	0.0 (355)	0.0 (217)	90.1 (355)	23.2 (164)	150	77
Methodist	High	0.0 (63)	0.0 (55)	92.0 (63)	13.3 (15)	30	70

[1]Sampling carried out 3 to 10 yr post-fire in belt transects; for details on methods and data for individual oak, pine, and other species, see Barton (1999, 2002).
[2]*Pinus leiophylla* included only; *P. engelmannii* does not resprout.

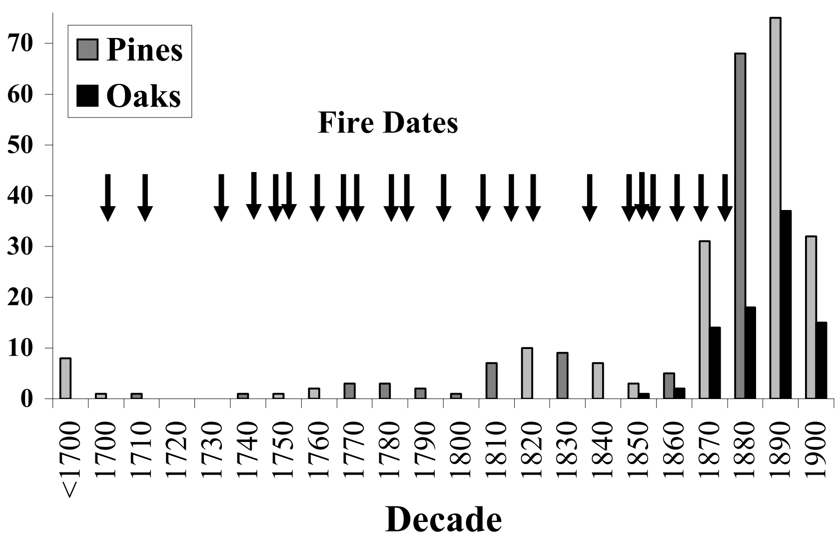

Figure 1—Age structure of pines (*Pinus engelmannii* and *P. leiophylla* combined) and oaks (*Quercus hypoleucoides*) in Rhyolite Canyon in relation to fire dates.

Community Response to Reduced Fire Frequency

Reduction in fire frequency has resulted in major changes in the structure and composition of Madrean pine-oak forests in the Chiricahuas. Age structures reveal large increases in the density of all species beginning in the late 1800s, coincident with the reduction in fire frequency (*fig. 1;* Fulé and others 1996, 1997; Rodriguez-Trejo and Fulé 2003). This suggests again that frequent presettlement fires maintained stands with more widely spaced trees than today (Mills 2002).

P. arizonica radial growth declined in the 20th century in association with fire reduction and increased stand density (Barton and others 2001). Superposed epoch analysis demonstrated that presettlement fires did not lead to growth releases (Barton and others 2001). However, similar to the 20th century, radial growth was significantly reduced during the 50 yr fire hiatus in middle Rhyolite compared to before or after, whereas no such differences occurred in lower Rhyolite where fires continued unabated (Barton and others 2001). Apparently, stands were generally so open during presettlement times that growth was stimulated through fire-caused thinning only after an unusually long inter-fire period.

Both age structures and responses to low-severity fires suggest that the ratio of pines to oaks was probably higher during presettlement times than today. Furthermore, frequent fire appeared to exclude *P. discolor* and *Pseudotsuga menziesii*, two relatively fire-sensitive species that have recently invaded some pine-oak sites (*fig. 2*). Interestingly, *P. menziesii* is a relatively shade tolerant species invading from moister, generally higher elevation sites, whereas *P. discolor* is a large-seeded, relatively shade tolerant pinyon (Barton 1993), invading from drier, generally lower elevation sites (Mills 2002).

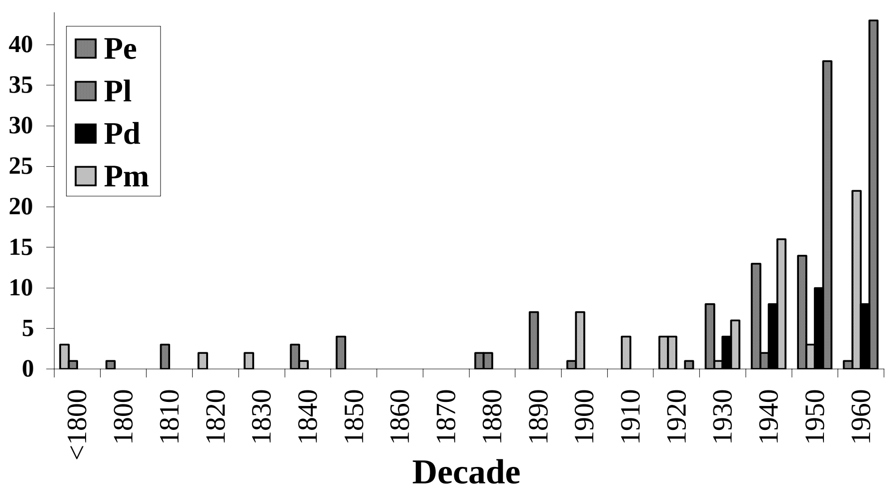

Figure 2—Age structure evidence of recent invasion of *Pseudotsuga menziesii* (Pm) and *Pinus discolor* (Pd) in a stand of *P. engelmannii* (Pe), *P. leiophylla* (Pl), and *Quercus hypoleucoides* in Cave Creek Canyon. All stems at least 2 m tall were cored in 1.75 ha, 0.5 ha, 0.3 ha, and 1 ha, respectively.

Community Response to Anomalous Crown Fires

In areas subject to complete top-kill (*table 1*), most oaks resprouted. *P. leiophylla* exhibited a much lower level of resprouting, which nonetheless could be an important source of pines. Seedlings were rare for all three species, much lower than after low-severity fires (*table 1*). Oak seedlings and sprouts were significantly taller than those of pines (Barton 2002). Several years after high-severity fire, then, fast-growing oak resprouts have formed a dense, coalescing canopy over scattered small pine seedlings and sprouts. These results suggest anomalous crown fires can transform ecologically complex Madrean pine-oak forests into more homogenous oak woodlands, perhaps for decades if not centuries (Fulé and others 2000).

Restoration of Madrean pine-oak forest

This synthesis reveals major changes in Madrean pine-oak forests resulting from a reduction in low-severity surface fires. Stands have become denser, especially for oaks, and an increase in high-severity crown fire poses long-term disruption. The consequences for species associated with these communities are likely to be serious. This synthesis argues strongly for restoration of Madrean pine-oak forests, especially the return of frequent surface fires. In designing restoration strategies, it is important to bear in mind that Madrean pine-oak forest varies tremendously in pine and oak composition over its range from the borderlands to far south in the Sierra Madre Occidental (Rodriguez-Trejo and Fulé 2003). Presettlement fire regimes also exhibited substantial variation over time and among mountain ranges (Swetnam and others 2001). This variability suggests the need for a variety of flexible restoration approaches (Allen and others 2002, Rodriguez-Trejo and Fulé 2003).

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National Environmental Policy Act Disclosure of Air Quality Impacts for Prescribed Fire Projects in National Forests in the Pacific Southwest Region[1]

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Abstract

A key purpose of the National Environmental Policy Act (NEPA) is to "promote efforts which will prevent or eliminate damage to the environment and biosphere and stimulate the health and welfare of man" (NEPA, Sec 2). The Council on Environmental Quality states "the NEPA process is intended to help public officials make decisions that are based on understanding of the environmental consequences, and take actions that protect, restore, and enhance the environment" (40 CFR Part 1500.1(c)). Smoke from prescribed fire affects air quality and has the potential to impact human health and quality of life. Public concern about the air quality impacts associated with prescribed burning has led to an increasing need to disclose these effects in environmental documents prepared under NEPA. Prescribed fire projects are subject to NEPA regulations (40 CFR 1500 -1508). Smoke from prescribed fires must meet Federal, State, and local air quality regulations. Environmental planning for prescribed fire activities must include careful consideration of air quality impacts and requirements. This paper describes information that should be considered to ensure that prescribed fire projects meet the requirements of NEPA.

Introduction

As managers in California's national forests plan prescribed burning projects, they consider three laws that contain provisions for managing air quality: 1) the National Environmental Policy Act (NEPA), 2) the Federal Clean Air Act, and 3) California State Clean Air Act. These laws establish the regulatory framework for assessing air quality impacts from prescribed burning projects in California's national forests. Federal actions must comply with these laws and implementing regulations.

National Environmental Policy Act

NEPA is the nation's basic charter for protecting the environment. NEPA declares that it is the Federal government's continuing policy to use all practical means and measures to create and maintain conditions where man and nature can exist in productive harmony and fulfill the social, economic, and other requirements of present and future generations of Americans. The Federal government is charged with cooperating with State and local governments and other concerned public and private organizations to achieve these goals.

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The Council on Environmental Quality (CEQ) has promulgated regulations (Title 40 of the Code of Federal Regulations, CFR, Part 1500) to implement NEPA. The CEQ regulations establish an issue-driven environmental analysis process that can be broadly summarized as follows: 1) the Federal agency, with help from other Federal, State, and local agencies and the public, identifies issues associated with implementing the proposed action, 2) the agency creates alternative ways of achieving the proposed action's objectives while addressing the issues, 3) the impacts of the proposed action and alternatives are evaluated and disclosed in the environmental document.

NEPA procedures ensure that environmental information is made available to public officials and citizens before decisions are made and before actions are taken. The regulations stress that the purpose of NEPA is to foster appropriate action on the part of Federal agencies. The NEPA process is intended to help public officials make decisions that are based on understanding environmental consequences and take actions that protect, restore and enhance the environment (40 CFR Part 1500.1(c)). The Forest Service's Environmental Policy and Procedures Handbook (Forest Service Handbook, FSH, 1909.15) provides additional direction regarding the Forest Service's policy and procedures for implementing NEPA and the CEQ regulations.

Federal Clean Air Act

The Clean Air Act is the nation's mandate for protecting and improving air quality. Forest Service managers consider two key elements of the Clean Air Act as they plan prescribed burning projects: 1) conformity with National Ambient Air Quality Standards and 2) protecting air quality related values in Class I Areas.

The U.S. Environmental Protection Agency (EPA) has established National Ambient Air Quality Standards (NAAQSs) for six criteria pollutants: particulate matter less than 10 and 2.5 microns in diameter (PM10, PM2.5), sulfur dioxide (SO2), nitrogen dioxide (NO2), ozone (O3), carbon monoxide (CO), and lead (Pb). An area that does not meet the NAAQS for any pollutant is designated a "non-attainment area" for that pollutant. Section 176 (c) of the Clean Air Act provides conformity rules, which prohibit Federal agencies from taking any action in a federal non-attainment area that: 1) causes or contributes to any new NAAQS violation, 2) increases the frequency or severity of an existing NAAQS violation, or 3) delays timely attainment of a NAAQS. Federal agencies must make a conformity determination whenever they conduct or approve a resource management project, such as a prescribed burning project, in a federal non-attainment area. Federal managers should work with local air quality regulators during development of State Implementation Plans (SIPs) for non-attainment areas to ensure that projected emissions from prescribed burning activities are included in the SIP emissions inventory. This will facilitate subsequent conformity determinations.

Class I areas include international parks, national parks larger than 6,000 ac, national wilderness areas larger than 5,000 ac, and national wildlife refuges in existence as of August 17, 1977. Federal land managers are responsible for protecting air quality related values (AQRVs) in Class I areas from the adverse effects of air pollution. AQRVs are features or properties of a Class I Area that can be affected by air pollution, such as flora, water, soil, and visibility. Final Regional Haze Rules (July 1999) allow no degradation of visibility in Class I areas during 20 percent of highest visibility days.

California Clean Air Act

The State of California's charter for improving air quality is embodied in the State's Clean Air Act. Managers planning prescribed burning projects in California national forests must comply with the newly revised smoke management guidelines in Title 17 of the California Code of Regulations. Another key consideration in planning prescribed burning projects is California's Nuisance Rule, which states, "No person will discharge from any source whatsoever such quantities of air contaminants or other material which cause injury, detriment, nuisance, or annoyance to any considerable number of persons or to the public or which endanger the comfort, repose, health, or safety of any such persons or the public or which cause, or which have a natural tendency to cause, injury or damage to business or property." Providing local air quality regulators with draft environmental documents for review ensures that issues related to State and local regulations can be proactively addressed during the planning process.

Relationship between the National Environmental Policy Act and Air Quality Laws and Regulations

NEPA requires Federal agencies to analyze the environmental impacts associated with a proposed action as well as alternatives to the proposed action. Once the environmental analysis has been completed, the responsible Federal official documents his or her decision regarding the proposed action, alternatives to the proposed action, and environmental impacts. The decision document includes findings required by other laws, including air quality laws. The environmental analysis provides the basis for these findings.

This paper describes NEPA procedures for documenting environmental analyses related to air quality impacts for prescribed burning projects in California National Forests. Relevant air quality laws and regulations are discussed under appropriate sections of the environmental analysis document, providing an integrated framework for ensuring that environmental analyses for prescribed burning projects comply with NEPA as well as State and Federal air quality laws and regulations.

Appropriate Environmental Disclosure Document

Federal land managers consider applying NEPA whenever they propose an action. Forest Service managers look to two key references to determine when and how to apply NEPA to their project proposals: 1) CEQ regulations for implementing NEPA (Title 40 of the Code of Federal Regulations (CFR), Parts 1500 through 1508) and 2) Forest Service Environmental Policy and Procedures Handbook (FSH, Part 1909.15).

Federal agencies use three different levels of environmental analysis and documentation for project proposals: 1) categorical exclusion (CE), 2) environmental assessment (EA), and 3) environmental impact statement (EIS). Categorical exclusions are used to document environmental analyses for routine administrative, maintenance, or other projects that normally do not have significant effects on the quality of the human environment. As potential impacts of a project become more extensive and intensive, the environmental analysis becomes more extensive and detailed. If an agency determines that an action will have significant impacts, as defined in 40 CFR Part 1508.27, the agency must prepare an EIS.

Key features common to all three levels of environmental analysis include: 1) a systematic, interdisciplinary approach for conducting the environmental analysis, 2) scoping, which is an early and open process for determining the range of issues to be addressed and for identifying the significant issues related to the proposed action, 3) written documentation of the environmental analysis process, and 4) documentation of the decision. Some routine administrative and maintenance activities are exempt from documentation requirements. Chapter 30 of FSH 1909.15 identifies these types of activities, which are categorically excluded and do not require a project file or decision memo. The appropriate level of environmental analysis is often clear; however, the dividing line between each level is not entirely distinct. The Forest Service line officer, acting as the responsible Federal official, relies on input from the project's interdisciplinary team and scoping to determine the level of analysis.

Categorical Exclusion

Projects likely to have minimal environmental impacts are typically documented in a categorical exclusion. Managers must determine whether a project falls within a category for exclusion (described in FSH 1909.15, Chapter 30.). In addition, small prescribed burning projects of limited duration that meet the following criteria can typically be documented under a categorical exclusion.: 1) the proposed project has few, if any, local, State and Federal government concerns or public controversy, 2) the relative "fire risk" is low to moderate, 3) the project is expected to produce limited amounts of smoke, and emissions are projected to fall within State and local guidelines. Projected emissions are of short duration, will not affect non-attainment areas or adversely affect Class I areas. (Title 17 suggests prescribed burning projects that treat less than 10 ac and generate less than 1 ton of PM10 meet these criteria.)

If the project meets the above criteria, managers should determine whether any extraordinary circumstances exist. An example of an extraordinary circumstance would be proximity to a sensitive area, such as a school, hospital, or Class I area (extraordinary circumstances are defined in FSH 1909.15). If there are no extraordinary circumstances, managers may proceed with a categorical exclusion. If extraordinary circumstances exist, managers must determine their potential significance by assessing the magnitude of the potential effects. If extraordinary circumstances exist, but the effects are considered minor in nature, the proposed action(s) may be categorically excluded. The responsible official ultimately decides whether a higher level of environmental analysis is warranted.

Environmental Assessment

Prescribed burning projects that are more extensive and intensive than those covered under a categorical exclusion can often be analyzed in an environmental assessment. An environmental assessment is prepared when a proposed project is anticipated not to have significant environmental impacts. For prescribed fire projects, managers must determine whether air quality impacts are likely to be significant. The presence of nearby sensitive receptors or nearness to a non-attainment area is a key criterion in assessing significance (criteria for significant impacts are defined in 40 CFR Part 1508.27). Throughout the scoping and environmental analysis process, the interdisciplinary team maintains ongoing discussions with the responsible official to assess whether any effects are potentially significant. If at any time it appears that effects may be significant, preparation of an EIS should be considered.

Environmental Impact Statement

A project with significant impacts, as defined in 40 CFR Part 1508.27, must be analyzed in an EIS. General guidelines for deciding whether an impact may be significant or needs further analysis include the following: 1) the project is highly controversial and likely to receive intense public scrutiny, 2) the project is located near a Class I area and activities are likely to impact visibility during “high visibility days” as defined in the Regional Haze Rules, 3) the project is viewed as precedent setting, for either air quality impacts or another aspect of its environmental consequences, 4) the project is located near key historical or cultural resources, parks or campgrounds, high-use recreational areas, or other sensitive areas, 5) the project is located in or near a non-attainment area, 6) the area has had a history of nuisance calls (CH2M Hill 1995).

Content of Environmental Assessments and Environmental Impact Statements

The general format for an EA or EIS includes four key chapters: 1) Purpose of and Need for Action; 2) Alternatives, including the Proposed Action; 3) Affected Environment; and 4) Environmental Consequences. Chapters 3 and 4 are often combined into a single chapter. The CEQ regulations (40 CFR Part 1502.10) provide additional requirements for formatting an EIS, including summary, table of contents, list of preparers, index, appendices, and so forth.

Chapter 1 Purpose of and Need for Action

The first chapter of the EA or EIS explains why the agency is considering an action. Existing conditions are compared with desired conditions to establish the need for the agency to take action. For example, if existing fuel levels exceed desired levels, the agency can establish a need to take action to reduce fuel loading. The purpose of the project is to meet management objectives in the national forest’s land and resource management plan.

Chapter 1 briefly summarizes the proposed action, explaining who wants to take action and when and how. It identifies the decision to be made; summarizes scoping efforts, including public involvement; and identifies significant issues used to formulate alternatives to the proposed action. An issue is a point of discussion, debate, or dispute. A debate about the effects of smoke from prescribed burning may lead to the formulation of alternative approaches for meeting the project’s objectives. Scoping and significant issues are sometimes described at the beginning of Chapter 2.

Chapter 2 Alternatives, Including the Proposed Action

The CEQ regulations (40 CFR Part 1502.14) describe Chapter 2 as the heart of the environmental document. Chapter 2 does more than simply describe the alternatives–it sharply describes differences between alternatives, particularly in terms of how their environmental impacts differ.

Chapter 2 describes the alternative development process, alternatives considered in detail, alternatives considered but not given detailed study, and environmental impacts of the alternatives. At a minimum, two alternatives are considered: the proposed action and the no action alternative. Significant issues provide the basis for the other alternatives. Title 17 of the California Code of Regulations requires consideration of a non-burning alternative for prescribed burning projects that propose to burn over 250 ac or over 100 ac near sensitive areas. All action alternatives must meet the purpose of and need for action described in Chapter 1.

Activities Common to All Alternatives

Activities common to all action alternatives are typically described in a single section and may include standard operating procedures for mitigation or monitoring.

Mitigation Measures are actions that avoid, minimize, rectify, reduce, eliminate, or compensate for impacts (40 CFR Part 1508.20). Mitigation techniques can be used to reduce smoke intrusions or smoke concentrations in sensitive areas. Mitigation measures also include items commonly described in burn plans, such as prescription windows and contingency measures.

Monitoring is used to determine whether the implemented alternative met the project's objectives and validate assumptions used to develop and analyze the implemented alternative. Real-time PM10 and/or PM2.5 monitoring of prescribed burning projects can be helpful in responding to nuisance calls as well as to ensure that NAAQS are not violated.

Unique Activities of Each Alternative

Activities unique to each alternative may include: 1) type, size, and amount of fuels to be treated; 2) acres to be treated in a specific manner; 3) time frames for burning; 4) general meteorological and fuel moisture conditions under which burning would occur.

No Action Alternative for prescribed burning projects is typically one that continues existing ongoing management activities but does not implement the proposed project. Wildland fire suppression activities would continue according to the specific land management plan. Some forest plans now have amendments that describe when and how a wildland fire will not be suppressed, and these should be described if applicable.

Action alternatives are designed to meet the purpose and need for action and respond to significant issues identified during scoping. These alternatives can explore different types of burns as well as other treatment methods. Newly revised Title 17 (2000) "Smoke Management Guidelines for Agricultural and Prescribed Burning" by California Air Resources Board (CARB) require consideration of "Alternatives to Burning" in NEPA documents for prescribed burns. Types of burns are dictated by the project objectives.

Comparison of Alternatives

This section compares and contrasts the alternatives based on how they meet the purpose and need and how they address the significant issues identified in Chapter 1. Air quality effects are displayed by comparing emissions generated under each alternative. The comparison, which is typically summarized in a table, is derived from the effects analysis presented in Chapter 4 "Environmental Consequences."

Chapter 3 Affected Environment

The "Affected Environment" chapter describes existing conditions in and around the project area. Key topics related to air quality include the following: 1) fuel types and fuel loadings; 2) sensitive smoke receptors are areas that could be impacted by the proposed burning activity and are considered sensitive due to legislation or public concerns (examples of sensitive smoke receptors are Class I areas, non-attainment areas, major transportation corridors nearby or downwind, hospitals, schools, and population centers); 3) regulatory environment–Agencies' current rules referring attainment status; 4) air quality conditions and trends, starting with the

County Emissions Inventory issued by the California Air Resources Board (USDA Forest Service); 5) typical activities on adjacent lands that could impact air quality; 6) visibility conditions; 7) geography and climatic influences; 8) proximity to non-attainment areas, based on the Conformity Handbook (USDA Forest Service).

Chapter 4 Environmental Consequences

The "Environmental Consequences" chapter forms the scientific and analytical basis for comparing the environmental impacts of the alternatives. The air quality section of this chapter should address the following topics: 1) methodology for analyzing air quality impacts, 2) direct and indirect effects on air quality under each alternative, 3) cumulative effects on air quality under each alternative, 4) a conformity determination if the project is in a non-attainment area for any NAAQS (Conformity Handbook), and 5) potential visibility impacts to Class I areas.

The methodology discussion should describe measures used to compare the alternatives (for example, emissions generated, visibility, and so forth). It should describe modeling methods used in the analysis and any technical assumptions or limitations. A worst-case scenario analysis should be conducted for multiple-day burns, using NFSPUFF or other dispersion models to verify that NAAQSs will not be violated.

Direct effects are caused by the action and occur at the same time and place as the action. Indirect effects are caused by the action but occur later in time or farther removed in distance than the action. The air quality analysis typically includes: 1) fuel consumption by alternative, 2) emissions (PM10/2.5 and NOx) generated under each alternative and 3) visibility impacts by alternative. Cumulative effects include potential air quality impacts from the project when combined with other present and reasonably foreseeable future actions.

A conformity determination must be made if the prescribed burning project is in a non-attainment area. The determination is based on total emissions analysis and air quality modeling to show that the project does not cause or contribute to the violation of any standard or increase the severity or frequency of existing violations.

Summary

Environmental analysis is all about fully disclosing environmental impacts that includes air quality impacts. Environmental analysis informs agency decision makers and the public about environmental trade-offs associated with alternative ways of meeting the project objectives. Air pollutants are generated during wildland burning. NEPA procedures require complete documentation of the analysis; if any aspect of the analysis is not documented, the responsible Federal official has not considered it in making the decision.

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Fire Management Over Large Landscapes: A Hierarchical Approach[1]

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Abstract:
Management planning for fires becomes increasingly difficult as scale increases. Stratification provides land managers with multiple scales in which to prepare plans. Using statistical techniques, Geographic Information Systems (GIS), and meetings with land managers, we divided a large landscape of over 2 million acres (White Sands Missile Range) into parcels useful in fire management. Within this hierarchy, 8 fire management areas (FMA), with major roads or drainages forming boundaries, were identified. Within these FMAs were 106 burn units (BU) defined by natural or human fire breaks. BUs can be used as prescribed burn areas or maximum allowable perimeters for fires. An intermediate level was needed between these two units. Using cluster analysis of vegetation data, 22 fire management zones were identified. Using this hierarchy, a large-scale vegetation map, and GIS, we were able to incorporate simplistic risk models based on fuel models, topography, and sites of concern for natural resource managers. The resulting maps provide a landscape view of the fire risk to natural and human resources and allow land managers to take a heads up approach in preventative fire management.

Introduction

Fire has been a major factor influencing the ecology, evolution, and biogeography of many vegetation community structures (Humphrey 1974, Bock and Bock 1988, Ford and McPherson 1996). The semi-desert grasslands and shrublands of the Southwest have evolved with fires caused by lightning strikes since the Pleistocene (Pyne 1982). Fires have played an important role in maintaining grasslands while reducing shrub invasion (Valentine 1971). Vegetation in New Mexico has developed under a fire influence over the past 10,000 years (Betancourt and others 1990, Anderson and Shafer 1991). Today, a proliferation of extreme fires is evident, because moderate fires can be controlled and only those burning in severe conditions and under dangerous circumstances impact the landscape, causing the most damage (Baisan and Swetnam 1997). The impact on the ecosystem depends on the current biological and physical environment and past land use patterns (Ford and McPherson 1996).

Little was known about the fire ecology on White Sands Missile Range (WSMR) and the deserts of the southwestern United States. This paper presents the results of an analysis that used existing data and incorporated Geographical Information Systems (GIS) to facilitate fire management planning on WSMR, providing fire personnel with a landscape view of fire risk.

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Study Area

White Sands Missile Range covers over 828,800 ha (2,210,117 ac) in south central New Mexico. Terrain includes steep, rocky mountains, steep to moderate footslopes, level to rolling grasslands, dunes, lava flows, and salt flats. The maximum elevation is 2,783 m at Salinas Peak. Rainfall typically occurs during the summer (July, August and September) as short, intense, localized storm events that account for 55 percent to 64 percent of the total annual precipitation (Barlow and others 1983). The intensity of rains generally results in massive runoff and very little infiltration, especially in steep areas and areas with exposed bedrock. Average precipitation varies from 200 mm in the basin to 400 mm in the mountains.

Methods and Results

Fire Management Units

The first step was to create fire management units that could be used within fire planning and fire management activities. Meetings were held with WSMR fire and environmental personnel to discuss creating fire management units. Through an iterative process we derived 106 zones based on roads and topography. These 106 zones were identified within four large geographic areas of the Range determined by the White Sands Fire Department (WS-ES-F). The center point for these four areas was roughly the middle of the range and corresponded to logistical constraints regarding response time.

These zone delineations were brought into a GIS (ArcView) by intersecting the appropriate digital elevation model (DEMs) derived contour lines, road, and stream coverages. The initial four areas were then differentiated into eight Fire Management Areas (FMA) after discussing terrain and response time with WSMR personnel. The four FMA's included lowland and mountain sites, with increased response times assumed for mountainous regions. The division into eight FMAs created four FMAs predominantly of lowland grassland and shrubland and four FMAs comprised of desert montane vegetation.

These two levels of fire management unit are useful, but because of the generality of the eight FMAs and the difficulty in placing management decisions on 106 Burn Units (BUs), an additional step in the hierarchy was developed. This unit allows analysis and management to be applicable at a scale that was beneficial for fire management planning. These intermediate fire management units (Fire Management Zones) were derived by conducting a cluster analysis using New Mexico Natural Heritage Program (NMNHP) land cover map. The area of the Level 2 community types from the within each BU were derived and used for the cluster analysis. Thus Fire Management Zones (FMZs) were created by the grouping of BUs with similar vegetation.

Fire Modeling

Modeling potential fire risk is dependent on basic coverages that include vegetation, slope, aspect, and natural and human created firebreaks. The vegetation can then be associated with particular fire behavior models. A rating model was derived for all coverages. Ratings were placed on the variables to allow a final score to be placed on each grid cell (Caprio and others 1998). Areas with increased potential for rate of spread, fuel loading, human structures, and sensitive biological communities were given higher weights and values. Ratings are presented in the following paragraphs for each of the coverages used within that model. Cell ratings were then compared to identify relative risk of fire based on all coverages derived for this project that were applicable to fire risk. Attribute ratings ranged from 0 to10.

Vegetation maps created by the New Mexico Natural Heritage Program (NMNHP) (Muldavin and others 1997) and the New Mexico Gap Analysis Project (NM-GAP) (Thompson and others 1996) were cross-walked by fire behavior fuel models (Albini 1976, Anderson 1982). The NMNHP mapped two levels of community corresponding to the alliance (34 types) and association (95 types) levels of the National Vegetation Classification System (NVCS) created by NatureServe (2002). The NM-GAP coverage mapped 19 types of vegetation to the formation level of the NVCS on or adjacent to WSMR. Because of the differing resolutions, all three vegetation maps were used for different analyses and modeling. The NMNHP maps are more detailed with greater accuracy and provide for more thorough analyses, but are limited to the boundary of WSMR. The NM-GAP map allows the incorporation of the adjacent lands into any analysis to bring a contextual aspect of fire ecology.

The standard fire behavior models (Anderson 1982) that were applicable to WSMR were identified by the main fuel component such as grass, shrubs, forest/woodland, or slash. Models were identified based on the hazard potential given with regards to WSMR and rated based on potential fire intensity (*table 1*).

***Table 1**. Classification of fuel models for fire risk modeling on White Sands Missile Range, New Mexico.*

Fuel Model	Typical Description	Level 1 Classification	Rating
1	Short grass (1 ft)	Grama grass grasslands	2
2	Timber (grass and understory)	Ponderosa pine and Juniper savannas	2
3	Tall grass (2.5 ft)	Alkali Sacaton	2
5	Brush (2 ft)	Shrublands	4
6	Dormant brush, hardwood slash	Montane scrub and Interior Chaparral	4
8	Closed timber litter	Ponderosa Pine, Piñon pine with grass understory	8
9	Hardwood litter	Can be used for Piñon-Juniper areas such as those on the Oscura Mountains	8
No Fuel Model			0

Slope was derived from DEMs and reclassified based on associated risks ($<10^o$=0; 10 to 20^o=2; 20 to 30^o=4; 30 to 40^o=7; $>40^o$=10). The aspect coverage was derived and reclassified based on associated risks (flat=0; SW,S,SE=2; W,E=6; NE, N, NW=10). On WSMR, many of the northeast-facing drainages along the mountains have high quantities of oak and other more mesic plant species that add to the fuel loading.

WS-ES-ES personnel identified biological and human resources of concern. These ranged from specific species (threatened, endangered, and sensitive) to habitats that are either limited in the southwest (e.g., riparian) or limited on WSMR (piñon-juniper community in Oscura Mountains). Human resources included existing structures and historical structures.

Habitats of concern included species-specific habitat pertaining to threatened, endangered, sensitive, and rare species and other habitat types. These habitat types include alkali sacaton (*Sporobolus airoides*); interior chaparral; piñon (*Pinus edulis*)-juniper (*Juniperus monosperma*) woodland of Oscura Mountains; ponderosa pine (*Pinus ponderosa*) at Salinas Peak; black grama grasslands (*Bouteloua eriopoda*); eastern valley grasslands of blue (*Bouteloua gracilis*) and hairy grama (*B. hirsuta*); juniper (*Juniperus monosperma*) encroachment in all grasslands; San Andres cactus community; cryptogamic soils; raptors on escarpments of mountains; wolf reintroduction areas; previous burn areas; springs and seeps; wetland habitats; and wildlife water units. These vegetation communities of concern were identified where possible from the New Mexico Natural Heritage Program land cover map. Specific areas such as the San Andres cactus community and cryptogamic soils are not identified because of lack of spatial data. All 3 land cover coverages were modified and concern communities labeled accordingly.

Point coverages of other biologically significant components were obtained including springs, seeps, and wildlife water units. Springs and seeps are important because these sites provide refugia for many wildlife species. These areas were buffered by 500 meters in order to clearly identify a defensible area. These polygons of buffers were than rasterized and each site given a rating of 10 for their ecological importance.

Similar to the biological points, point coverages of human components such as facilities, archeological sites, historical sites, and telephone poles were obtained. These point coverages were also buffered at 500 m to provide a defensible area and the polygons rasterized for inclusion into the modeling process. These areas were given a rating of 10.

Using grid coverages of each model variable previously discussed, we combined the grid values in an arithmetic overlay. This provided a value for each pixel based on the risk assessment of each input. After the initial model run, the slope and aspect variables were modified by multiplying the rating for slope by 1.5 and the rating of aspect by 0.5. This was done to eliminate an undue bias of the aspect modeling factor on model and give more importance to the slope factor. The model was run once for each of the three land cover coverages available.

Risk values for each model ranged from 3 to 45 out a possible high 50. For fire management purposes we classified these risk values based on natural breaks in the data as identified within the GIS software. This provided three levels of relative fire risk with low (3 to 13), moderate (14 to 22), and high (23 to 45).

Conclusions

Fire Management Units

WSMR was stratified into fire management units at three geographical scales to aid in the future development of a fire management plan. These units allow scale appropriate management strategies to be put into place. FMAs are large areas with major roads or drainages acting as the boundaries. FMZs are areas within these FMAs that were delineated based on ecological factors (e.g., vegetation, topography) as well as incorporation of some fire management and WSMR infrastructure logistical constraints such as roads and firebreaks. BUs are the smallest units and can be used for prescribed burns because either natural or human created firebreaks generally surround them. These BUs can also be utilized as the maximum allowable perimeter for wildfires.

Stratification by FMA, FMZ and BU provides a level of detail for fire management. On WSMR, one can view the number of endangered, threatened, rare, and sensitive species that occur within the fire management units, thus providing a planning level tool for prescribed fires and other ecological needs. Through this process it was identified that the habitat for 41 species could occur within one FMA. Similar analysis can be done with springs and associated riparian areas. Two FMZs had the overwhelming number of springs with 110 and 73. The number of wildlife water units or other biologically significant sites can also be analyzed and that information brought into fire planning

Fire Modeling

The fire risk modeling provides a relative risk ranking for the entire landscape of WSMR using a scale from 3 to 45, with higher numbers having a higher risk for intensive burning and extreme fire behavior. The resulting grids can be used with a multiple number of classifications depending on the needs of the fire managers. I presented these based on three levels of classification based on natural breaks within the data. This classification was normalized over all three grids to present a comparison of the models in regards to the spatial scales.

Plant communities and fuels are based on the NMNHP land cover maps. Basing fuel behaviors and fuel loadings on these remotely sensed land cover maps can be problematic. Remotely sensed data generally do not distinguish between high and low fuels loads. The purpose of these maps is to present communities. Therefore determination of fuel loading, fire behavior, and risk assessment using these coverages must be viewed as the average of that variable associated with that community type. This study presents fire ecology at a landscape level, and even at the most refined scale (Burn Unit) fuel loadings are quite variable, as are plant communities. It is difficult to quantify fuels in these large areas because of variability and cannot be discussed adequately without going into more site-specific detail.

Risk models will change through time as natural and human fire breaks change and military missions change the landscape of WSMR. Also fires, natural or management ignited, will change the community types that were modeled. Using the standard model created for this project and modifying it through time will allow this model to remain dynamic.

The risk models validate many concerns expressed by WSMR personnel during meetings. The piñon-juniper communities in the Oscura Mountains were identified as concern areas because of the threat of a stand replacement fire. Risk models as each thematic scale indicate that this area is at a relatively higher risk then most of the range. Although WSMR personnel drove some aspects of the modeling, it should be viewed as independent because of the additional variables included within modeling.

Overall

Fire plays a major role in many of the vegetation communities that occur on White Sands Missile Range. These communities on WSMR are structurally different from similar communities in the Southwest because of the accumulation of fine fuels and high densities of closed canopy piñon woodlands. The impact of fire suppression on these fuel loads on WSMR cannot be determined because comprehensive historical fire suppression documentation is not available. Historical reports that do exist indicate that many fires were actively attacked, but the degree and success in which those fires were observed, located, and controlled is unknown.

The information generated by this study and the fire management plan will enable WSMR to strengthen their fire management program. The products derived from this work can be used in conjunction with the LCTA monitoring program that provides baseline and postfire data to evaluate and understand the fire program and its ecological context within the Southwest.

The fire modeling and risk assessment conducted in this study provides basic spatial information for management decisions regarding fires. The fire management units (fire management areas, fire management zones, and burn units) provide a method for stratification of these management decisions.

Finally, any fire management of a large area such as WSMR must take into account the boundary of that parcel with private, state and federal landowners. Fires near these boundaries take on further complications as management and suppression response should be reviewed with those individuals or entities. A landscape approach such as presented here can identify the initial threats to such a fire and provide fire managers with tools for effective communication and management should that occasion arise.

Acknowledgments

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Communication and Implementation of GIS Data in Fire Management: A Case Study[1]

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Abstract

Remotely sensed data and Geographical Information Systems (GIS) can be an effective tool in fire management. For the inclusion of these tools, fire management and research personnel must be effective in communication regarding needs and limitations of the data and implementing that data at various scales. A number of personnel can be involved within fire management including land managers, landowners, natural resource professionals, and researchers. Remotely sensed data and GIS can be an effective tool for bridging those involved in fire management. One barrier can be the lack of communication or understanding as to what individual entities provide. We present a case study of the Walnut Fire Complex involving the Southwest Regional Gap Analysis Project (SWReGAP), Bureau of Land Management (BLM), and New Mexico State Forestry. Each entity has different objectives, but effective communication can allow specific data to be incorporated into management. Data obtained from BLM and New Mexico State Forestry and provide SWReGAP with additional data for an enhanced land cover map. This map can then be used as a tool for large-scale fire management as presented in the case study.

Introduction

Wildfire management responsibility has evolved from a "save the local community - protect your own backyard" and let the backcountry burn philosophy of the late 19th century to a full-scale industry in the early part of the 21st century. This change was caused mainly by past practices of total wildfire management and increase of urban/interface growth throughout the west. Wildfires that threatened the communities of the old west (prior to the 1920s) were quickly suppressed because of the fear of catastrophic fires burning entire towns and destroying the lumber, fuel wood, pastures and other natural resources needed for communities to survive.

When the U.S. Forest Service was created in the late 1800s, this protection responsibility was one of its primary functions. Other agencies and entities soon accepted responsibility for wildfire suppression, as these duties became part of their enabling legislation. In New Mexico this was no different. The Taylor Grazing Act (1934) provided the emphasis for the BLM and other agencies of the U.S. Department of the Interior to begin fire management. State Forestry organizations (e.g., New Mexico), State Land Offices (e.g., Arizona), or local sheriff's departments (e.g., Colorado) were statutorily given responsibility for wildfire suppression on

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private and state lands. This is now a complex situation because these agencies and organizations must interact due to the complex nature of wildfire management. Total wildfire suppression has shifted to wildfire management in which not all wildfires are suppressed. This meets a paradigm shift in natural resource management strategies.

Another paradigm shift has included technological advances in fire management and fire planning. One such technological advance is the use of geographic information systems (GIS) and remotely sensed data. The use of these technologies has further advanced fire management and planning. The abundance of data and availability of software has allowed these tools to become commonplace not only within the office but in the field before, during, and after fires. Critical to proper use and incorporation of these technologies is the understanding of their limitations.

Limitations to the implementation of GIS data are centered on the concepts of scale and resolution. This includes not only the actual differences in scale between datasets, but also the resolution at which datasets were derived. Scale refers to the representation of the data based on the actual features on the ground. The detail of a large scale data layer (e.g., 1:24,000 topographical map) and smaller scales (e.g., 1:100,000 land status map) have been used for years in fire management. As a smaller scale is used some detail is lost in favor of displaying a larger landscape in which to put the management in context. Resolution refers to the size of the pixel used within imagery. There are many remotely sensed imagery products available for use including Enhanced Thematic Mapper Plus (ETM+) and Digital orthophotoquads (DOQ). ETM+ is derived from a satellite that acquires images of the same area every 16 days. The resolution of the images obtained is 30 m (30 x 30 m pixel). At finer scales there are DOQs or georectified aerial photographs. These can have a resolution of 1m (pixel=1 x 1 m).

Setting the Stage

In southwestern New Mexico, the complex scenario of interagency agreements is somewhat simplified because there are a limited number of participants. The USDA Forest Service is represented by the Gila National Forest, with responsibility for over 3.3 million ac (1.3 million ha). The BLM is represented by two field offices. The Las Cruces Field office is responsible for over 5.5 million ac (2.2 million ha). The Socorro Field office is responsible for over 1.5 million ac (600,000 ha). Three National Wildlife Refuges are also represented. White Sands Missile Range (Department of Defense) is represented, but due to the secure nature of work at the range, limited assistance is needed. The remaining state and private land falls under the jurisdiction of the New Mexico Energy, Minerals, and Natural Resources Department, Forestry Division, Socorro District, which maintains joint power agreements with all the federal agencies, along with local county governments, to supply suppression resources to agencies when needed.

Wildfire management has to be based on recent and accurate mapping of not only jurisdictional boundaries, but also vegetation and fuel boundaries. A current field map with this information would provide wildfire managers on the ground as well as in dispatch and command type settings with valuable information. Currently, no regionally consistent land cover maps exist for the five southwestern states. This lack of regional data causes planners to rely on various maps often with differing scale and resolution. The SWReGAP objective of a regional land cover data set will provide this data layer (Boykin and others 2000).

Maps derived from satellite imagery have several limitations when applied to fire management and other natural resource endeavors. Scale and resolution of the map, both thematically and spatially, may or may not be sufficient for various aspects of fire management. Accuracy is also variable depending on the land cover mapped and the methods used. Land cover maps do not make calculations regarding fuel loads, canopy cover, or stem density. These are implied based on the thematic classification of the pixel, but variation exists and fire managers must be aware of these assumptions.

We focus on three cooperators in southwestern New Mexico with varying objectives from ecosystem sustainability to fire suppression to conservation of plant and animal species. These cooperators are BLM, New Mexico Energy, Minerals, and Natural Resources Department, Forestry Division (State Forestry), and the New Mexico Project of the SWReGAP. We discuss the cooperative integration between the three for current fire management and future fire planning.

Objectives

Bureau of Land Management

The BLM goal of fire management is based on using the full range of fire management activities to achieve ecosystem sustainability, including interrelated ecological, economic and social components. The BLM recognizes fire as a critical natural process and integrates fire into land and resource management plans and activities on a landscape scale, and across agency boundaries. The BLM response to a wildland fire is based on ecological, social, and legal consequences of the fire. The circumstances under which a fire occurs, and the likely consequences on firefighter and public safety and welfare, natural and cultural resources, and values to be protected dictate the appropriate management response to fire. The BLM uses wildland fire to protect, maintain, and enhance resources and, as nearly as possible, be allowed to function in its natural ecological role. The use of fire is based on approved Fire Management Plans and follows specific prescriptions contained in operational plans. Protection of human life is the single, overriding priority. Setting priorities among protecting human communities and community infrastructure, other property and improvements, and natural and cultural resources is based on the values to be protected, human health and safety, and the costs of protection. When people have been committed to an incident, these human values become the highest value to be protected.

New Mexico State Forestry

State Forestry is statutorily responsible for fire management on all private and state lands. Suppression strategies are based on the value of the resources at risk (structures, natural resources, watersheds, timber, grazing, recreation, etc.) and all tactics are determined by using the appropriate suppression response (control, confine, or contain) with safety to firefighters and the general public the major concern. Cost effectiveness is also important. In areas where wildfire management plans have been completed and approved by the landowners or New Mexico State Land Office, suppression is replaced by aggressive management where possible if all involved have agreed to the objectives and public safety and cost effectiveness is insured.

Southwest Regional Gap Analysis Project

The SWReGAP is a regional update of the National Gap Analysis Program's mapping and assessment of biodiversity for the five state region of Arizona, Colorado, Nevada, New Mexico, and Utah. It is a multi-institutional cooperative effort coordinated by the U.S. Geological Survey. The primary objective is to use coordinated mapping approaches to create detailed, seamless GIS datasets of land cover, predicted habitat for native terrestrial vertebrate species other than fish, land stewardship, and management status, and to analyze this information to identify those biotic elements that are underrepresented on lands managed for their long term conservation (gaps). Creation of a seamless land cover dataset is underway using the National Vegetation Classification System (NatureServe 2000) and labeling land cover based on ecological systems and alliances (Boykin and others 2000).

Case Study–Walnut Fire Complex

The Walnut Fire occurred in 2002 in southern Hidalgo County in extreme southwestern New Mexico, in an area commonly called the bootheel. The southern half of the county is made up of several medium sized ranches (approximately 5,000 ac) along with one large (over 300,000 ac of deeded land) ranch known as the Gray Ranch. This ranch has had a long history, at one time being part of the Diamond A Cattle Company that encompassed a large part of southern New Mexico. In 1992, the ranch was sold to the Nature Conservancy who then sold it to the non-profit Animas Foundation. Due to this ranch's rich and diverse habitat types, species and location, the new ownership wanted to return natural wildfire to the landscape. This management strategy was different than past management which grazed up to 16,000 head of livestock and suppressed any wildfire. This new strategy caused concern from smaller ranchers who maintained wildfires had no boundaries and felt threatened by this change in management.

This caused formation of the Malpais Borderlands Group, a group of local ranchers and Animas Foundation individuals. The group then developed a wildfire management map that designated let-burn areas, suppress areas, and contain areas. In 1997, State Forestry, with assistance from the Las Cruces Field Office of the BLM, completed the Bootheel Wildfire Management Plan (Boykin 1997), that explained and delineated these areas. Fire Management Areas (FMAs) were created based on fuels, topographic landmarks, landownership concerns and priorities, and statutory requirements. The major delineations for the FMAs were based on vegetation types and expected climax successional vegetation and their effects on wildfire behavior. This plan is currently being updated by Animas Foundation personnel with State Forestry and BLM assistance.

Since completion of the 1997 plan, numerous fires have occurred in the region. All managed under recommendations provided for in the plan. The Walnut Fire is an example of using local vegetation types and topography to maximize resource benefits of wildfires, minimize resources damage to surrounding landowners, and minimize cost and exposure to dangerous situations for firefighters.

On 27 June 2002, volunteer fire department engines were dispatched to a fire near Center (*fig. 1*). Initial reports placed the fire near Center Peak within the Gray Ranch and in the Rough Creek FMA.

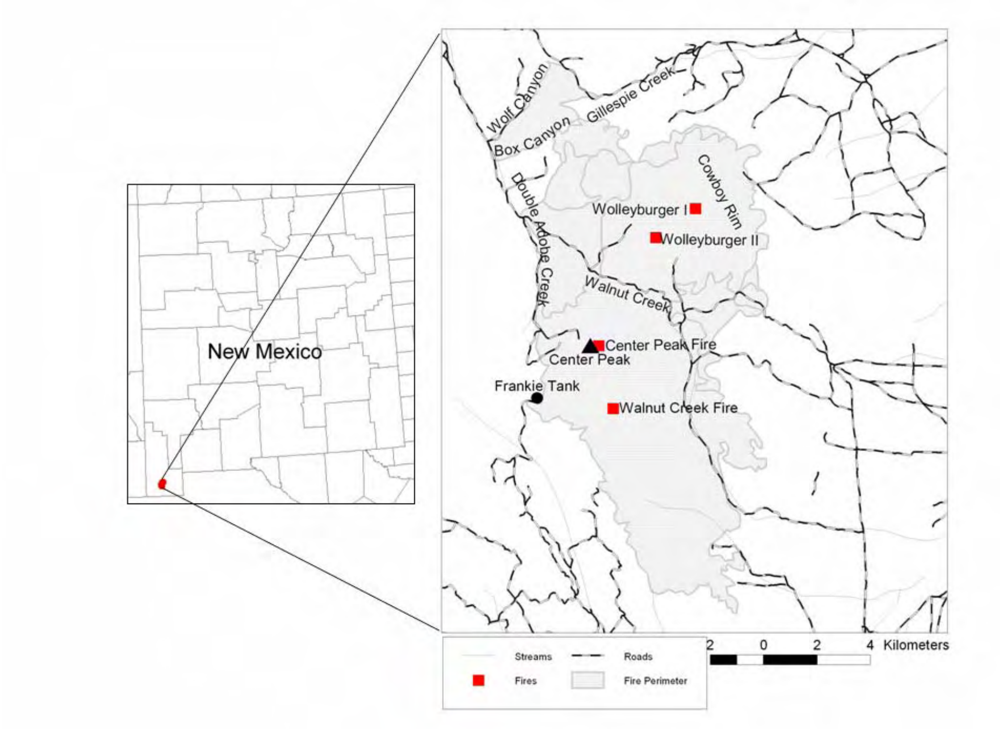

Figure 1—Map of fire extent (shown in grey) with four ignition points of each fire and prominent land marks for Walnut Fire Complex.

Using a land cover map, vegetation communities and fuel models can be used pre-incident to provide firefighters a management context of the fire and the natural and suppression resources available. Because of the varying land management occurring within the Bootheel region, the only available land cover map was from the original New Mexico Gap Analysis Project (Thompson and others 1996). This information coupled with coverages of FMAs and prior agreed upon appropriate suppression response, can provide management with information to facilitate logistical support. An updated more resolved map would further enhance this support.

Two additional ignitions (Wolleyburger 1 and Wolleyburger 2) were discovered on 28 June 2002 northeast of the Walnut fire. On 29 June the Walnut fire was less than 250 ac and still burning in the Center Peak area, while the Wolleyburger Fires were about 100 ac each. Personnel assigned to the fire adopted recommendations for management as outlined in the Bootheel Fire Management Plan (Boykin 1997). Post-fire analysis identified the potential for high intensity burns to the west of the existing fires in the Madrean Lower Montane Conifer Forest, Madrean Closed Conifer Woodland, Madrean Open Oak Woodland (Encinal) Rocky Mountain Montane Scrub & Interior Chaparral and Rocky Mountain Montane Deciduous Scrub based on the New Mexico Gap Analysis Project land cover map (Thompson and others 1996).

On 30 June, allowable areas were identified based on known and presumed fire breaks and behavior. Incorporating digital road coverages and a land cover maps in a GIS would provide additional information to further identify these allowable areas. Additional modeling of vegetation and terrain could identify areas for potential blow ups or hotspots. Further, inclusion of aerial photography and satellite imagery can provide information for logistical support. The fourth fire of the complex (Center fire) was discovered north of the Walnut Fire and was estimated to be about 50 acres. The Walnut fire's movement toward the southeast was stopped by a change in fuel type. Use of a land cover map and validation by fire personnel may have freed up resources for addition suppression assignments elsewhere.

Fuel type changes on the east flank of the Cowboy Rim area were expected to keep the fire from moving east, and light fuels between Gillespie Creek and the north base of the Cowboy Rim would limit fire growth north. This left the only area of escape from Frankie Tank to Double Adobe Creek. Burn out operations began along the road near Frankie Tank on July 1st. Use of land cover maps in conjunction with fire behavior models may have identified these critical areas long before the personnel actually verified the site from in the air and on the ground.

On 1 July, slurry was dropped to protect a pump jack. Movement east was not a concern as the fuels changed from grass/shrub to creosote/desert pavement. The movement toward this pump jack was unexpected and current land cover maps could have been helpful in identifying exactly how far down hill and to the east that the fire could move. Burn out operations along the road proceeded north. By this time the Wolleyburger fires had joined but were north of the Walnut Creek Road, the Center fire was south of the road, and the Walnut Fire had jumped Lower Walnut Creek and began making runs to the east. The south flank had black lined itself. This is what fire managers had expected, but more resolved land cover maps may have identified these fuels type changes potentially limiting fire growth.

On 2 July, burnout operations continued. All fires had joined during the night, and had moved close to the burnout along the road south of Frankie Tank. Fire management personnel identified Wolf Canyon as the northwest tie between Double Adobe Creek and Gillespie Creek. Wolf Canyon was chosen over Box Canyon due to easier burnouts, lighter fuels, better roads, and increased firefighter safety. Again resolved land cover maps could have identified these features. From July 3 to 4, burn out operations were completed and on 5 July the fire was turned over to the Gray Ranch. Total acres burned were 27,713 with a total cost of suppression at $192,010.

Synthesis

Through discussions and preparation of this manuscript the three parties involved recognized the contribution a land cover map can have to resource planning, fire and natural resource management. Realizing the importance of this land cover map and the need to make the map as accurate as possible, the BLM, State Forestry and New Mexico project of the SWReGAP are working together to try to create this data layer. BLM and State Forestry have provided and continue to provide training site data to assist in land cover map creation.

The Walnut fire complex was managed without the use of a land cover map. We believe it could have been managed more effectively if an accurate land cover map was available and could have been used in the fire management process. Fire managers must assess the validity and application of these types of maps to each individual incident paying particular attention to the scale and resolution of the dataset.

Potential ways for GIS and remotely sensed data to play a part in fire management include prior planning and modeling as well as coordination during the event. Fire planning and modeling can be conducted in a wide variety of ways, from the simplistic to more complex approaches, depending on constraints of time, budget, or infrastructure. The most simplistic method of using an existing land cover data set is a simple crosswalk from the land cover labels to the fuel models. This can be facilitated when working with the National Vegetation Classification System (NVCS) because of the fire ecology and effects sections provided within the document.

Land cover maps should not take the place of existing processes or procedures. These maps should be implemented with the understanding of the limitations and the scale at which those limitations occur. Further, land cover maps should be recognized as simple predictions of the occurrence of a vegetation type based on substantive ecological and geographical knowledge.

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Community Participation in Fire Management Planning: The Trinity County Fire Safe Council's Fire Plan[1]

Yvonne Everett[2]

Abstract

In 1999, Trinity County CA, initiated a participatory fire management planning effort. Since that time, the Trinity County Fire Safe Council has completed critical portions of a fire safe plan and has begun to implement projects defined in the plan. Completion of a GIS based, landscape scale fuels reduction element in the plan defined by volunteer fire fighters, agency staff and community members has been a highlight of the past 3 years' work. Current efforts are focused on implementing fuels reduction projects. This paper reports on the plan development process, project implementation, and the challenges of involving landowners, particularly absentee owners, in fuels reduction activities. A demonstration effect is noted, in which a few landowners participate initially and model creation of defensible space for neighbors, who join later. The critical role of volunteer fire departments as first responders on many fire starts and the lack of funding support they face are discussed.

Introduction

On September 2, 2001, while mop up was still underway on the Oregon Mountain Fire that had burned 14 homes on 1,680 ac in nearby Weaverville, the Hyampom Fire started just west of Hayfork, CA. The Hayfork Volunteer Fire Department and California Department of Forestry crews were quick to respond to the wildfire. They had some welcome assistance from a resident GIS expert, who brought his computer to the fire line and was quickly able to provide the crews with accurate maps of the area including terrain, roads, vegetation types and structures. The GIS database had been developed by community members working with local state and federal agency staff in a participatory fire management planning effort of the Trinity County Fire Safe Council. The Hyampom Fire was contained after coming perilously close to homes and burning 1,060 ac just outside of town. The maps provided may have helped to speed the initial attack and limit the acres burned.

In California, a growing number of communities are forming more or less formal fire safe councils. The councils share a common goal of improving fire safety, but vary in their size, structure, membership composition and activities which depend on local circumstances. Many focus on fuels reduction around private homes in the wildland urban interface, and most benefit from the sanction of the California State Fire Safe Council umbrella organization and resulting cooperation and support of state or federal agencies involved in fire management (California Fire Safe Council 2003). A few fire safe councils are forging ahead with collaborative fire management

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planning efforts. This paper focuses on the planning process and plan developed by one group and on successes and challenges the group is experiencing.

The Trinity County Fire Safe Council

Trinity County is rural, extending over 2,000,000 mountainous acres with fewer than 13,000 residents. Over 75 percent of the county is managed by the federal government, primarily in the Shasta-Trinity and Six Rivers National Forests (*fig.* 1). The vegetation is predominantly mixed conifer forest and oak woodland (Sawyer and Keeler-Wolf 1995) with fire as the dominant disturbance regime. Wildfires have burned over 100,000 ac in the county in several large fires over the last 25 yr.

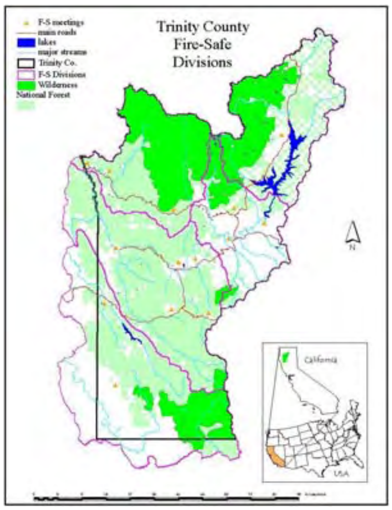

Figure 1—Location and land ownership map of Trinity County showing partitioning in five "Fire Safe Divisions" for Trinity County Fire Safe Council fire management efforts.

In mid 1998, the Trinity County Board of Supervisors' Natural Resources Advisory Council appointed a sub-committee to address the issue of wildfire management. This committee evolved into the Trinity County Fire Safe Council (the Council) that has met on average monthly since. The Council includes representatives from local volunteer fire departments (VFD), non-governmental organizations (NGOs), the county, state and federal land and fire management agencies, and other community members. All have signed a Memorandum of Understanding to cooperate on fire management planning (TCFSC 1999) and have agreed to a mission to reduce the risk of large, high severity fires in Trinity County by establishing priorities and implementing fuel reduction on a landscape scale for improved forest health, water quality, and community well-being (TCFSC 2003). In 1999, the Trinity County Resource Conservation District (TCRCD) and Watershed Research and Training Center (WRTC), both members of the Council, were successful in gaining grant support for fire management planning and implementation from the California Department of Water Resources and the USDA Forest Service Pacific Southwest Research Station[3]. The funding allowed TCRCD to provide staff support for the Council and initiate the fire safe plan development process. The outside recognition further legitimized and energized the Council's efforts.

[3] DWR funding was for $400,000 for 3 yr. USFS funds provided for additional GIS and planning staff support.

Planning by the Trinity County Fire Safe Council

Members of the Council meet monthly to discuss current issues, funding, progress on plan development, and implementation. A planning subcommittee of TCRCD staff and volunteers has taken the lead on most plan development activities. The plan is seen as a work in progress with some high priority elements fully developed and under implementation, while others are still being discussed. Various Council members have taken the lead on different plan elements in coordination with the Council staff. The draft document is distributed in an easily updated loose-leaf binder format. The Trinity County Fire Safe Plan has nine elements (*table 1*). Along with an element focused on public outreach (Element 5) the first two plan elements have received the most attention to date and will be discussed here.

Table 1—*Trinity County Fire Safe Plan Elements.*

1. Landscape scale management and fuel reduction plan
Participatory GIS based mapping of local knowledge pertinent to emergency response, values at risk in the landscape, identification and prioritization of fuels reduction treatments that could protect these values and identification of landscape level fuel management zones.
2. Supporting local fire suppression forces
VFD needs assessment and prioritization
3. Coordinating among all Actors
Identification of explicit mechanisms for enhanced communication and coordination.
4. Building local pre-fire treatment and fire suppression capacity
Identification of employment opportunities and mechanisms for local workforce capacity building in fire management
5. Promoting public education and involvement
Outreach to the community through schools, public events and media presentations on fire protection and management and TCFSC activities.
6. Funding fire management activities
Collaborative efforts to support member and joint council grant proposals, share expertise and equipment.
7. Identifying regulatory conflicts
Cooperation with regulatory authorities to test alternative approaches that can reduce or eliminate barriers to project implementation.
8. Cooperating with Trinity County planning department
Working with county planning department staff on the revision of the safety element of the County General Plan.
9. Monitoring plan implementation and effectiveness
Development of monitoring protocols for the review and maintenance of projects and development of organizational and community capacity through FSC activities.

Element 1: The Landscape Scale Fire Management and Fuel Reduction Plan.

Development of this element of the Fire Safe Plan depended upon existing GIS capacity and previous work that had been done by the not-for-profit TCRCD and the WRTC's Trinity Community GIS to compile and make compatible GIS data layers for Trinity County from a range of state and federal sources. Using these base layers, the Council planning team applied a participatory mapping approach to gather community input. The importance of knowledge of place and community based mapping (Aberley 1993) and the potential for enhancement of emergency response through local community mapping using GIS have been noted (McRae and Walker 2001). Participatory mapping, in which community members jointly define, what will be mapped, and how the data will be used, has been applied in a wide range of community development processes (Obermeyer 1998, Sieber 1997, Talen 2000) and has the benefits of engaging and empowering participating community members, making explicit information that might not otherwise be accessible.

Development of the landscape scale fire management and fuel reduction plan proceeded in three phases from fall 1999 to spring 2002. The first phase focused on gathering data pertinent for emergency response. The second phase identified and ranked values at risk from fire and made recommendations for protecting these values. The third phase involved defining preferences for placement of strategic fuel management zones in the landscape. In each case, meetings were held in local fire halls and community centers in five geographically dispersed sub-units of the county. Many meetings were held in the evening or on weekends and were publicized through locally effective mechanisms. At each meeting the Council's planning team introduced the fire management planning effort, explained the task for the day, and requested participants' assistance.

Phase I: Capturing Local Knowledge for Emergency Response

There were two goals for the initial round of 13 community meetings in winter 1999/2000. The first was to map information held by local residents that was pertinent to emergency response. The second was to make community members aware of the Council's efforts and identify individuals who might be interested in participating further. The Council's planning team arrived at each meeting with large sized paper maps of the local topography, surface hydrology, roads and structures. Participants gathered around the map depicting the area they were most familiar with and, working with a member of the planning team, used pens to correct or add information to the existing maps. Potential water sources on private land, locked gates, locations where a helicopter might land, where a bridge was too weak to carry a fire truck, where a road was washed out or too narrow to allow emergency vehicles to turn around, and similar information were noted (Everett and others 2000). Data accuracy was enhanced by local experts working together in groups.

The planning team took notes during the meeting, later entered the changes into the GIS, and sent print-outs of the new maps and notes back to all participants for review. Many new data points, information to that point not available to emergency responders, were mapped. Over 100 community members participated in these meetings. In each case at least one or two very knowledgeable people were identified who could be called upon to provide assistance to the Council in the future. Often the experts were people who drove the back roads as part of their jobs and encountered issues of concern on a daily basis. As a further result of the meetings, community members in far-flung corners of the county became aware of the Council and of county efforts to support VFDs and work on fire management collaboratively. A one day planning meeting was held with Council members, agency staff and interested community members to plan the next steps. At this meeting, five geographically distinct divisions of the county were recognized (*fig. 1*). Each unit was large enough to be a recognizable unit of the county while small enough to be considered familiar territory to local residents.

Phase II: Identifying and Prioritizing Values at Risk

In a second round of five one to two-day workshops in spring 2000, the Council planning team met with community members identified in the previous mapping effort, agency staff, VFD members and other interested residents to identify values at risk from fire. Again, the team brought maps of each county division and, in addition, a laptop computer and projection system for live, interactive mapping in which people could see modifications being made in the GIS based on their ideas. Working systematically across maps of their sub-unit of the county, participants identified, discussed and ultimately ranked the relative importance of subdivisions, resorts, prime campsites, pockets of old growth forest, communications towers and relay stations, road corridors, and other assets that could be at risk from wildfire. They also discussed a range of fuels reduction and potential fire suppression options and recommended specific treatments to protect each of the key assets identified. These ranged from encouraging private homeowner maintenance of defensible space to defining placement of proposed shaded fuel-breaks along roads. These meetings resulted in the identification of over 100 community defined and prioritized fuel treatment projects across the Trinity County landscape (Everett and others 2000).

Phase III: Landscape Level Fuel Management Zones

Fuel reduction projects surrounding specific assets at risk are one approach to saving key community assets from fire, once a fire reaches that location. However, they are not designed to manage the fire at a larger, landscape level. Council fire fighters and agency staff suggested that to better protect communities in case of wildfires, additional fuel reduction projects at strategic locations in the landscape might assist fire fighters in slowing or possibly stopping a fire. The National Fire Plan also calls for such fuel management zones (FMZs).

In a third round of participatory mapping meetings with community members and local agency staff, held in spring, 2002, locations for a number of strategic FMZs were recommended on key ridgelines or road corridors in each of the five county sub-units. The critical criteria for FMZ recommendation were that ridgelines 1) lie within 1.5 mi of communities; 2) are located between the community and the primary or secondary wind direction; 3) are accessible with pre-existing roads or jeep trails linked to pre-existing mid-slope roads; 4) lie in areas with a history of past fire occurrence; and 5) would have reasonable FMZ construction and maintenance costs.

To summarize, the landscape fire management and fuel reduction plan includes a three-part GIS database for emergency response, fuels reduction proposals to directly protect assets at risk, and recommendations for strategically-placed FMZs to alter fire behavior at the landscape scale. The management and fuel reduction plan for the area surrounding the community of Hayfork is shown below (*fig. 2*). The second element of the Trinity County Fire Management Plan calls for supporting local VFDs. It is important to note the pattern of fire starts in Trinity County (*fig. 3*). Of the over 9,000 fire starts recorded for Trinity County between 1911 and 2001, 4,939 fires were started by lightning. During late summer, many spot fires may be ignited on ridge tops in a single storm. From May or June through September each year, the US Forest Service retains crews of firefighters to control such fires. Of the rest of the recorded fires, 4,097 were caused by people and were concentrated in communities, along roads, and at campgrounds. As in many rural areas, the VFDs are first responders in emergencies including the majority of these fires (Cox 2002). In 2001, Trinity County VFDs responded to 1,700 medical emergencies, 225 fires and

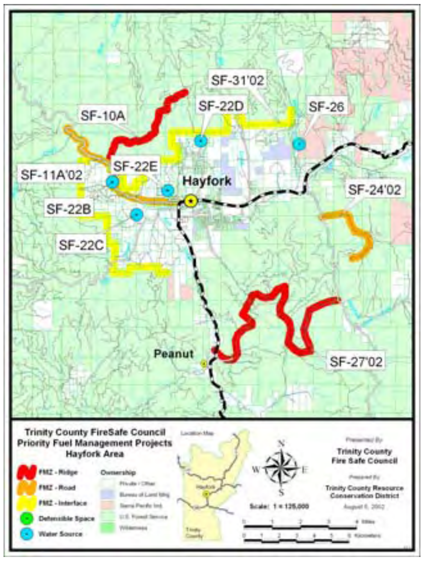

Figure 2–Hayfork Area Priority Fuel Management Projects include fuel reduction for defensible space inside specific subdivisions (assets). Strategically located fuel management zones on ridges and along roads surrounding the town slow fire at the landscape level.Plan Element 2: Support for Local Fire Suppression Forces.

600 general requests for public assistance (Cox 2002). The Council carried out a needs assessment among the Trinity County VFDs. It showed that in Trinity County, with 75 percent of the land in National Forest, only 4 of 16 VFDs receive consistent tax-based funding. Many of the VFDs support their activities with local fundraisers. Most of the fire engines are over 20 yr old, and several are 30 to 40 yr old and costly to repair. The regulation Nomex suits, boots and other equipment are expensive. A set of self-contained breathing apparatus (SCBA) and oxygen tanks, which is required for each fire fighter, costs around $2,500.

In this rural landscape, a fire started in a settlement of a few houses on private land or along a county road quickly spreads into the public wildland, and a wildland fire whether started at a primitive campsite or on a ridge top can quickly engulf both public and private assets in its wake. The failure to provide more support for VFDs in such landscapes is an important gap in fire suppression services. While more grants are becoming available through the National Fire Plan and state resources, it is often difficult for small VFDs to tap resources that require grant writing skills and ability to meet deadlines. The Council has become a strong advocate for the VFDs, and the Trinity County Fire Safe Plan calls for supporting the volunteers' fundraising efforts through communications about funding opportunities and grant writing.

Successes and Challenges of the Fire Planning Process

A critical measure of the success of the Council, its fire safe plan, and its collaborative development process is the degree to which proposals made in the plan are being accepted and implemented. There has been little or no controversy about the Council to date. None of the fuel reduction projects that have been carried forward for implementation have been challenged. Most of these have been on private land, but several have been highly visible shaded fuel breaks along county roads and on National Forest. It is possible that there will be increased scrutiny of efforts to implement the recommended FMZs on National Forest land. Of course any such activities will require full compliance with the National Environmental Policy Act (NEPA), the Endangered Species Act (ESA) and all pertinent regulations.

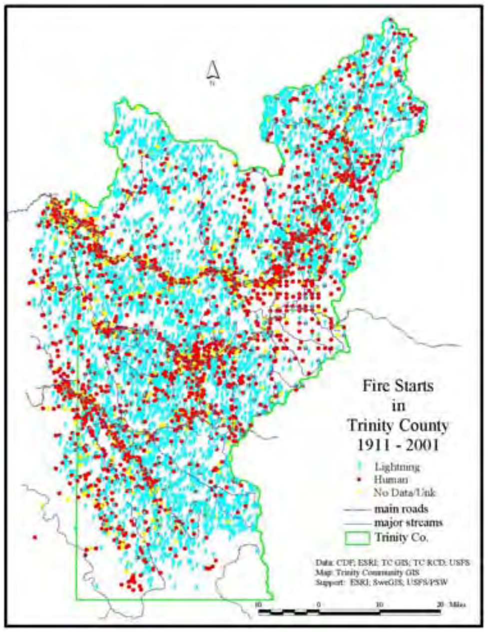

Figure 3—Fire start locations in Trinity County 1911-2001. Blue bolts indicate lightning fires, red dots indicate human caused fire. Note red dot concentration along roads and in settlements.

A number of projects were underway even as the finishing touches were being put to the final draft of the plan. Seventeen projects identified in the planning process were funded through Council-endorsed solicitation of grants from 2000 to 2002. The projects brought $783,256 into the county for fuels management, public education and outreach, and equipment, and projects were implemented by the TCRCD, Watershed Research and Training Center and several volunteer fire departments (TCRCD 2002).

Beside the fuels reduction projects undertaken on public lands, such as a shaded fuel break on land managed by the Bureau of Land Management along Hwy 299, over 1,000 private landowners were contacted and invited to participate in creating defensible space around structures or on boundaries of their land. Of the landowners contacted, 185 (16.5 percent) participated in fuel reduction activities by 2002 (TCRCD 2002). The comparatively low initial response rate is not surprising. Getting landowners involved in fuels reduction efforts is a major challenge for fire safe councils. An important demonstration effect operates, in which a few initial landowners take advantage of the support for fuels reduction work offered through the local fire safe council and provide an example that their neighbors may follow in later years. Staff working on the Council's projects reported progressive involvement of landowners who were initially cautious or even hostile (Baldwin 2002). In Trinity County, many absentee owners are difficult to reach and get involved.

While fewer than 300 ac were treated by 2002, it is important to remember that these acres were strategically placed in and close to communities and may help protect key assets or gateways to far larger areas. The task of finding funds to implement proposed projects is an ongoing challenge, particularly for VFDs. The Council has initiated efforts to support VFDs with joint proposal development and grant writing. One important project that was funded by the Trinity County Resource Advisory Council was to have VFD members canvass homes in their response areas to discuss defensible space measures on an individual basis with their neighbors. Council members believe that landowners are more likely to welcome visits from familiar VFD members than from representatives of state or federal agencies in this regard. Numerous other public education efforts have been undertaken in conjunction with Council activities (TCRCD 2002). Gaining the attention and involvement of absentee landowners, who may only be present for short periods during the year, is a persistent challenge. One approach the Council has taken is to approach absentee owners at their annual homeowners' events, (e.g., around the July 4th weekend, when a larger proportion of landowners are present).

Next Steps

As the key elements of the plan are completed, the process of plan implementation, monitoring and renewal will keep the Council busy. The emergency response data will need to be updated periodically and systematically, which could include reviews of the database every year or two with local experts and other interested participants. The data will need to be revised and redistributed to emergency responders and other users. Emergency responders such as VFDs need to be trained to use not only paper map books developed from the data sets, but also direct GIS based dispatching. More detailed emergency response databases that might include, for example, storage locations of hazardous materials for each response area could be developed.

Perceptions of values at risk in the landscape will change over time and need to be revisited. Fuels treatment projects that have been implemented will need to be monitored and maintained on a regular basis. Participating landowners will need to be reminded of maintenance needs and supported in implementation, even as newly participating landowners are encouraged to create defensible space. There will likely be wildfires that test the effectiveness of current fuel reduction efforts and FMZs and indicate ways to improve future efforts to protect community values at risk from wildfire.

Conclusion

It is clear that the Trinity Fire Safe Council has achieved some success with its fire management planning process. Certainly, it has captured some of the early state and federal funding made available for community fire planning and fuel reduction projects. A number of projects that could make a difference on the ground have been implemented. Some local residents were employed to do the work. The Council has also compiled data that could be critical in emergencies and that can be used in a range of management applications. Yet the true value of such efforts is difficult to quantify and may only become clear in the event of the big fire everyone would like to avoid. Even less tangible is the value of the participatory structure of the Trinity County Fire Safe Council and its community grounded efforts to tap into and share local knowledge, expertise, and energy, although clearly community capacity is being strengthened through the relationships developed among its members and with the community at large. Perhaps the clearest measure of the promise of this fire safe council and its vision lies in the members' continuing commitment to the effort that has brought them together once a month for 4 years.

Acknowledgments

The Trinity County Fire Safe Council (TCFSC) work was funded by the USDA Forest Service Pacific Southwest Research Station and California State Water Resources Control Board's Delta Tributary Watershed Program (Proposition 204). Activities were carried out by the TCFSC, Trinity County residents and experts on fire in California. In particular, Kenneth Baldwin, Noreen Doyas, Patrick Frost and Kelly Sheen of Trinity County Resource Conservation District and Phil Towle of Trinity Community GIS-Watershed Research and Training Center, Hayfork provided critical leadership for the planning process.

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Forest Service Fire Suppression Expenditures in the Southwest[1]

Krista M. Gebert[2] and Ervin G. Schuster[2]

Abstract

Fires in the Southwest (SW) are notorious and prominently featured in the national media, including the evening news—Storm King, Los Alamos, and the 2002 fires of Colorado and Arizona. The cost of suppressing these fires is staggering and continues to grow. Consequently, suppression costs for fires in the SW are one of the primary forces behind the National Fire Plan with its emphasis on fuel treatments and suppression cost reduction. In this paper, we inspect Forest Service (FS) fires in the SW and contrast them with fires in a larger geographical context—the U.S. and the rest of the West. We inspect characteristics of SW fires, such as their size and fuel types, and how these characteristics affect suppression costs. Cost contrasts focus on several key aspects of fire suppression, including predicting fire-specific suppression costs (making reference to regression analysis), differences in the make-up of suppression costs (e.g., personnel versus equipment), where firefighting resources come from (SW as a sink or source), and agency cooperation (other federal and state).

Introduction

The official title of this 2002 fire conference is “Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern U.S.” The title seems innocent enough. Managing fire and fuels sounds like a perfectly neutral, professional undertaking, but there probably is substantial disagreement as to why one would pursue such a venture. Some might argue that management is needed to restore ecosystems to some pre-European settlement condition. Others might argue that management is needed to attain some less historically based state of nature. Those perspectives seem well-grounded in physical and biological sciences. They reflect the perspective that natural systems are out of balance and need to be fixed.

This paper takes the perspective that fire and fuels are to be managed because the current state of nature is intolerably expensive—because natural systems are out of balance. In fiscal year (FY) 2002, the Forest Service (FS) spent about $1.3 billion in fire suppression. Expressed in constant 2001 dollars, annual expenditures for the past 30 yr averaged about $250 million. In FY 2000 and again in FY 2002, we broke the billion-dollar mark, four times the annual average. The hard-nosed might argue that the entire reason for undertaking fire prevention and detection, improving initial attack, expanding fuel treatments, and restoring ecosystems is simply to lower fire suppression expenses.

If that is the case, what are the determinants of suppression expenses for FS fires in the southwestern U.S.; how do Southwest (SW) fires fit into the bigger FS picture; how have FS expenses changed over time; and how do FS expenses fit into the bigger, national picture? We address these questions in reverse order.

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[1]An earlier version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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Federal Fire Suppression Expenditures

In the arena of Federal fire suppression on wildlands and open spaces, there is one agency from the U.S. Department of Agriculture (USDA) and four from the U.S. Department of the Interior (USDI). The FS is the sole USDA agency. USDI includes the Bureau of Land Management (BLM), Bureau of Indian Affairs (BIA), National Park Service (NPS), and Fish and Wildlife Service (FWS).

The best information on Federal fire suppression expenditures is the National Interagency Fire Center (NIFC) website (http://www.nifc.gov/). Among the five Federal agencies, the FS spends the most (*table 1*). Between 1994 and 2000, the FS incurred nearly three-fourths of the Federal expenses. Annual FS suppression costs ranged from about 60 to 80 percent of the total; BLM was second, incurring about 15 percent. Although agency expenditures vary annually, the order does not: FS and BLM are always first and second, with BIA, NPS, and FWS following in that order.

Table 1—*Fire suppression expenditures in dollars by Federal agencies, FYs 1994 to 2000.*

Year	Bureau of Land Management	Bureau of Indian Affairs	Fish and Wildlife Service	National Park Service	USDA Forest Service	Totals
1994	98,417,000	49,202,000	3,281,000	16,362,000	678,000,000	845,262,000
1995	56,600,000	36,219,000	1,675,000	21,256,000	224,300,000	340,050,000
1996	96,854,000	40,779,000	2,600	19,832,000	521,700,000	679,167,600
1997	62,470,000	30,916,000	2,000	6,844,000	155,768,000	256,000,000
1998	63,177,000	27,366,000	3,800,000	19,183,000	215,000,000	328,526,000
1999	85,724,000	42,183,000	4,500,000	30,061,000	361,000,000	523,468,000
2000	180,567,000	93,042,000	9,417,000	53,341,000	1,026,000,000	1,362,367,000

Federal agencies are not the only forces fighting fire. Most western states have firefighting organizations that coordinate local units. Although comprehensive data are not available on the role of FS suppression activity in this larger context, we developed some relevant data from our research. We identified total suppression expenditures for large fires fought by the FS in the west during 1996 to 1997. The FS incurred 87 percent of costs, State and county forces incurred similar suppression expenses (7 percent) to other Federal agencies (6 percent).

Forest Service Expenditures Over Time

FS fire suppression expenditures are increasing and becoming more erratic. Suppression expenditures can be expressed as actual values in the year they occurred and as inflation-adjusted. Time series data of actual suppression costs for FY 1970 to 2001 show increases in expenditures and variability (*fig. 1*). Analyses indicate a statistically significant ($P<0.01$) compound growth rate of expenditures averaging 7.7 percent per year. In FY 1988, the year of the Yellowstone National Park fires, expenditures averaged less than $90 million annually: afterwards they averaged $370 million. Moreover, expenditure variability also increased. The standard deviation of the mean cost increased 350 percent, from pre- to post-1988. Similarly, the standard deviation as a percentage of the mean, increased (from 63 to 70 percent).

Fire suppression expenditures can be expressed in constant dollars, reflecting an adjustment for inflation. *Figure 2* displays suppression expenditures expressed in FY 2001 dollars. When a time series of dollar values are expressed in terms of latter-year constant dollars, the constant dollar year acts as a fulcrum and early-year values are increased, thus flattening the time series. The compound growth rate, though still statistically significant (P=0.03), decreased to 3.1 percent per year. The breakpoint at FY 1988 is still present, but much less apparent. Prior to FY 1988, constant-dollar expenditures averaged $200 million annually. From FY 1988 on, constant-dollar expenditures averaged more than $400 million annually, a two-fold increase. However, based on constant dollars, relative variability increased. Although the standard deviation of the mean expenditure increased about 140 percent, from pre- to post-1988, the mean expenditure increased by only 105 percent. That means the standard deviation as a percentage of the mean increased by 17 percent, from 55 percent (pre-1988) to 64 percent (post-1988).

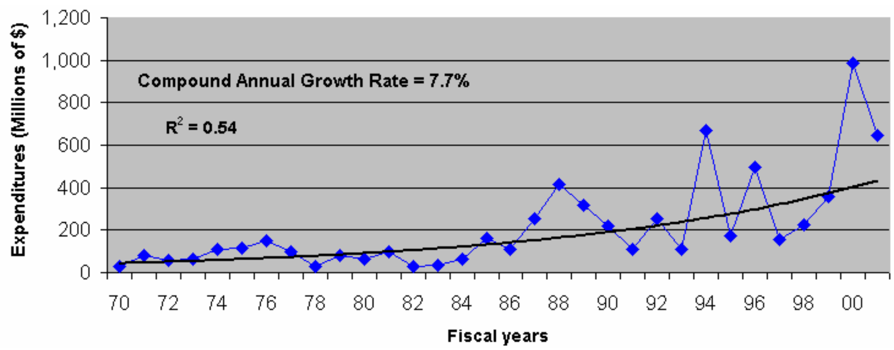

Figure 1—Forest Service fire suppression expenditures, FYs 1970 to 2001.

The role of key years in these analyses is critical. In particular FYs 1988, 1994, and 2000 are very influential. Continuing with constant-dollar analyses, but with the key years removed, the annual compound growth rate drops below 2 percent and is not statistically significant (P=0.21). However, those years did occur. As a result, we can attest to an annual, compound growth rate of more than 3 percent, along with increased volatility.

Forest Service Fires and the Southwest

With FS fire suppression costs dominating total suppression expenditures and with costs on the rise and becoming increasingly erratic, we turn to the role played by the SW. Data to address this question derive from the National Interagency Fire Management Integrated Database (NIFMID). The role played by SW fires depends on the focus, be it number of fires, acres burned, or suppression expenditures.

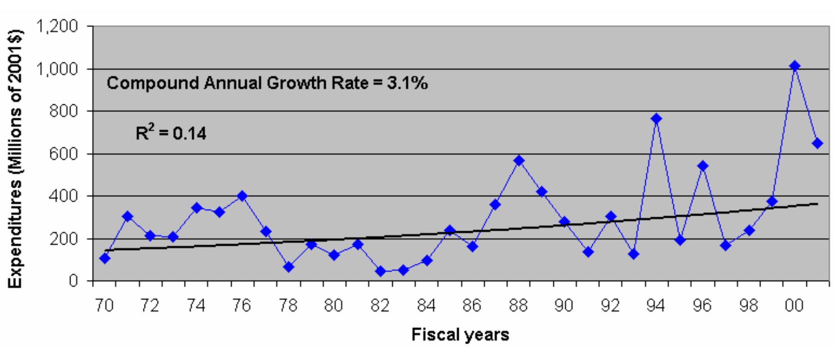

Figure 2—Forest Service fire suppression expenditures; 2001 dollars, FYs 1970 to 2001.

Over the past 3 decades, the FS has been involved with suppressing more than 370,000 fires, averaging more than 11,000 fires annually. The most fires occurred in 1970 and 1994 (*fig. 3*). Over those 3 decades, fires in the western US (FS Regions 1 to 6) dominate, accounting for about 80 percent of all FS fires. Within the west, the SW accounts for 60 percent of the fires: nearly half of fires nation-wide. The SW averages 5,500 fires annually, ranging from 3,400 (1983) to 7,300 (1972).

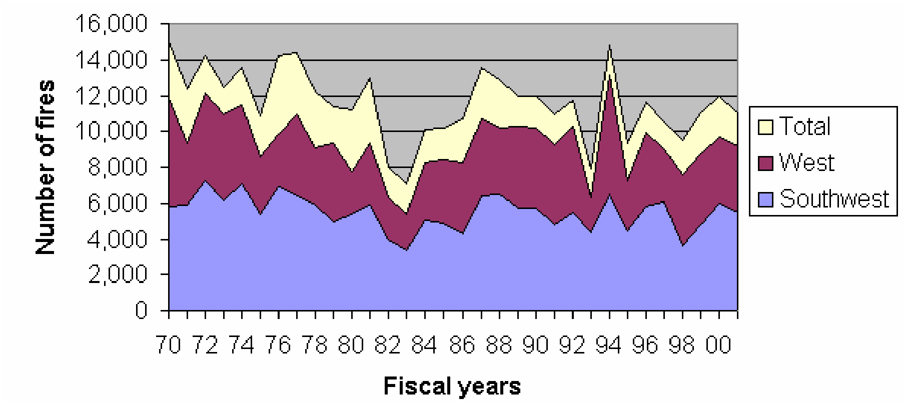

Figure 3—Number of Forest Service fires, FYs 1970 to 2001.

Nationally, fires with FS involvement, burned more than 21 million acres over the past 3 decades, averaging 660,000 ac annually. The most acres burned in 1988 (Yellowstone NP fire) and 2000 (Bitterroot NF fires), with 3.6 and 2.6 million ac respectively (*fig. 4*). Fires in the western U.S. accounted for 92 percent of all acres burned. Because the west dominates acres burned nationally, the temporal pattern in the west mirrors the overall national pattern. Such is not the case for the SW. Acres burned in the SW peaked in 1987 at 890,000 ac and dropped to 180,000 ac during 1988. Whereas the SW accounts for 60 percent of the western fires, it only accounts for 47 percent of the acres burned in the west. Over the past 30 yr, the SW accounted for 43 percent of acres burned nation-wide.

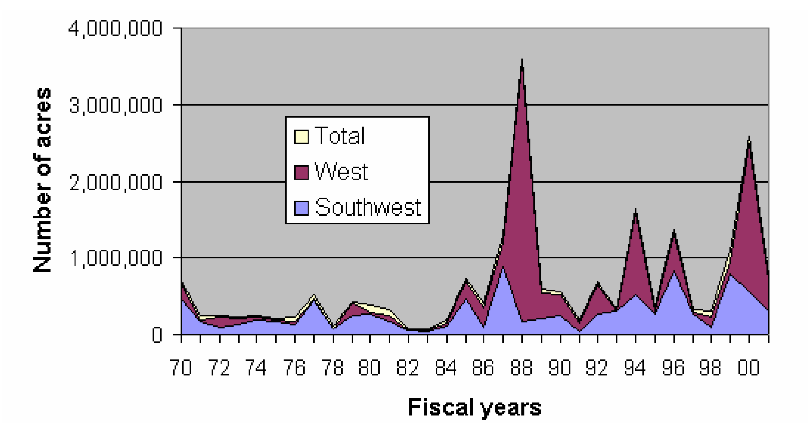

Figure 4—Forest Service acres burned, FYs 1970 to 2001.

NIFMID, the official FS records on fire-related activity, is not a perfect record. Information on estimated fire suppression costs lack credibility prior to 1986. However, cost estimates afterwards are adequate for comparisons. NIFMID-based information for FY 1986 to 2001 indicates FS annual suppression costs averaged \$356 million (*fig. 5*), whereas the financial system indicates \$388 million. Expenditures were highest in 2000 and lowest in 1986, both nationwide and for the west. The \$339 million average annual suppression costs in the west were 95 percent of the national total. Fire suppression costs in the SW roughly approximate the temporal pattern of the entire west. Costs averaged \$174 million, 53 percent of the west's total, with most (\$729 million) in 1987 and least (\$109 million) in 1998. With information on fire number, acres burned, and suppression costs, we can assess differences in suppression cost per fire and acre burned. Nationwide, fires cost more than \$31,000 (2001\$) to suppress. The SW averaged \$33,000 per fire, while the rest of the west averaged \$40,000 (differences are not statistically significant: P=0.49). However, the non-west average, \$8,000 per fire, is a statistically significant difference from western values (<0.01). We analyzed the annual trend in SW cost per fire (2001\$) and found no statistically significant trend (P=0.52).

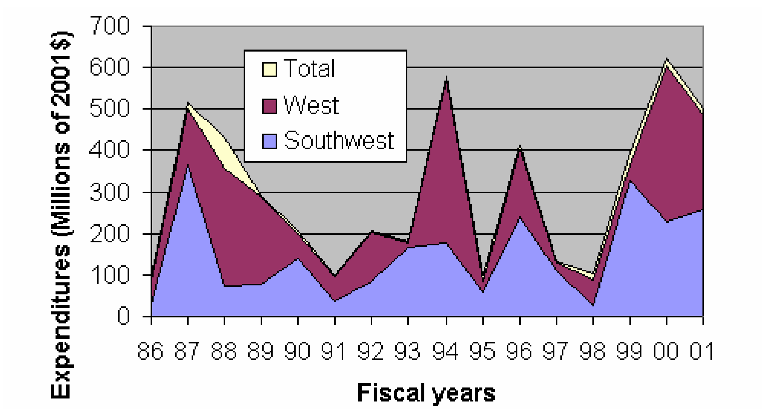

Figure 5—Forest Service fire suppression expenditures, FYs 1986 to 2001.

Nationwide, annual suppression cost per acre averaged \$430 ac^{-1} (2001\$), but there are marked geographical differences. The SW averaged \$510 ac^{-1} and the remainder of the west averaged about \$552 ac^{-1}, a nonsignificant difference (P=0.71). The non-west portion averaged \$266 ac^{-1}, a statistically significant difference from the western values (P=0.03). An analysis of annual cost per acre from 1985 to 2001 failed to reveal a significant trend (P=0.52).

SW Fires—Large and Small

To better understand the differences between small (classes A, B, and C fires of < 100 acres) and large fires (class D and better fires, >100 ac), we focus on the number of fires, acres burned, and suppression costs. Because needed data derive from NIFMID, fires must be restricted to FYs 1986-2001. Since FY 1986, the FS has participated in fire suppression on more than 82,000 fires in the SW. Most occurred in FY 1994 (6,300 fires) and the least occurred in FY 1998 (3,500 fires) (*fig. 6*). Averaging about 5,100 fires annually, most were small. ABC fires constituted almost 80 percent of fires, averaging over 5,000 annually. That means there were only about

100 class D+ fires annually, spread over the six SW states. The portion represented by large, D+ fires is barely discernable (*fig. 6*).

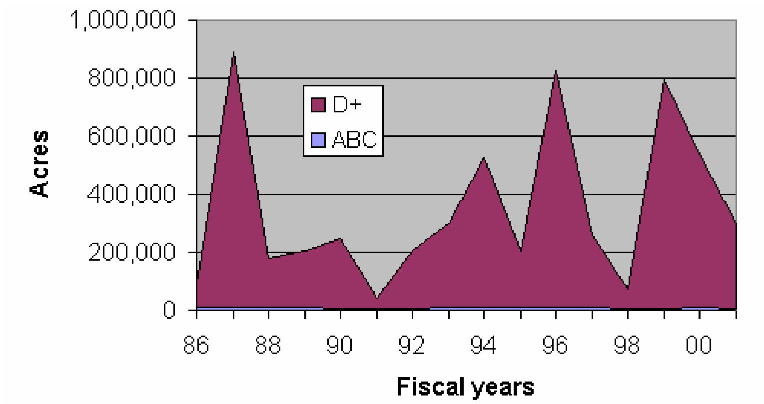

Figure 6—Number of Forest Service fires in the Southwest, FYs 1986-2001.

D+ fires dominate acres burned, and ABC fires dominate the number of fires. About 355,000 ac burn annually in the SW. The most acres burned in FY 1987 (almost 890,000 ac), followed by FYs 1996 and 1999 when about 800,000 ac burned (*fig. 7*). Between FY 1986 and 2001, D+ fires burned an average of 346,000 ac annually, and accounted for 98 percent of acres burned. In *figure 7*, the portion of acres burned represented by the smaller, ABC fires is barely discernable.

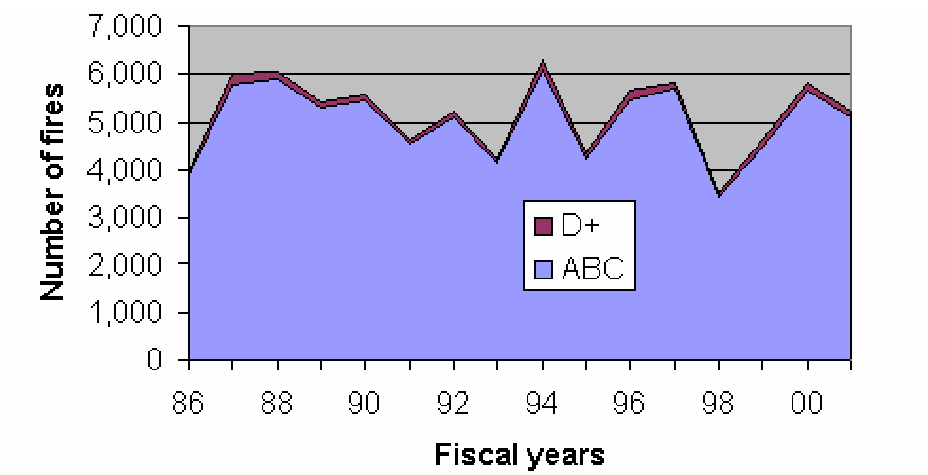

Figure 7—Forest Service acres burned in the Southwest, FYs 1986 to 2001.

Over the period FY 1986 to 2001, suppression expenditures in the SW averaged $175 million annually (2001$). FY 1987 was the most ($518 million) (*fig. 8*), and FY 1998 was lowest ($27 million), D+ fires accounted for the bulk of suppression expenditures at 84 percent while ABC fires comprised 16 percent.

SW Expenditures—Prediction

To better understand suppression expenditures for individual, large fires, we conducted a series of statistical analyses to identify and evaluate major causal factors. We used multiple linear regression, where the dependent variable was total suppression expenditures (2001$) and the independent variables were several dozen, fire-specific variables, such as acres burned and fuel type. Suppression expenditure information for each fire was obtained from FS accounting systems (Central Accounting Data Inquiry—CADI (for years prior to FY 2000) and Foundation Financial Information System—FFIS (for FY 2000 and beyond)). Fire-specific information was obtained from the NIFMID database and geospatial queries made of the 2000 Census of Population. We evaluated 485 D+ fires from the SW occurring from FY 1995 on, for which accounting and fire-specific information was available.

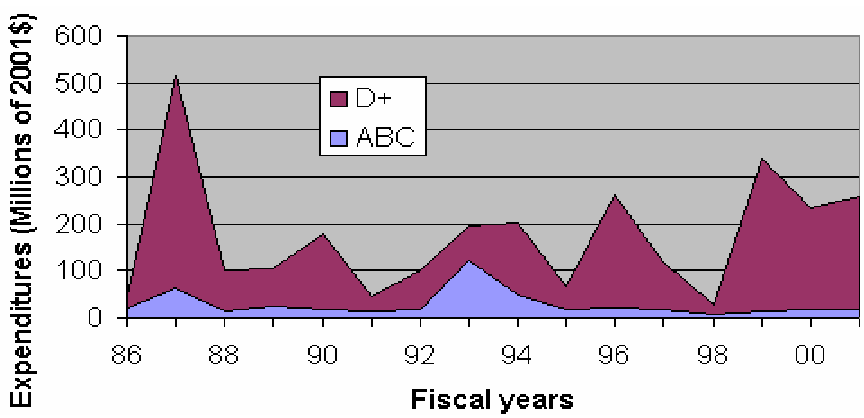

Figure 8—Forest Service fire suppression expenditures in the Southwest, 2001$, FYs 1986 to 2001.

We used simple correlation coefficients to eliminate variables. For inclusion in the regression, the correlation between suppression expenditures and the variable had to be significant at $P<0.10$. This eliminated two categories of variables. First, none of the initial suppression strategies (confine, contain, and control) were significantly correlated with suppression expenditures. Second, we used two types of measures to represent the wildland-urban interface: 1) distances to population centers of various sizes and 2) the amount of population within various distances of the fire. Population size was never significant while the distance to population was.

We built a multiple linear regression model from 16 candidate independent variables. Using stepwise and other model building techniques, we developed a model containing seven variables significant at the $P=0.15$ level (*table 2*). This model had an adjusted R2=0.44, and standardized coefficients (depicting relative importance). First, with a standardized coefficient of 0.527, Acres Burned is more that twice as important as Net Value Change in influencing suppression expenditures. Second, the significance of Net Value Change means that suppression costs increase with more valuable lands. Third, suppression expenditures on timber/slash fires are significantly greater than grass/shrub fires. Fourth, other things equal, suppression expenditures on California fires are significantly higher than other SW states. Finally, there is evidence that proximity to a small town increases suppression costs.

Table 2—*OLS estimates of fire suppression expenditure model for large (D+) Forest Service fires in the SW: dependent variable = total per-fire suppression expenditures.*

Variable	Std. coef.	P-Level
Acres burned	0.527	0.00
Net value change	0.182	0.00
Slope	0.148	0.00
Fuel type	0.140	0.00
California	0.111	0.01
Distance to place; 2,499 people	-0.063	0.11
Fire intensity level 6	0.055	0.15

SW Expenditures—Detailed

Questions arise as to what things are purchased through fire suppression expenditures. In accounting circles, “things” are “objects,” as outlined through “budget object codes.” As expenditures are entered into the accounting system, each type is recorded according to a hierarchical coding system. The Federal coding system has two problems. First, the system is not static over time, thus limiting our ability to accomplish detailed comparisons between years. Second, coding policies and procedures change over time, resulting in less detailed coding from year to year. Both problems frustrate detailed analyses, thus compelling more generalized evaluations. We accessed the Forest Service’s accounting system (Central Accounting Data Inquiry—CADI (prior to FY 2000) and Foundation Financial Information System—FFIS (FY 2000 and beyond)) records toward understanding SW suppression expenditures and how they differed from the rest of the west and nation in general.

The bulk of fire suppression expenditures are made for supplies and services (*fig. 9*). For large, D+ fires occurring from FY 1995 to 2001, supplies and services constituted nearly two-thirds of all suppression expenditures. This includes aviation, rent and utilities, printing, equipment, and various services. Personnel compensation accounts for about 30 percent of suppression expenditures. This category includes compensation and benefits of current employees and benefits to former personnel. The general pattern of expenditures does not differ in other geographical contexts. The percentage distribution of expenditures by category for the SW is almost identical for other western fires and, non-western fires nationwide.

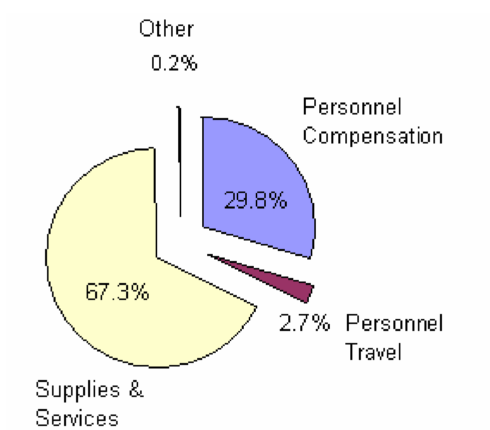

Figure 9—Forest Service fire suppression expenditures by type in the Southwest, FYs 1995 to 2001.

SW Expenditures—Financing

The final aspect of SW suppression expenditures deals with financing. Which FS fire-fighting organizations are responsible for suppressing large fires in the SW? To address this question, we used the NIFMID database to identify large SW fires and the FS accounting system database to identify cost patterns.

The SW area, consists of four FS regions: Pacific SW (PSW)-California, SW (SW) Arizona and New Mexico, Intermountain INT-Nevada and Utah, and Rocky Mountain (RM)-Colorado. *Table 3* shows SW suppression expenditures provided by FS regions and what each spent on large fires (e.g., firefighting expenditures from PSW accounted for 57 percent of SW fire suppression). Only about 20 percent came from the SW Region. The PSW expenditures accounted for 91 percent of its fire suppression expenditures; for the SW region, they accounted for 87 percent.

Table 3—*Percentage of SW suppression expenditures provided by Forest Service regions and of each region's suppression expenditures spent on large SW fires.*

Forest Service Region	Pct SW expenditures by Region	Pct Region expenditures in SW
Northern	1.41	6.65
Rocky Mountain	1.81	35.51
SWern	20.43	87.02
Intermountain	4.71	27.96
Pacific SW	56.57	90.96
Pacific Northwest	1.65	5.67
Southern	0.99	14.17
Eastern	0.65	29.81
Alaska	0.42	34.04
Washington Office	11.37	51.48

Conclusions

The FS is the dominant player in Federal suppression expenditures. Over time, FS suppression expenditures are increasing in both level and variability, including constant dollar measures. Suppression expenditures in the SW account for half of FS suppression expenditures. Large fires constitute 2 percent of the FS fires in the SW, but 84 percent of suppression expenditures. Acres Burned is the most important variable affecting fire-specific suppression expenditures, followed by Net Value Change, Slope, and Fuel Type. Two-thirds of SW suppression expenditures go to supplies and services, the remainder to personnel. About three-fourths of FS suppression costs in the SW come from the Southwestern and PSW regions.

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Spatial Analysis of Fuel Treatment Options for Chaparral on the Angeles National Forest[1]

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Abstract

Spatial fuel treatment schedules were developed for the chaparral vegetation type on the Angeles National Forest using the Multi-resource Analysis and Geographic Information System (MAGIS). Schedules varied by the priority given to various wildland urban interface areas and the general forest, as well as by the number of acres treated per decade. The effectiveness of these spatial treatment schedules was compared to 'No Action except fire suppression' using stochastic simulation performed by the model Simulating Vegetative Patterns and Processes at Landscape Scales (SIMPPLLE). Results are presented in terms of acres burned by wildland fire. The effectiveness of treatments in reducing acres of high-intensity wildland fire varied by the spatial distribution of fuel treatments.

Introduction

Chaparral shrublands of southern California comprise one of the most fire-hazardous landscapes in North America (Keeley 2002). Fires in this vegetation type tend to be stand-replacing crown fires with spectacular fire behavior (Philpot 1977). A large percentage of the acreage that burns occurs during severe weather when fire suppression efforts are least effective (Minnich 1983, Keeley and Fotheringham 2001b).

Complicating this situation is the fact that the wildland urban interface has been continually expanding in recent decades (Davis 1988, Keeley and others 1999, Keeley and Fotheringham 2001a) and the number of visitors on the forest has increased. As a result, there are more people on the landscape, which increases the chance of a human-caused fire. With more housing in the interface there is greater property loss from fires regardless of ignition source.

Fuel treatments have been suggested and applied to reduce the hazard from wildland fire (Philpot 1974, Minnich and others 1993). Alternative strategies have ranged from the use of prescribed fire to create a mosaic of age classes on the landscape (Philpot 1974, Minnich and Franco-Vizcaino 1999) to concentrating treatments in the wildland urban interface (Rice 1995, Conard and Weise 1998).

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Ability to conduct fuel treatments is limited by a number of factors, including budget, smoke, and number of days suitable for controlled burns. Forest managers need tools and methods for analyzing where to place treatments on a landscape to maximize their effectiveness.

This paper presents an analytical approach for spatially and temporally scheduling fuel treatments and comparing their effectiveness in reducing wildland fire and suppression costs in chaparral. It was conducted as part of the Joint Fire Science project "A Risk-Based Comparison of Potential Fuel Treatment Trade-off Models."

Study Area and Treatment Scheduling Criteria

The San Gabriel River District of the Angeles National Forest served as the study area. It is a 472,000 ac area with an extensive wildland urban interface bordering the suburbs to the northeast of Los Angeles, CA. Chaparral is the dominant vegetation type covering 63 percent of the area.

According to direction from the National Fire Plan, Angeles National Forest managers developed spatial priorities for fuel treatments based on housing densities in the wildland urban interface (*fig. 1*). The forest plan directs managers to conduct fuel treatments on 15,000 ac of chaparral per year. Current funding levels, however, are sufficient to treat only about half that amount annually. For hydrologic reasons, managers desire to treat no more that 40 percent of any fifth code watershed per decade. Finally, managers want to treat every 15 yr on priority areas and every 25 yr on the general forest outside priority areas.

Four fuel treatment scenarios were developed for this study (*table 1*). Scenario C represents the desired level of treatment (150,000 ac per decade), applies the spatial priorities and the watershed treatment limit of 40 percent, and follows the current policy of no active treatments in designated Wilderness. Scenario B reduces acres treated to 75,000 per decade (the current funding level) while maintaining the remaining criteria in C. Scenario A differs from B in that spatial priorities are not applied. This provides a test of the effectiveness of spatial priorities. Finally, Scenario D includes all treatable chaparral acres, while relaxing the watershed treatment limit.

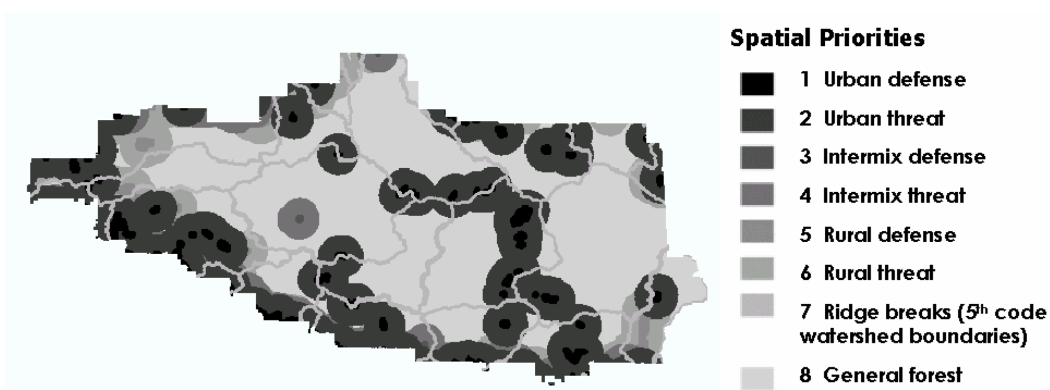

Figure 1—Spatial priorities for fuel treatments.

Modeling Approach

The approach presented here involves the sequential use of two models, SIMPPLLE (Chew 1997), a simulation model, and MAGIS (Zuuring and others 1995), a decision model for scheduling activities.

Table 1—*Fuel treatment scenario specifications.*

Item	Scenario A	Scenario B	Scenario C	Scenario D
Chaparral acres treated per decade	75,000	75,000	150,000	230,600
Apply spatial priorities	No	Yes	Yes	Yes
Apply treatments in wilderness	No	No	No	Yes
Treat <40 pct of each 5th code watershed per decade	Yes	Yes	Yes	No

SIMPPLLE

Simulating Vegetative Patterns and Processes at Landscape Scales (SIMPPLLE) (Chew 1995) is a stochastic simulation modeling system that predicts changes in vegetation over time and space by using a vegetative state/pathway approach. A vegetative state is defined by the dominant existing vegetation, size class/structure, and density. Change between vegetative states is a function of natural disturbance processes (insects, disease, and fire), and management treatments. The probability of a natural disturbance process occurring in a given plant community polygon is determined by both the state present in that polygon and the vegetative pattern represented by neighboring polygons. The disturbance probabilities provide the basis for stochastic simulation of location and timing of disturbance processes. Once a process occurs for a plant community polygon, logic is used to model its spread to neighboring polygons. Whether wildland fire spreads to a neighboring polygon is based on vegetation and type of fire on the polygon where fire is present, location of the neighboring polygon with regard to slope and direction of prevailing winds, and vegetation within the neighboring polygon.

For chaparral and the associated vegetation, we developed vegetative states and associated pathways for use in SIMPPLLE applying corporate information provided by Region 5 of the National Forest System from their Forestland and Resource Database (FRDB) and associated ArcInfo[6] coverage. Vegetation data was derived from crosswalks of the regional types, the CALVEG[7] hierarchical classification system, and the western forest types used by the Forest. A rule-based system was developed to translate vegetation information from the initial data-set into the vegetative state categories of habitat type group, species, class/structure, and density. Time steps were set at 10 years, and 5-acre grid polygons represented plant communities. We overlaid these 5-acre polygons onto the stand polygons, and assigned each 5 acre polygon the vegetative state of the predominant stand. Probability of fire starts was based on fire history for the area, and fire-spread logic

[6]Trade names are provided for information only and do not constitute endorsement by the U.S. Department of Agriculture.
[7]Developed by USDA Forest Service, Region 5 Remote Sensing Lab, 1920 20th Street, Sacramento, CA 95814.

was developed for two types of weather—spread under average conditions and extreme fire-behavior conditions.

We tested the version of SIMPPLLE developed for chaparral by making 20 stochastic simulations of a "No Action" scenario (fire suppression is the only management activity) and comparing the results with fire history for recent past decades. Average acres burned by wildland fire per year in the 20 stochastic simulations fell well within the range of average acres burned in the 1970s, 1980s, and 1990s (*fig. 2*). Distribution of fires by size classes up to 99 ac for the 20 simulations were also quite comparable to the same three past decades, although simulations predicted fewer fires in the larger fire size classes (*fig. 3*). This suggests that additional work may be needed in the fire spread and suppression logic in SIMPPLLE for the chaparral vegetation type for the fire-size distribution from simulations to more closely match past decades.

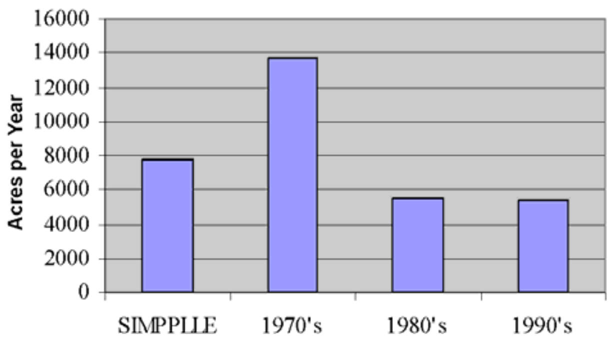

Figure 2—Average acres burned per year from 20 SIMPPLLE simulations for "No Action" compared to fires in three past decades.

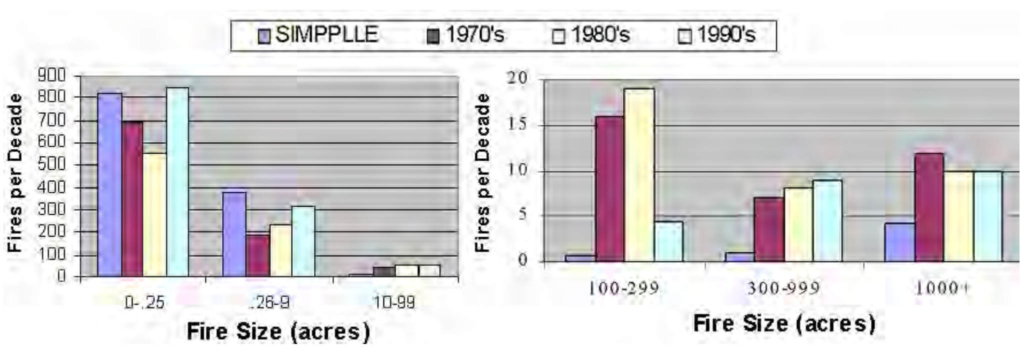

Figure 3—Distribution of fires by size class from 20 SIMPPLLE simulations for "No Action" compared to fires from past decades.

MAGIS

The Multi-resource Analysis and Geographic Information System (MAGIS) is an optimization model for spatially scheduling treatments that effectively meet resource and management objectives while satisfying user-imposed resource and operational constraints (Zuuring and others 1995). MAGIS accommodates a wide variety of types of land management treatments, together with associated costs, revenues, and effects, all of which can be used to control a treatment schedule.

MAGIS also contains a road -network component for analyzing road construction, re-construction, and closure.

The version of MAGIS developed for this study used decade time steps and the same vegetative states and pathways used in SIMPPLLE. We believed it impractical to schedule treatments for land units as small as 5 ac, so treatment unit polygons used in MAGIS were aggregations of the 5 ac vegetation polygons used in SIMPPLLE. Possible treatments included machine crush and burn; cut only; cut and burn; cut, stack, and burn; and burn only. Re-treatments were assumed to occur every decade in the priority areas (*fig 1.*), and every second decade in the general forest. We established 5th code watersheds and priority areas as zones within MAGIS and developed effects functions to compute critical information for these zones, such as acres treated in the watersheds.

Developing Spatial Fuel Treatment Scenarios

The process used to develop treatment scenarios is illustrated in *Figure 4*. Frequencies of fire on polygons from the 20 “No Action” simulations represent an estimate of relative fire danger on the landscape. We used them to develop a risk index that was input into MAGIS (Jones and Chew 1999). This fire-frequency risk index is combined with the scenario specifications (*table 1*) in MAGIS to control allocation of treatments.

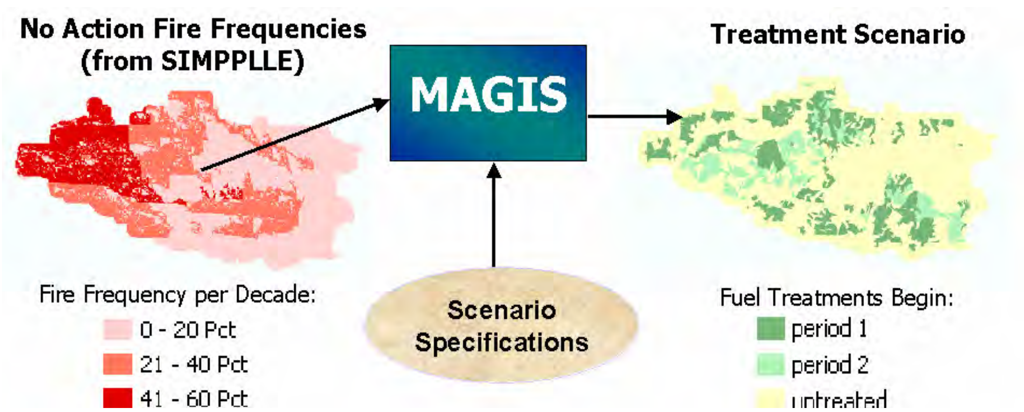

Figure 4—Process for developing spatial fuel treatment scenarios.

We made a series of MAGIS solutions in the process of developing each treatment scenario. Treatments were allocated by first minimizing the risk index for the highest priority level. In the next solution, risk for the highest priority was held at the minimum level while minimizing the risk index for the second priority level, and so on. In the last step we minimized the overall fuel treatment cost while holding the value of previously minimized risk indexes constant.

Treatment schedules developed for the scenarios are displayed in *figure 5*. No new areas are treated after decade 2 because treatment acres are achieved by re-treating areas scheduled initially for decades 1 or 2. We imported the four treatment scenarios into SIMPPLLE to model their effect on extent and frequency of wildland fire (*fig. 6*). Twenty simulations were run for each scenario, and fire frequencies were tabulated by polygon.

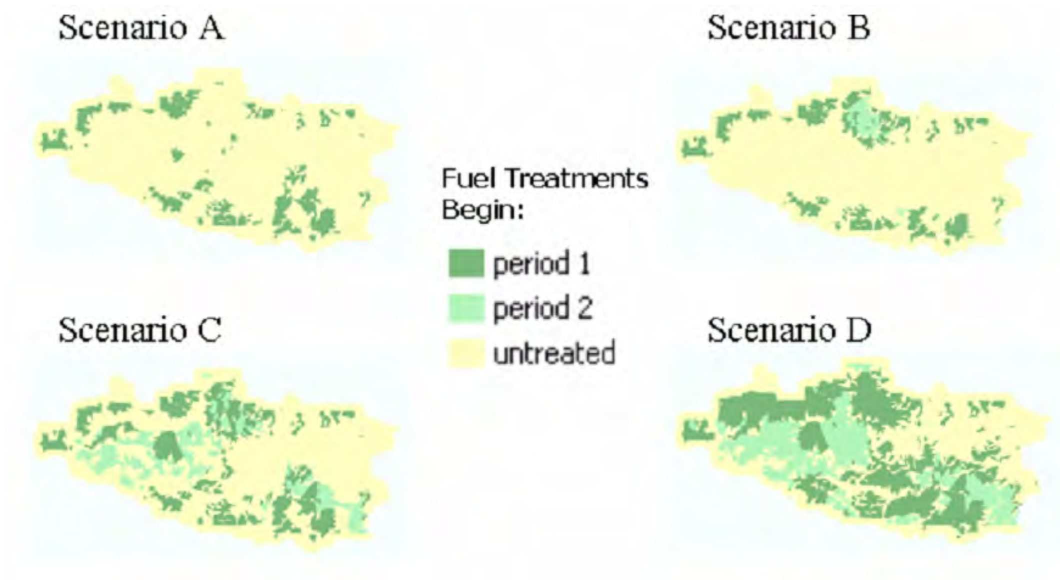

Figure 5—Spatial location of treatments in the four scenarios.

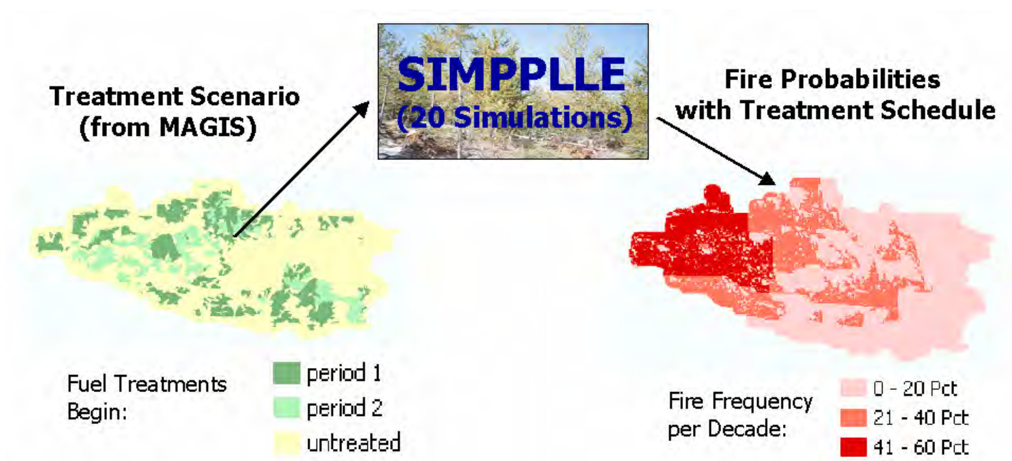

Figure 6—Process for developing spatial fuel treatment scenarios.

Results

Fuel treatments imposed by these scenarios made little difference in the average percent of the total landscape burned by wildfire per decade over 5 decades (*fig. 7*). In fact, the average percent of area burned for Scenario B is slightly higher than the "No Action" Scenario. This unexpected result was explained when we discovered there were some very large fires simulated for that Scenario, apparently by random chance in the stochastic process. This suggests more than 20 stochastic simulations are needed for this vegetation type to average out effects of low probability disturbance events. Measurable reductions, however, are observed in the average percent of Urban Defense Zone burned by wildland fire (*fig. 7*). Similar reductions in wildland fire due to fuel treatments were observed within other priority zones.

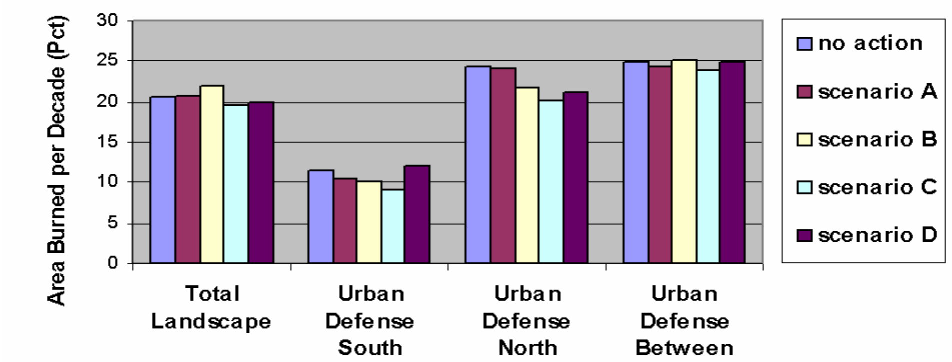

Figure 7—Percent of total landscape area burned per decade compared to the Urban Defense Zones.

Figure 8 depicts two specific locations that received priority treatments, and the effects of those treatments in Scenarios B and C. The oval area (*fig. 8*) received numerous fuel treatments in Scenario B. The fire frequency map developed from the stochastic simulations for Scenario B shows a corresponding lower wildland fire frequency in the oval area (lighter color) than does the "No Action" map. The average acres burned per decade in the oval area dropped from 6,998 for "No Action" to 3,533 in Scenario B. The rectangular area (*fig. 8*) provides a second example of location-specific effects of fuel treatments. This area received numerous fuel treatments in Scenario C, and fire frequency map shows a noticeably lower fire frequency (lighter color) than does the map for "No Action." Here average acres burned per decade dropped from 8,668 for "No Action" to 6,996 for Scenario C.

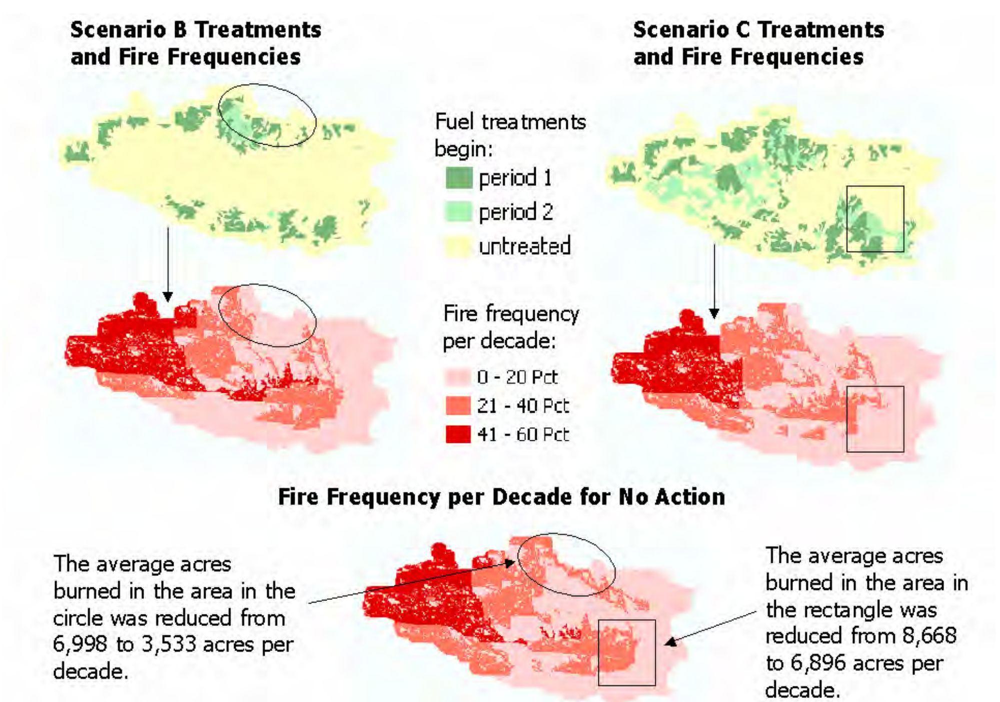

Figure 8—Fuel treatments and fire frequencies in two areas.

Discussion

Measurable reductions in wildland fire due to fuel treatments are observed in the Urban Defense Zone and other priority zones. However, this modeling effort shows essentially no reduction in the percent of total landscape burned as a result of fuel treatments. This may in part be a function of fire spread/suppression logic used in these simulations. Compared to three recent decades (*fig. 3*) the simulations for "No Action" predicted a higher percentage of the fires stopping in the .26 to 9 ac size class and not expanding into larger fires. This has the potential of masking the effect treatments have on the ability to suppress fires at smaller sizes. Additional work is needed in SIMPPLLE's fire spread/fire suppression logic to draw conclusions about the effect of fuel treatments on reducing wildland fire on this landscape.

This study did demonstrate the two models' valuable capability to conduct spatial fuel treatment analyses on a 400,000+ ac landscape. First, fuel treatments were scheduled spatially and temporally, despite a relatively complex set of spatial priorities for treatment, watershed limitations, treatment acreage limitations, and a cost minimization objective. Second, the modeling approach estimated the extent and location of wildland fire on the landscape, both with and without fuel treatments. This provides a good basis for evaluating the effectiveness of spatial fuel treatment strategies. Third, the stochastic simulations of treatment scenarios provide a good foundation for future work to quantify other important aspects in fuel treatment analyses. For example, location and severity of individual fires in the stochastic simulations, along with GIS information on the location and value of private structures, provide a basis for estimating private property loss due to wildland fire. This would offer the capability to predict reduction in private property loss associated with a specific spatial pattern of fuel treatments.

Another promising possibility is to use the treatment locations and the stochastic simulations to predict the combined resource effects of treatments and wildland fire. This offers the capability to compare overall resource effects (from treatments and wildland fire) across scenarios, including "No Action," which in some instances may display the greatest resource effects. This modeling approach has the potential to address many of the landscape fuel treatment questions posed at this fire conference.

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The Importance of Considering External Influences During Presuppression Wildfire Planning[1]

Marc R. Wiitala[2] and Andrew E. Wilson[2]

Abstract
Few administrative units involved in wildland fire protection are islands unto themselves when it comes to wildfire activity and suppression. If not directly affected by the wildfire workload of their neighbors, they are affected by the availability of nationally shared resources impacted by wildfire activity at the regional and national scale. These external influences should be taken into account when local administrative units plan their presuppression organizations. Failure to account for the external influences of fire workload and resource availability can lead to incorrect assessment of local resource needs. Recent advances in fire planning technology funded by the National Fire Plan have made it possible to better account for these external forces. Testing this new fire planning technology on the Umatilla National Forest in eastern Oregon and Washington, we found common planning assumptions about the influences of external fire activity can lead to significant errors in estimating suppression program performance.

Introduction

A major function of wildfire planning is to prepare for the next fire season by choosing the type, number, and location of initial attack fire suppression resources to best meet management objectives within budget constraints. In addition, fire planners must develop policies and rules for deploying local resources and requesting resources in response to fire activity. Planning a fire suppression organization would be easier if it were reasonable to ignore fire events outside an administrative unit's border. Most administrative units, however, find wildfire related events and activities beyond their border frequently to have a significant impact on local initial attack operations and performance.

An administrative unit's interaction with the external fire environment arises in several ways. Mutual response arrangements may exist with neighboring protection organizations. Or, reliance may exist on nationally shared aerial resources, like smokejumpers and air tankers. In either instance, an administrative unit may find itself in competition for shared suppression resources during periods of simultaneous fire activity at regional and national scales.

To the degree these external influences exist, a fire planner would attempt to account for them in the design of the local initial attack organization. However, judging the magnitude, nature, and frequency of these external influences in the absence of extensive empirical analysis is fraught with many difficulties (Cleaves 1994). The designs of current planning tools, like the Interagency Initial Attack

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Assessment model of the National Fire Management Analysis System (USDA Forest Service 1985) widely used on Federal lands or the California Fire Economics Simulator (Fried and Gilless 1999), do not explicitly take into account the ebb and flow of fire activity and related events outside administrative planning boundaries. Not adequately considering these external fire activities can lead to misjudged presuppression needs.

A few approaches exist to address problems arising from the impact of external fire activities. One is to increase the geographic scale of planning so that fire workload outside the boundaries is minimal relative to that within. LEAPORDS (McAlpine and Hirsch 1999), a forest fire planning decision support system for the province of Ontario, is an example of this approach. Recent efforts to do interagency planning also increases geographic scale and, thereby, similarly reduce the relative importance of mutual response arrangements during planning. Even at this scale, presuppression planning is challenged by the issue of how to effectively deal with competition for nationally and regionally shared initial attack resources.

This paper presents the Wildfire Initial Response Assessment System (WIRAS), a simulation model that features the ability to examine performance of a local fire suppression organization within the context of regional fire suppression activities that affect the local availability of nationally shared aerial resources. We use WIRAS to quantify and overcome some of the biases arising from assumptions commonly made about external fire by local fire planners using planning models not designed to explicitly address influences of external fire events and suppression activities.

Model

As a planning technology, WIRAS combines many of the best features of its predecessors while adding new features that give it the capability of addressing fire planning issues arising from external wildfire forces. As a clock-driven, next-event simulation model, like those of Fried and Gilless (1988), Martell and others (1994), and Wiitala (1998), it expends considerable effort representing the spatial and temporal interactions of fire occurrence and initial suppression response. Similar to Fried and Gilless (1999) and Martell and others (1984), WIRAS addresses how the frequency and magnitude of multiple fire events should influence the complexion of the presuppression organization.

WIRAS differs from its predecessors by featuring the ability to plan fire preparedness programs at local, national, and mixed scales. It builds on earlier methods used to study nationally shared initial attack resource programs, like smokejumpers and airtankers, with their large geographic footprint (Wiitala 1998). With funding from the National Fire Plan, WIRAS tackled the problem of adding capability for intensive examination of a local suppression resource program within the context of the regional or national fire environment (Wiitala and Wilson, in press). Against several full seasons of historical fires, WIRAS simulates the initial attack process for all spatially and temporally located fires at both the local and regional or national geographic scales. As a result, during periods of concurrent fire activity at the different scales, the simulation model can account for impacts on local program performance arising from it having to compete for use of aerial resources shared regionally and nationally. This capability permits WIRAS to directly address the issue of accounting for external fire events and activities in local presuppression fire planning.

Methods

On the 1.4 million acre Umatilla National Forest in eastern Oregon and Washington, we use WIRAS to simulate the performance of initial attack suppression for three scenarios with different planning assumptions regarding the level of interaction with the surrounding fire environment of national forests in Northern California, Oregon, Washington, Idaho, and Western Montana. At one extreme, the isolation scenario, we achieve minimal interaction with the external fire environment by assuming no requests for suppression assistance from aerial resources off the forest. To simulate this scenario, WIRAS makes smokejumpers, airtankers, and external helicopters unavailable to the forest. At the other extreme, the optimism scenario, we assume external aerial resources are always available when requested, an assumption fire planners often make. The competition scenario falls between the extremes by assuming only uncommitted external aerial resources are available to the forest. In this latter scenario, the forest must compete for use of nationally shared aerial resources.

We distinguish the scenarios in terms of local suppression performance and external suppression resource use. Contained and escaped fire counts and burned acres provide our measures of suppression performance. The number of aerial resource deliveries to fires on the Umatilla NF measures external interaction.

To examine our scenarios, we selected the 1986 and 1994 fire seasons. The 1986 fire season contained a below average 4,520 fires for the fire area surrounding the Umatilla NF, but an above average 175 fires on the Umatilla NF. As shown in *Figure 1*, a high degree of concurrent activity existed between the two areas, primarily during two consecutive days where the Umatilla experienced nearly half its fires for the season. We also note that most of the fire activity during the peak period of August 10 and 11 (Julian dates 222 and 223 in *figure 1*) occurred from eastern Oregon through central and northern Idaho. This close proximity to the Umatilla NF caused heavy competition for nationally shared resources such as airtankers and smokejumpers. The many fires igniting in wilderness and roadless areas with limited accessibility to ground resources further exacerbated competition for aerial resources.

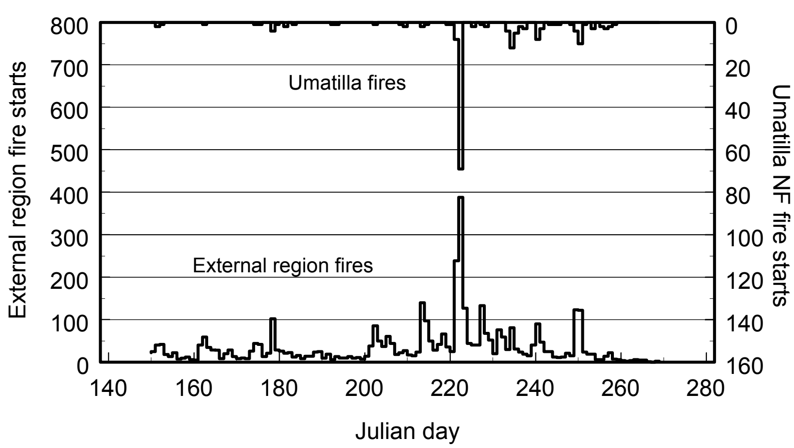

Figure 1—Daily fire starts in 1986 on the Umatilla National Forest and external region.

We selected the 1994 season to show that an atypically large number of fires at home (203) and beyond (6,943) do not necessarily pose a problem for a local fire suppression organization. *Figure 2* shows the Umatilla NF=s 1994 fire workload more evenly distributed over the fire season than in 1986. This was true for the external region as well. Compared to 1986, there was less dramatic coincidence in 1994 between the fire workloads of the two areas. We speculate that there would be less competition for aerial resources as a result.

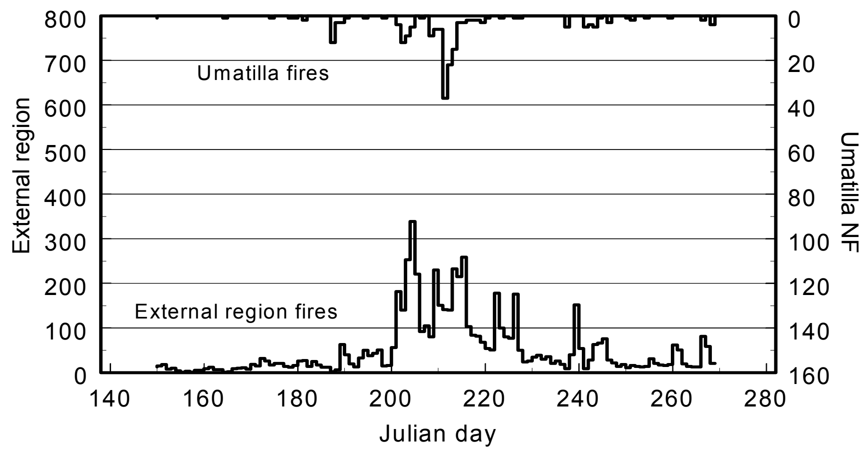

Figure 2—Daily fire starts in 1994 on the Umatilla National Forest and external region.

In our simulation runs, we use a local suppression organization that approximates what was in place on the Umatilla NF in 1994. Dispatch rules in WIRAS favor the use of ground resources over aerial resources when ground resource attack times are reasonable for the potential severity of a fire=s behavior and forest management objectives.

WIRAS currently meets most of its data needs with information in existing National Fire Management Analysis System (NFMAS) databases. In the case of the Umatilla NF, examples of these data include ground resource locations, attack times, duration of water for engines, and engine water retrieval times. Additionally, WIRAS relies heavily on a variety of fire related attributes available through NFMAS databases, like slope, aspect, elevation, intensity, spread, and fuel model. Suppression responses to fires are determined by the forests preplanned fire dispatches for management areas, fire danger levels, and single or multiple fire days (USDA Forest Service 2001).

Results and Discussion

The results of our simulations for the Umatilla NF=s two fire seasons show substantial contrast between years and planning scenarios. The 1986 fire season with its single, very heavy 2 day workload highlights the difficulties that arise in assessing suppression organizational needs when external influences are present. We start by looking at the competition scenario in which WIRAS competes to use shared aerial resources in the presence of significant concurrent external fire activity. The result we see in *table 1* is eleven escaped fires and 16,870 total burned acres. Umatilla NF actual fire history for 1986 shows twelve fires in excess of 100 ac and 13,805 total burned acres. Escaped burned acres are based on historical escaped fire sizes whereas contained burned acres are computed from elliptical fire size at containment.

Table 1—*Selected suppression performance statistics by scenario on the Umatilla NF for 1986 and 1994.*

		Numbers of fires		Burned acres		
Scenario	**Year**	**Contained**	**Escaped**	**Contained**	**Escaped**	**Total**
	1986					
Isolation		161	14	688	20,846	21,534
Optimism		171	4	439	5,596	6,395
Competition		164	11	491	16,379	16,870
Historical		165	10	651	13,154	13,805
	1994					
Isolation		200	3	527	4,467	4,994
Optimism		202	1	286	1,489	1,775
Competition		200	3	288	4,467	4,755
Historical		200	3	294	5,468	5,762

By contrast, the optimism scenario, with its assumption of external aerial resource always available upon request, results in a remarkable performance of just 4 escaped fires and only 6,395 burned acres. To achieve this outcome, use of smokejumpers and airtankers approximately doubles (*table 2*). The reduction in helitack use we suspect results from the greater availability and use of smokejumpers with quicker attack times. We point out that the optimism scenarios view of external resource availability, to the extent adopted, would lead to a false sense of security that could result in an underestimation of true local resource needs.

Although we mentioned earlier that a fire protection organization is seldom an island unto itself, the isolation scenario takes this position to the extreme by choosing not to use external aerial resources. *Table 1* shows, without any compensating increase in the ground resource organization, both escaped fires and burned acres increase about 28 percent over the results of the competition scenario. As expected, ground resources use was much higher here than in the other scenarios for both years (*table 2*). A look at the 1994 suppression performance results confirms our earlier speculation about the implications of magnitude and distribution of fire workload. Because of the more evenly distributed 1994 fire workload both on and off the forest, using just local ground resources in combination with the local helicopter (isolation scenario) gives a performance similar to the competition scenario which used available shared aerial resources. Both scenarios generate three escaped fire sizes (*table 1*). However, the lower number of contained burned acres in the competition scenario suggests use of external aerial resources leads to smaller fires.

Table 2—*Resource use by scenario on the Umatilla NF for 1986 and 1994.*

		Resource deliveries			
Scenario	**Year**	**Ground**	**Jumper**	**Helitack**	**Air tanker**
	1986				
Isolation		290	0	43	0
Optimism		253	71	37	85
Competition		250	29	45	47
	1994				
Isolation		340	0	39	0
Optimism		272	92	76	16
Competition		269	43	63	15

Looking at *table 1*, the optimism scenario again shows in 1994 that, if one assumes aerial resources were always available upon request, the result would look good on paper. However, the result would unlikely appear in practice. A typical planning response to the illusory and inflated performance of aerial resources would be to incorrectly trade off a part of the ground resource organization.

Conclusions

In this paper, we have highlighted some of the problems that can arise in the local initial attack fire planning process in dealing with the influences of external fire activity and resource availability. Current local fire planning techniques that do not model competing fire suppression activities beyond an administrative unit's boundary make assumptions about the availability of external suppression resources. In the WIRAS simulations presented here, we have shown that these assumptions, when taken to the extreme, can lead to flawed assessments of local resource needs. As we demonstrated in looking at the quite different fire seasons on the Umatilla NF, the severity of this problem for local presuppression fire planning is highly dependent on the dynamics of both the internal and external fire environment.

Statistically estimating the magnitude and nature of these external influences for meaningful use by existing planning models poses significant challenges, as well as research opportunities. As an alternative, local fire planning models could be modified to directly incorporate the influences of external fire activity and resource availability on an administrative unit's fire suppression performance, similar to the approach of WIRAS. This allows fire planners to plan their local organizations not only with respect to the dynamics of their internal fire environments, but also with respect to external fire activity which affects the availability of regionally and nationally shared initial attack resources.

Acknowledgments

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Reconstructing Fire History of Lodgepole Pine on Chagoopa Plateau, Sequoia National Park, California[1]

Anthony C. Caprio[2]

Abstract
Information on fire's role in pre-twentieth-century lodgepole pine forests of the southern Sierra Nevada is limited. It has generally been assumed that fire plays only a minor role in lodgepole's dynamics unlike in other portions of its range. This assertion was examined by sampling fire-scarred trees and reconstructing fire history in monospecific stands of lodgepole pine (*Pinus contorta* var. *murrayana* [Grev. & Balf.] Engelm.) on Chagoopa Plateau in the Kern River drainage of Sequoia National Park. Using dendrochronological methods 17 fire events were dated between A.D. 1385 and 2000. Prior to 1860 and Euro-American settlement, fire event dates showed mixed degrees of synchronization among sites with a number of widespread fires of the plateau. Mean fire return interval among sites was 45.4 yr, ranging from 31 to 74 yr by site. The frequency of past fire occurrence on the plateau indicates fire had a strong influence on this ecosystem, which continues through the present. These findings differ significantly from the generally held notion that fire does not play an important role in lodgepole ecosystems in the Sierra Nevada. Also of interest was a cluster of 1880s fire dates at sites near Sky Parlor Meadow suggesting burning around meadows by Euro-American shepherds.

Introduction

Prior to Euro-American settlement, fire occurred frequently in the Sierra Nevada and influenced the dynamics of most Sierra Nevada ecosystems (Kilgore and Taylor 1979, Skinner and Chang 1996). Fire histories derived from tree rings have shown variable fire-return intervals prior to Euro-American settlement (Kilgore and Taylor 1979, Swetnam 1993, Caprio and Swetnam 1995, Stephens 2001) but nearly all these results are from lower- and mid-elevation conifer forests. Change in this important ecological process, beginning with Euro-American settlement, has resulted in widespread impacts to Sierran ecosystems (Kilgore 1973, Parsons and DeBenedetti 1979).

Within the range of lodgepole pine, fire has been recognized as an important ecological process in the "interior" Rocky Mountain variety (*P.c.* var. *latifolia* Engelm. ex S. Wats.), with stand replacing fires occurring at 100-to-400 year intervals (Brown 1975, Arno 1980). This contrasts to the Sierra Nevada where fire has generally not been recognized as having had a significant ecological influence on monospecific stands of this species (Parker 1986, 1988; Rundel and others 1988, Rourke 1988, Lotan and Critchfield 1990, Skinner and Chang 1996). However, lodgepole in the Sierra Nevada has not been well studied and no fire history sampling has been conducted to examine past fire occurrence (Skinner and Chang 1996).

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[1]An earlier version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2-5, 2002, San Diego, California.

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The goal of this study was to document past fire occurrence patterns in monospecific lodgepole pine forests in the upper Kern River drainage of the Sierra Nevada. This information will improve our understanding of current and past vegetation composition and processes in this community. This paper will primarily review fire frequency patterns. Spatial patterns and climate relationships of fire occurrence were also examined but will not be addressed here due to space limitations.

Study Area

Chagoopa Plateau (4,379 ha) is located in the upper Kern River drainage in the southern Sierra Nevada (*fig. 1*). The plateau is bounded on the east by the Kern River Trench and to the west and south by the Big Arroyo drainage, both deeply incised by glacial activity. Vegetation on the plateau is dominated by lodgepole pine which grades into foxtail pine (*P. balfouriana* ssp. *balfouriana* Grev. & Balf.) on higher elevation sites and xeric conifer at lower elevations. Several meadow complexes also occur with Sky Parlor being the largest.

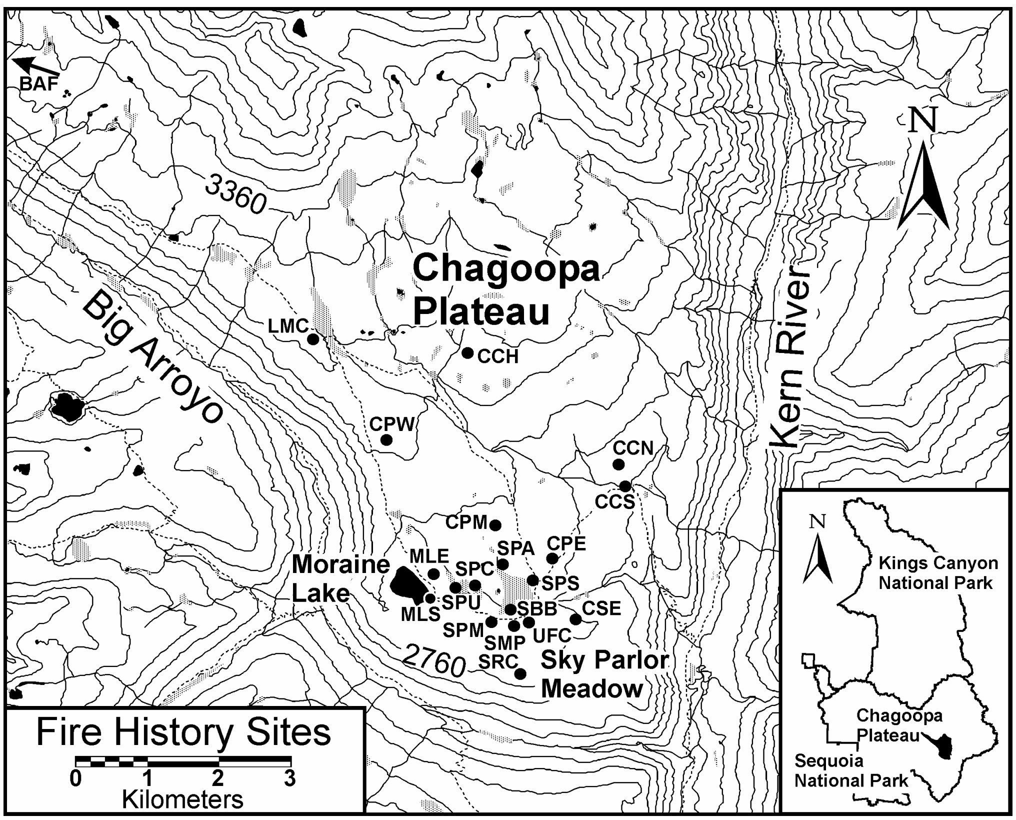

Figure 1—Map of Chagoopa Plateau (120 m contour intervals) showing location of fire history collection sites (black circles) and relative location of plateau within Sequoia National Park (inset). Lakes are shown as irregular black polygons, meadows as shaded areas, and trails as dotted lines. Site BAF is off the map to the west.

Climate of the area is Mediterranean with moist winters and dry summers. Total annual precipitation averaged 580 mm at the Crabtree Meadow weather station. Precipitation in the upper Kern is strongly influenced by the high ridges and peaks of the Great Western Divide that create a strong rain-shadow effect on storm tracks moving west-to-east. Sporadic thunderstorms can produce a limited amount of summer moisture. Temperatures on the plateau are cool during the winter (January mean -6.7ºC) and moderately warm in the summer (mean 9.8ºC during August).

Methods

Samples were obtained from 20 sites on the plateau that were predominantly monospecific stands of lodgepole pine. Area encompassed by each site ranged from about 0.25 to 1 ha. Sites were selected to reduce possible effects of the size of area sampled or within site variations on fire frequency estimates (Arno and Peterson 1983). Sites were homogeneous with similar vegetation, topography, soils, and aspect. No obvious fire spread barriers were present within sites.

Samples were collected from multiple trees at each site since it is uncommon to find all fire events recorded by a single tree (Agee 1993). Potential samples were examined in the field for evidence of fire other than the catface scar to assist in determining whether scars were fire caused. Number of samples obtained at a site was determined by the amount and quality of available material. Samples were collected as either cores using methods described by Sheppard and others (1988) or as partial sections (Arno and Sneck 1977).

Samples were surfaced and crossdated using standard dendrochronological techniques (Stokes and Smiley 1968) allowing fire scars to be assigned to the calendar year of occurrence. Mean fire return intervals (MFRI) were determined for each site (a site was considered a replicate and individual trees sampled as subsamples) based on the number of times the site (considered a point on the landscape) had burned over a specific time period.

Results and Discussion

Nineteen sites were sampled (18 on the plateau and one in Big Arroyo) with samples collected from 93 trees. An additional set of samples from the Funston Cabin (cores from seven logs) also provided evidence of past fire from growth suppressions apparent in the tree-ring series. Sample size varied from three to fourteen fire-scarred trees per site (with one exception at site MLS where only one tree was sampled). Cross dating was successful on 81 (87 percent) of the scar samples.

Fire scar dates spanned the period from 1455 to 1996 (tree-ring series from samples spanned the period from A.D. 1385 to 2000). However, prior to 1751 only a single recorder tree was sampled and fire history and frequency estimates become less reliable. A total of 186 fire indicators (scars or other fire related markers such as growth suppressions) were found on the 88 trees crossdated and these indicators recorded 17 fire event dates. Six of these event dates were widespread and found at multiple sites while 11 were found at single sites. Pre-Euro-American settlement MFRI by site ranged from 31 to 97.8 yr (average for plateau was 50.1 yr) for the full period of record prior to 1860 and 31 to 74 yr (average for the plateau was 45.4 yr) for the period from 1744 to 1860 (*table 1*). For the latter period, during which there were many replicate samples, individual fire return intervals ranged from 9 yr (one site recorded both the 1806 and 1815 fire events) to 74 yr.

Table 1—*Summary of fire history information by site prior to 1860 with mean fire return intervals (MFRI), number of intervals, and range in years of individual intervals between fires.*

	Mean fire return interval		
Site	Interval	(no.)	Range (yr)
Meadow sites (6)	47.5	(2)	31–64
SMP	31.7	(3)	9–55
SPU	49	(2)	34–64
MLS	-	(0)	-
MLE	64	(1)	-
CPE	40	(1)	-
CPM	31	(1)	-
CSE	97.8	(4)	40–155
SRC	48.0	(2)	41–55
CCS	74	(1)	-
CCN	-	(0)	-
CPW	31	(1)	-
CCH	40	(1)	-
LMC	-	(0)	-
BAF	-	(0)	-

Widespread fires burned across large portions of the plateau in 1751, 1806, 1815, 1846, and 1996. These five fire event dates, found at multiple sites, indicate fire spread over variable and moderate-to-large areas of the plateau. This suggests fuel loads were sufficient to carry a fire and/or that extreme burning conditions existed. The eight sites recording the 1751 fire were confined to the southwest portion of plateau. Early in the nineteenth century, two mutually exclusive fires occurred within nine years of one another. Sites recording the 1806 fire (six sites) were located on the eastern portion of the plateau and those recording the 1815 date (eight sites) on the western portion. The 1846 fire event was recorded at 14 sites located across the eastern and northern portion of plateau. The fire in 1996, recorded at 13 of 15 sites within the burn perimeter, was lightning caused and burned 1,115 ha on the plateau.

Six single-site event dates were recorded prior to 1860. Dates prior to 1751 (1455±, 1610, and 1744) were from a single recorder tree at CSE. Prior to the 1880s other single year dates were found in 1772, 1847, and 1849. These events may have been: 1) small fires, 2) fires that burned adjacent to the plateau and only burned a small area on the plateau, 3) larger fires where the record was lost or 4) large fires that were missed by the current site localities. Post-1860 fires at single sites were recorded in 1867 and 1989.

Also of interest was a cluster of fires during the 1880s (1882, 1885, 1887, 1889) at sites around Sky Parlor Meadow. All 10 sites recording these events were within 500 m of Sky Parlor or an associated meadow. The number of fires recorded during this 7 yr period, coupled with reports of the utilization by shepherds, strongly suggests the fires were human-caused and probably related to grazing. Many reports from the period allude to shepherds burning areas in the fall as they left the mountains to improve grazing conditions (Barrett 1935 in Vankat 1977).

It is currently unknown whether fire return interval patterns found in the lodgepole community on Chagoopa Plateau are representative of lodgepole throughout the southern Sierra Nevada. The FRI values for lodgepole forest on

Chagoopa Plateau are intermediate between those reported for closely associated red fir forest (MFRI 15 to 30 yr) and xeric conifer forest (MFRI 30 to 60 yr), and generally higher elevation subalpine conifer forest (MFRI 187 to 374 yr) found in the parks (Caprio and Graber 2000).

The frequency of pre-Euro-American fires on Chagoopa Plateau was unexpected. Assumptions about fire regimes in this community have usually suggested that fire was rare or that fires were small and not important (Lotan 1975, 1976; Rundel and others 1988) or that frequencies were long–in the order of every 100 to 200 years (Vankat 1977, Caprio and Graber 2000, van Wagtendonk and others 2002). In Yosemite National Park, Parker (1986, 1988) found little evidence of fire in lodgepole pine stands that were studied. This contrasts markedly with the results from the present fire history analysis. The importance of fire in lodgepole in the southern Sierra was also suggested by Keifer (1991), who examined age structure of lodgepole near Rock Creek, immediately east of Chagoopa Plateau. This study found pulses of regeneration closely linked to specific fire dates. Overall, these regional differences suggest there may be geographical variation in the importance of fire in Sierran lodgepole pine communities and that these may vary over fairly short distances.

A question that cannot be answered by the current results is what past patterns of fire severity occurred across the landscape. The many trees with multiple fire scars indicate that fires were often not stand replacing. Observations of the sampled stands indicate a mixed severity regime. The stand age patterns found by Keifer (1991) in Rock Creek also indicated a mixed fire severity pattern with pulsed recruitment and even-aged patches. Severity patterns might also be inferred from contemporary fires such as the 1996 Big Arroyo Fire. Descriptions of this fire by National Park Service fire monitors (Monica Buhler personal communication) indicate fire behavior varied between low intensity surface fire that would flare into a high intensity stand replacing canopy fire when conditions warmed and moderate to high winds occurred.

Conclusions

It has generally been assumed that fire was not common in monospecific lodgepole pine forests of the Sierra Nevada and thus played an insignificant ecological role. However, results from this fire history study contrast with this view. It suggests fire's role in lodgepole pine, in at least some areas of its range in the Sierra Nevada, may be quite different than expected and that it may have a significant ecological influence. The geographic differences that exist may relate to local variations in site productivity and the frequency and scale of disturbance.

The results also have important management implications. First, they begin to establish reference conditions for fire and its natural range of variability in this forest type in the southern Sierra Nevada. They show that widespread fires can occur in lodgepole pine-dominated forest types in this region. Lastly, they reflect how variable pre-Euro-American fire regimes and their relationships to vegetation dynamics can be. Even within nearby portions of the Sierra Nevada, care needs to be taken when extrapolating fire history results away from the local area sampled. This variability is a result of complex climatic, topographic, historic, and biotic interactions that can vary over fairly short distances. As of yet, we do not have sufficient baseline information to allow us to fully understand how these components interact to produce a specific fire regime.

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Relative Impact of Weather vs. Fuels on Fire Regimes in Coastal California[1]

Jon E. Keeley[2]

Abstract

Extreme fire weather is of over riding importance in determining fire behavior in coastal chaparral and on these landscapes fire suppression policy has not resulted in fire exclusion. There is regional variation in foehn winds, which are most important in southern California. Under these severe fire weather conditions fuel age does not constrain fire behavior. As a consequence prefire fuel manipulations have limited impact on fire spread during mass fire events, although strategic placement of fuel modifications may determine the outcome of fires under moderate weather conditions.

Introduction

Natural crown fire regimes include diverse ecosystems from California shrublands to Rocky Mountain lodgepole forests and pose special problems for fire and resource managers. Crown fire ecosystems stand in striking contrast to Southwestern ponderosa pine forests where fire suppression policy has meant fire exclusion from much of that landscape throughout the 20th century (Allen and others 2002). In general, fire suppression activities have not succeeded in excluding fire from many crown fire ecosystems (Keeley and Fotheringham 2001b, Johnson and others 2001). This is illustrated by burning patterns in the coastal ranges of central and southern California (*fig. 1*); clearly, on these landscapes fire suppression policy cannot be equated with fire exclusion.

So why is fire suppression policy in much of the Western U.S. so effective at excluding fires, but incapable of excluding fires on coastal chaparral landscapes? It certainly is not for lack of trying; in California expenditures on fire suppression have steadily increased during the 20th century (Clar 1959, Pyne 1982). Two potential factors are weather and fuels.

Role of Weather

If chaparral wildfires ignite during moderate weather conditions they are readily contained by suppression forces (Herbert Spitzer, County of Los Angeles Fire Department, May 2001). However, it is a different story when they ignite during severe fire-weather conditions, and such conditions are possible year round. In many parts of coastal California, autumn foehn winds known as Santa Anas (*fig. 2*) produce the worst fire weather conditions in the country (Schroeder and others 1964).

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[1]An earlier version of this paper was presented at the Fire Conference 2002: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, January 2–5, 2003, San Diego, California.

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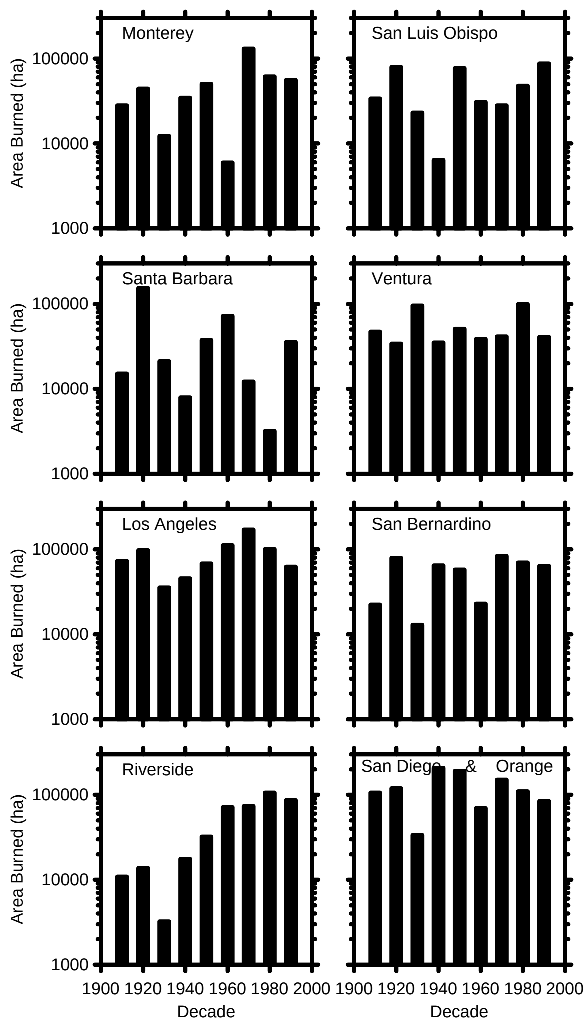

Figure 1—Area burned (log scale) by decade for the nine counties in central coastal and southern California. Data from the Statewide Fire History Data Base, California Department of Forestry, Fire and Resource Assessment Program (FRAP), Sacramento CA, which includes historical fire records from U.S. Forest Service national forests, California Division of Forestry ranger units and other protected areas, plus city and county records; minimum fire size recorded varied with agency from 16 to 40 ha.

While large wildfires throughout the western U.S. are nearly always associated with severe weather conditions (Schroeder and others 1964), southern California Santa Anas are more destructive because they typically follow 6 months or more of drought and, unlike severe fire weather in other parts of the West, Santa Anas are ***annual*** events. The role of Santa Ana winds is evident when one compares the seasonal distribution of fire in this region with other parts of the country. In the southwestern U.S. both fire occurrence and area burned peak in the summer months (Swetnam and Betancourt 1998), whereas in southern California fire occurrence peaks in early summer but area burned peaks in the autumn (Keeley and Fotheringham 2003).

In the southwestern ponderosa forests the majority of fires are ignited by summer lightning storms (Keeley and Fotheringham 2003), and atmospheric conditions at the time of ignition slow the initial spread of the fire and provide a window of opportunity for suppression before weather conditions deteriorate. In contrast, the vast majority of chaparral fires are ignited by people and thus are not greatly constrained by season. When ignited during an autumn Santa Ana wind event, they produce wildfires that are virtually unstoppable until the weather changes (Countryman 1974, Pyne 1982). Because these winds are capable of driving fire through young fuels, as well as jumping over such fuels with fire brands that can be carried more than a kilometer beyond the fire front, the ability of fire managers to alter the course of Santa Ana wind driven fires by prefire fuel manipulations, e.g., prescribed fire, is limited (Keeley 2002a).

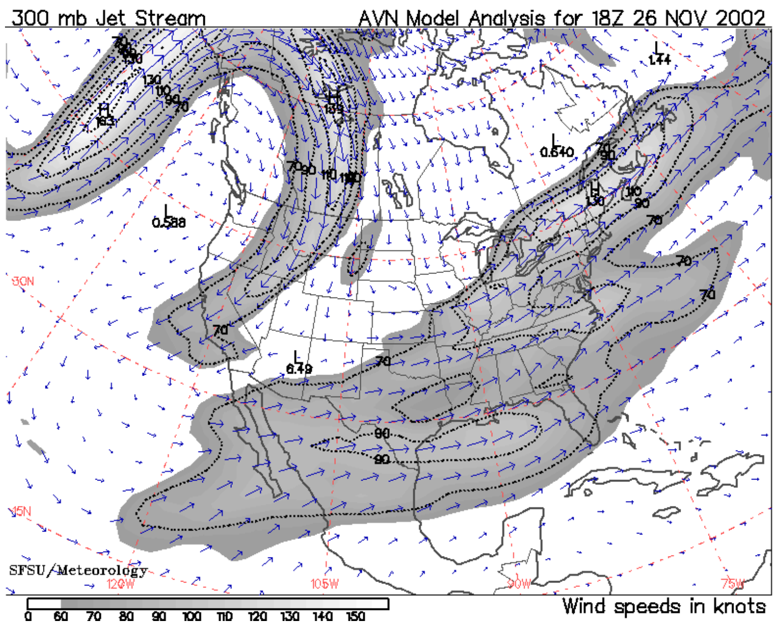

Figure 2—Wind direction during a recent Santa Ana wind event (http://squall.sfsu.edu/crws/jetstream.html, accessed 27 Nov 2002). High pressure in the Great Basin coupled with low pressure off the coast creates foehn conditions of high winds, often sustained wind speeds >100 km hr^{-1} and relative humidity <10%. Wind direction varies due to local topography (Keeley and Fotheringham 2001a) but in general it is from the north in northern California (where they are known as "northerlies") and more easterly in southern California. South of the U.S. border Santa Ana winds are less predictable and often (as illustrated here) wind direction shifts to onshore flow. The direction most Baja California wildfires burn is from west to east, whereas north of the border large fires burn from east to west (Minnich and Dezzani 1991). Further into Mexico, south of Ensenada and including the San Pedro Martir Mountains, Santa Ana winds are reportedly unknown (Jose Delgadillo, personal communication, January 2002).

In light of the low incidence of lightning fires in coastal California, coupled with the close temporal juxtaposition of late summer lightning fires and autumn Santa Anas, historically the bulk of the chaparral landscape likely burned during Santa Ana events (Keeley and Fotheringham 2003).

In addition to the synoptic weather conditions necessary for Santa Ana wind development, there are many local topographic effects related to the orientation of mountain ranges and presence of low passes to funnel the air downward (Keeley and

Fotheringham 2003). For example, the front side of the Santa Ynez Range above Santa Barbara apparently lacks Santa Ana winds (Moritz 1999). This region does have severe fire weather known as "sundowners," but these local down canyon winds are not annual events like Santa Anas. This lack of Santa Ana winds has resulted in a reduction in large fires relative to the surrounding region (Moritz and others 2004). On the west face of the southern Sierra Nevada, Santa Ana winds likewise fail to develop due to the sharp eastern escarpment, lack of appropriate passes, and diminished influence of the coastal low pressure trough. This is reflected in very different burning patterns relative to southern California. For most of the counties in coastal California fire rotation intervals are 30 to 40 yr (Keeley and others 1999). This contrasts with the southern Sierra Nevada, where 43 percent of chaparral landscape has never had a recorded fire (Keeley and Pfaff unpublished data).

Role of Fuels

The dominant fuels consumed in the surface fire regime of western forests are quite different than in chaparral crown fires. In southwestern ponderosa pine ecosystems natural fires are typically low intensity and are primarily spread by burning surface fuels of sloughed off leaves and branches. Frequent lightning ignitions in these forests are capable, under natural conditions, of consuming fuels at a rate sufficient to greatly reduce the possibility of fire spreading from surface fuels to ladder fuels that connect surface fires with tree canopies. In recent years much of this landscape has experienced an unnaturally long cycle of fire exclusion (Allen and others 2002). This is primarily due to a combination of intense grazing that diminished herbaceous fuels and a fire suppression policy that has successfully extinguished the majority of lightning fires at a relatively small size. The success of fire suppression policy in this region derives from the fact that the types of fuels that carry fire ultimately generate fires that are often easily contained.

In mature chaparral, surface fuels are generally of limited importance because of lower rates of fuel production, and most chaparral shrubs fail to self-prune dead branches. Since fire spread is through slowly accumulating canopy fuels, fire is constrained by stand age and rates of dead fuel accumulation in the canopy. Indeed, experimental manipulations of fuel structure in chamise chaparral have shown that total stand fuel volume is of less importance to fire intensity than fuel structure. Using prescription burns in *Adenostoma fasciculatum* dominated chaparral, Schwilk (2002) demonstrated that if dead branches are cut from the canopy and left on the soil surface, peak fire temperatures at both ground level and 30 cm above ground level dropped from over 250^{o}C in controls to only 100^{o}C. This was not statistically different from treatments where stand fuel volume was altered by removing these dead branches from the site.

Because chaparral fires are not spread by surface fuels, there are greater limitations on the use of prescription burning. Highly productive forests produce sufficient surface fuels to allow low intensity surface fire rotations at 5 yr intervals, greatly decreasing hazard of higher intensity crown fires (Allen and others 2002). Canopy fuels in chaparral seldom will carry prescription burns in stands less than 20 yr of age (Green 1981) and much longer when dominated by scrub oaks and other species that self-prune dead branches (Keeley and Fotheringham 2003). This limitation of prescription burning to mature chaparral stands greatly limits the benefits of prescription burning, as well as increasing the hazards of prescription burning (Keeley 2002a). These constraints, however, diminish if fires ignite outside the weather limits (of wind and humidity) for prescription burning, if the brush is first mechanically crushed, or if alien grasses invade (Keeley 2002b).

Thus, an important question is to what extent is fire regime controlled by stand age. Minnich (1998, 2001; Minnich and Dezzani 1991) has argued that fire occurrence in chaparral is entirely constrained by the rate of fuel accumulation and thus probability of burning increases with stand age. In order to rectify this model with current estimates of fire rotation intervals, he maintains that chaparral is largely immune to fire until 60 yr of age. However, empirical measures don't support that model. Schoenberg and others (2001) report that in Los Angeles County there is little change in probability of burning in chaparral stands older than 20 yr. Moritz and others (2004) applied the Weibull model to stand age maps for 10 areas from Baja California to Monterey to determine how fire hazard changes with fuel age. In all but one region (coastal Santa Barbara) fire hazard was largely independent of stand age.

The Myth of Fire Suppression Impacts on Southern California Chaparral

In his masterful account of fire history in America, Stephen Pyne (1982) laid out the fundamental paradigm of how fire suppression policy had succeeded in excluding fire from forests of the western U.S. and the subsequent impact of increasing fire hazard in these forested ecosystems. Shortly afterward Minnich (1983) invoked this model to account for differences in fire size north and south of the international border. Subsequently, it has been argued that fire suppression has grossly altered the natural fire regime in southern California chaparral (Minnich 1998, 2001, this volume). This is based on the hypothesis that fire suppression policy has excluded fire from the southern California landscape for a sufficient time to allow unnatural accumulations of fuels.

However, *figure 1* in a sense is a test of Minnich's hypothesis, and clearly it is demonstrated that fire suppression has not ***excluded*** fire during the era of 20th century fire suppression policy. Historical evidence demonstrates that the effectiveness of fire suppression increased markedly after World War II (Clar 1959, Pyne 1982, Dombeck 2001); however, in coastal California this did not result in dramatic reductions in burning (*fig. 1*). Recently Minnich (2001) has proposed that the reason there is no evidence of fire exclusion throughout the 20th century (*fig. 1*) is because fire suppression forces were effectively excluding fires during the latter decade of the 19th century and that unnatural levels of fuels began accumulating early in the 20th century. Thus, according to his model, all of the burning of the 20th century is an artifact of fire suppression. However, the history of forestry and fire protection in California reveals that fire suppression capabilities were non-existent prior to the 20th century, were weak in the early decades, and developed very gradually through the first half of the 20th century (Brown and Show 1944, Clar 1959, Pyne 1982).

In summary, large wildfires that threaten the urban/wildfire interface are a natural feature of this landscape. There is an abundance of historical evidence demonstrating large chaparral wildfires on the order of 10,000 to 100,000 ha occurred long before the fire suppression policy of the 20th century (Pyne 1982, Keeley and Fotheringham 2003, Keeley and others 2004). Prefire fuel manipulations have relatively limited ability to alter the course of such events. It is imperative that fire managers and scientists better educate the public on the inherent limitations to reducing fire hazard on these landscapes.

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Fuel Loadings in Forests, Woodlands, and Savannas of the Madrean Province[1]

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Abstract

Natural fire regimes in the southwestern United States have been significantly altered by past land-use practices and the fire suppression polices of land management agencies. One consequence of this alteration has been to increase the loadings of downed woody fuels. Ecologists and land managers are reintroducing fire into the ecosystems of the Madrean Province to reduce the excessive buildups of fuels that have a high probability of igniting and becoming unnaturally severe wildfire. However, incomplete knowledge of fuel loadings often constrains the successful reintroduction of fire because of the inability to forecast the resulting fire behavior. We have quantified fuel loadings in the pine-oak forests and oak woodlands and savannas of the Madrean Province for a range of site conditions and past land-use histories to facilitate the preparation of effective fire prescriptions.

Introduction

Natural fire regimes of the southwestern United States have been significantly altered by past land-use practices and the fire suppression policies of land management agencies. One consequence of this alteration has been to increase the loadings of downed woody fuels and, therefore, increasing the threat of high severity wildfire. In response to this situation, ecologists and land managers are reintroducing fire into the ecosystems of the Madrean Province to reduce the excessive buildups of fuels. Prescribed fire can also improve watershed conditions, wildlife habitats, and other multiple-use values. However, incomplete knowledge of fuel loadings in these arid and semi-arid ecosystems often constrains the successful reintroduction of fire because of the inability to forecast fire behavior. We have quantified fuel loadings in the extensive and diverse pine-oak forests and oak woodlands and savannas of the Madrean Province for a range of site conditions and past land-use histories to facilitate the preparation of effective fire prescriptions.

"Fuel loadings" are the oven-dry weights of fuels per unit of surface area for the size-fraction(s) of interest. Fuel loadings are also measures of the potential energy that might be released by a fire. The "downed woody fraction" of fuel loadings is the focus of this paper.

Study Area

The Madrean Province is at the convergence of the Sonoran, Chihuahuan, Madrean, and Rocky Mountain biogeographic regions and the Sierra Madre Occidentalis and Rocky Mountain ranges. Madrean ecosystems are found in

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southeastern Arizona and southwestern New Mexico and northwestern Chihuahua and northeastern Sonora in Mexico. Elevations range from 2,300 to more than 8,500 feet. The isolated mountains are characterized by pine-oak and other montane forests on the higher elevations and oak and pinyon-juniper woodlands on the lower elevations. These mountains are separated by broad plains and valley floors covered with a variety of desert-shrub and desert-grassland plant communities.

Pine-Oak Forests

Fuel loadings were measured on five sites in the pine-oak forests of the Madrean Province. Trees contributing to the downed woody fraction of fuel loadings on these sites included (in varying intermixtures) Apache (*Pinus engelmanii*), ponderosa (*P. ponderosa*), and southwestern white pine (*P. strobiformis*), alligator juniper (*Juniperus deppeana*), border pinyon (*P. discolor*), and Emory (*Quercus emoryi*), Arizona white (*Q. arizonica*), silverleaf (*Q. hypoleucoides*), and netleaf oak (*Q. rugosa*). Two of the study sites, McClure and Upper Sawmill, were in the Huachuca Mountains of southeastern Arizona; one site, El Tigre, was in the Chiricahua Mountains of southeastern Arizona; and two sites, Plantio and Pinos Grandos, were in the Sierra de Los Ajos of northeastern Sonora, Mexico. All of these sites have the same climate, vegetation, and physiography. It might be assumed, therefore, that the fuel loadings on the sites would be similar. However, their different land-uses have impacted their fuel-loading characteristics. There was a significant reduction of widespread and recurring fires with the settlement of the southwestern United States by Anglos in the late 1880s. The scarcity of tree fire scars after the 1880s is attributed largely to the effects of heavy livestock grazing and other intensive land uses that fragmented the natural landscape and, as a result, reduced the occurrence of fires over large areas (Swetnam and Baisan 1996). The fire suppression policies of public land management agencies after the 1900s have also contributed to this reduction of fires. Fire is more frequent in northeastern Sonora. Because lightning-caused fire has been and currently is infrequently suppressed, fire continues to be a landscape-structuring process throughout northern Mexico (Bojorquez-Tapia 1990). Fire regimes in northeastern Sonora have been largely unchanged for more than a century.

Emory Oak Woodlands

Fuel loadings were also measured on three sites in the Emory oak woodlands of the San Rafael Valley on the southeastern slopes of the Huachuca Mountains in southeastern Arizona. While similar in their site quality, the three sites differed in stand stocking conditions. One stand was stocked with mature Emory oak (60 yr and older) and a few scattered stump sprouts and root suckers. This stand mimicked the stocking conditions commonly encountered in unharvested Emory oak stands. Selection cutting of firewood had last taken place 20 yr earlier in the second stand, which as a result of the harvesting supported a smaller number of mature trees, few (less than 10 stems per ac) pre-harvest stump sprouts and root suckers, and a large number (350 to 400 stems per ac) of post-harvest stump sprouts. Firewood had also been selectively harvested 20 yr earlier in the third stand measured, with clumps of post-harvest stump sprouts thinned to 1, 2, or 3 of the largest and most vigorous 4 yr after harvesting to study the effects of coppice thinning on the growth and volume of the thinned Emory stump sprouts (Touchan and Ffolliott 1999). The resulting slash was left in place on both of the harvested sites.

Emory Oak Savannas

Fuel loadings were measured on 12 small watersheds situated in the Emory oak savannas on the east side of the Peloncillo Mountains of southwestern New Mexico. Characteristics of these watersheds, which vary in size from 20 to over 80 ac, have been described elsewhere (Gottfried and others 2000). The savannas on the watersheds differ from the Emory oak woodlands of San Rafael Valley in that they support fewer trees and, as a consequence, greater amounts of herbage. These watersheds have been established by the USDA Forest Service to evaluate the impacts of prescribed burning treatments on the hydrology and ecology of oak savannas in the southwestern United States. The information on fuel loadings presented in this paper should help in the preparation of these prescriptions.

Methods

Fuel loadings on each site were measured on a randomly located 2.5 ac grid of 25 systematically located plots established at equally spaced intervals. Errors of 20 percent or less, adequate levels of precision for most fuel loading inventories (Brown 1974), were obtained with these plot numbers. Plot centers were the starting points for randomly oriented planer-intersects that were established to measure the fuels. The planar-intersect technique has the same theoretical basis as the line-intersect method (Van Wagner 1968). It involves counting the intersections of fuels with vertical sampling planes that resemble guillotines dropped through the accumulated fuels. A similar grid has been used in studies to measure fuel loadings in other pine-oak and montane forest ecosystems in the region (Harrington 1985, Fule and Covington 1995, Arno and others 1997).

Diameter classes of the fuels that were intersected by the planar transects were tallied by the classification of Brown (1974), that is, sound wood fuels 0.1 to 0.25 inch, >0.25 to 1 in, >1 to 3 in., and >3 in. and decaying (rotten) woody debris >3 in. These diameter classes correspond to the 1-hour, 10-hr, 100-hr, and 1,000-hr time-lag fuel-moisture classes of the National Fire Danger Rating System (Pyne and others 1996), respectively. Sound and decaying fuels >3 inches in diameter were also classified as “coarse woody debris,” which includes limbs, stems, and roots of trees and shrubs in varying stages of decay (Graham and others 1994). Tallies of the respective diameter classes with their standard specific gravity values were used in determining fuel loadings in tons per acre.

Results and Discussion

Means and standard errors of the fuel loadings of downed woody fuels for the measurement sites are summarized in *table 1*. These loadings have been arbitrarily separated into those of “smaller fuels” up to 1 inch in diameter, “larger fuels” greater than 1 inch in diameter, and “coarse woody debris” for presentation and discussion purposes. Again, coarse woody debris includes all fuels >3 inches in diameter.

Smaller Fuels

Fuel loadings in these smaller diameter classes were heavier in the pine-oak forests and Emory oak woodlands than in the Emory oak savannas, but there was no difference between the pine-oak forests and Emory oak woodlands (*table 1*). This finding was not surprising, however, as there was a higher stocking of trees and in

Table 1—Means and standard errors of loadings of downed woody fuels.

	Fuels/wood (in.)					**Coarse woody debris**	**Total fuel loading**
	Small		**Large**				
	sound		**sound**		**decaying**	**all fuel**	**downed**
Site	**0.1-0.25**	**>0.25-1**	**>1-3**	**>3**	**>3**	**>3**	**fuel**
Pine-oak forest				*tons per acre*			
McClure	0.25 0.0091	0.40 ± 0.018	1.0 ± 0.091	1.8 ± 0.38	6.5 ± 0.22	8.3 ± 0.44	9.9 ± 0.46
Upper sawmill	0.16 ± 0.0073	0.56 ± 0.019	3.0 ± 0.093	8.7 ± 0.40	0.9 ± 0.19	9.6 ± 0.19	13.3 ± 0.47
El Tigre	0.73 ± 0.0092	0.98 ± 0.036	4.1 ± 0.090	12.6 ± 0.39	3.2 ± 0.20	15.8 ± 0.46	21.6 ± 0.48
Plantio	0.15 ± 0.012	0.16 ± 0.021	0.9 ± 0.12	3.3 ± 0.48	0.6 ± 0.27	3.9 ± 0.52	5.1 ± 0.48
Pinos Grandos	0.14 ± 0.011	0.22 ± 0.020	1.3 ± 0.26	3.1 ± 0.46	0.3 ± 0.26	3.4 ± 0.54	5.1 ± 0.42
Emory oak woodlands							
Unharvested stand	0.29 ± 0.0026	0.39 ± 0.0034	0.55 ± 0.011	0	0.12 ± 0.0048	0.12 ± 0.0048	1.3 ± 0.022
Harvested stand	0.22 ± 0.0019	1.2 ± 0.0086	0.77 ± 0.014	2.5 ± 0.065	0.34 ± 0.0094	2.8 ± 0.071	5.0 ± 0.10
Harvested and Thinned Stand	0.13 ± 0.0013	0.77 ± 0.0068	1.5 ± 0.020	1.8 ± 0.033	0.33 ± 0.013	2.1 ± 0.026	4.5 ± 0.074
Emory oak savannas							
A	0.0081 ± 0.00022	0.059 ± 0.0017	0	0	0	0	0.067 ± 0.0019
B	0	0.063 ± 0.0024	0.33 ± 0.0073	2.4 ± 0.98	0	2.4 ± 0.98	2.8 ± 0.11
C	0.043 ± 0.00091	0.28 ± 0.0049	0.22 ± 0.0061	0.38 ± 0.010	0	0.38 ± 0.010	0.92 ± 0.022
E	0.013 ± 0.00032	0.20 ± 0.0071	0.55 ± 0.011	4.4 ± 0.10	0	4.4 ± 0.10	5.2 ± 0.12
F	0.024 ± 0.00061	0.10 ± 0.0023	0.11 ± 0.0044	0	0	0	0.23 ± 0.0073
G	0.044 ± 0.00094	0.082 ± 0.0025	0	1.3 ± 0.052	0.19 ± 0.0076	1.5 ± 0.061	1.6 ± 0.062
H	0.022 ± 0.00045	0.20 ± 0.0043	0.44 ± 0.010	31.8 ± 0.49	0	31.8 ± 0.49	32.5 ± 0.51
I	0.023 ± 0.00073	0.20 ± 0.0047	0.44 ± 0.014	0	0	0	0.66 ± 0.019
J2	0.012 ± 0.00021	0.16 ± 0.0036	0	14.4 ± 0.41	0	14.4 ± 0.41	14.6 ± 0.41
K	0.083 ± 0.0015	0.36 ± 0.0065	0.22 ± 0.0088	0.79 ± 0.022	0	0.79 ± 0.022	1.4 ± 0.038
M	0.025 ± 0.00043	0.083 ± 0.0019	0.44 ± 0.0082	25.8 ± 0.62	3.7 ± 0.015	29.5 ± 0.63	30.0 ± 0.65
N	0.063 ± 0.0011	0.53 ± 0.010	0.44 ± 0.014	0.19 ± 0.0076	0	0.19 ± 0.0076	1.2 ± 0.032

pine-oak forests and Emory oak woodlands. Within the pine-oak forests, the sites in southeastern Arizona averaged about three-times the fuel loadings as those in northeastern Sonora, presumably because of the infrequent occurrence of fire in the former. Smaller fuels are more readily ignited and consumed by fire than are larger fuels and, therefore, less likely to accumulate and persist in large amounts on sites experiencing repeated or recent fire. El Tigre, a site with no known fires since 1900, had the heaviest loadings of downed woody fuels in these diameter classes.

Fuel loadings in the Emory oak woodlands of San Rafael Valley were heaviest on the site where cutting of firewood had last taken place 20 yr earlier. Subsequent mortality of less dominant post-harvest stump sprouts of low vigor followed by their eventual falling to the ground is the main contributor to these heavy accumulations of fuels in these diameter classes. At the opposite end of the spectrum of measurements, fuel loadings on the unharvested site were indicative of those found in Emory oak woodland stands representing “natural” stocking conditions.

Fuel loadings in these smaller diameters classes were almost 10 times less in the Emory oak savannas in the Peloncillo Mountains than those measured in the Emory oak woodlands. The few trees and occasional shrubs found in these savannas are the causal factor for these low fuel accumulations.

Larger Fuels

Loadings of downed woody fuels in these larger diameter classes were similar among the three broadly defined ecosystems studied (*table 1*). These larger fuels tend to ignite and burn more slowly than the smaller fuels and, consequently, might not be totally consumed in a fire of low to medium severity. However, the larger fuels can burn vigorously with fire of high severity and the rate of their reduction is greater than in the case of smaller fuels. Once again, the sites in pine-oak forests of the southeastern Arizona averaged about three-times the fuel loadings as those in northeastern Sonora. The general pattern of loadings for both smaller and larger downed woody fuels in the pine-oak forests is that lesser loadings occurred on the sites with a more frequent occurrence of fire (Escobedo and others 2001).

Loadings of larger downed woody fuels in the Emory oak woodlands followed the same general sequence as noted for the loadings of smaller fuels. That is, the heaviest fuel loadings were found on the site where firewood cutting had taken place 20 years earlier and the lowest loadings on the unharvested site.

Heavy loadings of sound wood >3 inches in diameter on three of the small watersheds in the Emory oak savannas contributed significantly to the overall heavy loadings of larger downed woody fuels in these ecosystems. Much of the variability in the loadings of larger fuels is attributed to the infrequent occurrence of sound wood fuels in these larger diameter classes on the three watersheds. The estimated loadings of these fuels were skewed to the “high end” of the range in the few instances when the larger fuels were measured on the planar-intersects.

Coarse Woody Debris

The range of coarse woody debris loadings in the pine-oak forests observed in this study (*table 1*) is comparable to the loadings reported in other pine-oak forests of the Madrean Province (Sackett 1979, Harrington, 1981, Alanis-Morales 1996).

However, the loadings of coarse woody debris that are required to sustain the "health" of the soils in the pine-oak forests of the Madrean Province are not known. Graham and others (1994) concluded that loadings of coarse woody debris from 5 to about 25 t ac^{-1} were "optimal" to maintain the mycorrhizal activity of soils in the forest types of the Rocky Mountains. Lesser loadings are likely to disrupt the functioning of soil that is necessary to sustain a "high level" of plant productivity in these forests. Little is known about the levels of coarse woody debris necessary to sustain the efficient cycling of nutrients in the Emory oak ecosystems.

Total Loadings of Downed Woody Fuels

Total loadings of downed woody fuels in the ecosystems studied where similar when all of the measurement sites are considered together (*table 1*). All of the study sites in pine-oak forests had experienced some timber harvesting in the past and, therefore, unknown amounts of decaying logging slash likely contributed to the total fuel loadings measured in this study. The total loadings of downed woody fuels found in the pine-oak forests were similar to values reported for ponderosa pine forests in the southwestern United States in earlier studies (Sackett 1979, Harrington 1981, Graham and others 1994).

Total loadings of downed woody fuels in the Emory oak woodlands of the San Rafael Valley were heaviest on the sites where firewood had been selectively harvested in the past. Firewood cutters are required to remove stems and limbs that are >2 inches in diameter from the site in these harvesting operations (Bennett 1992). The smaller residuals left on site contribute significantly to the total fuel loadings of a site. The comparatively low total loadings in the unharvested stand are thought to represent the downed woody fuels associated with "pristine" woodland conditions.

Total fuel loadings in Emory oak savannas of the Peloncillo Mountains were largely dependent on the occurrence of woody fuels > 3 inches. Values exceeding 10 t ac^{-1} were measured when comparatively large quantities of material were present. Otherwise, the loadings of downed woody fuels were 1 t ac^{-1} or less.

Conclusions

While both site-specific and specific to the size-fraction of the fuel loadings measured, the information presented in this paper should help in improving the knowledge fuel conditions necessary to prepare prescriptions for the successful reintroduction of fire into the Madrean Province. Depending on the loading that might be acceptable for a site, an initial schedule of introduced fire occurrences could be prescribed, tested, and evaluated relative to maintaining the fuel loading at the acceptable level while attaining other land management objectives. This schedule could then be modified as necessary through time to sustain the desired fuel loading conditions and multiple-use values.

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One-Year Postfire Mortality of Large Trees in Low- and Moderate-Severity Portions of the Star Fire in the Sierra Nevada[1]

Jo Ann Fites-Kaufman,[2] David Weixelman,[2] and Amy Merrill[2]

Abstract

Following the Star Fire, 287 trees greater than or equal to 75 cm diameter at breast height (DBH), were tracked for post-fire mortality. The Star Fire, located in the western Sierra Nevada, burned mostly at low to moderate intensity during late summer of 2001. The mixed conifer forest was dominated by *Psuedotsuga menziesii* (Douglas-fir), *Abies concolor* (white fir), *Pinus labertiana* (sugar pine), and *Libocedrus decurrens* (incense cedar). Several *Pinus jeffreyi* (Jeffrey pine) and *P. ponderosa* (ponderosa pine) were also sampled. Trees were measured for crown length, height of crown scorch, live crown height, and bole scorch within several weeks of the fire in the fall 2001. Species, DBH, and location of each tree were recorded. After 1 yr, 92 percent of the trees were still alive, although 8 percent appeared in poor condition. Four percent had died and were standing and a little over 1 percent had died and fallen. Two percent were cut as hazard trees. These mortality rates were compared with those predicted in the U.S. Department of Agriculture (USDA), Forest Service's fire model, First Order Fire Effects Model (FOFEM). Logistic regression was used to develop models to predict probability of mortality. Independent variables were DBH, percent live crown height, percent live crown volume, and species. For all species, percent live crown height was the best predictor of mortality. Results for individual species varied. Percent live crown height was the best predictor of mortality for Douglas fir, live crown height was the best predictor of mortality for white fir, and percent live crown volume was the best predictor of mortality for Sugar pine.

Introduction

In order to plan post-fire management after both wildfires and prescribed burns, managers need to be able to predict fire-induced tree mortality rates. Past research has demonstrated strong relationships between fire mortality and tree characteristics such as proportion of live crown killed or scorched by fire (Ryan 1982, Ryan and others 1988, and Peterson 1985). Bark thickness has also been found to be an important correlate with fire-induced mortality (Ryan and Reinhardt 1988). Mortality rates can vary among species and among regions. Pacific Northwest conifer species have been ranked according to their fire resistance based on differences in bark thickness, rooting depth, and crown form and development (Brown and Davis 1973, Minore 1979).

California sites and species are underrepresented in commonly used models that predict fire mortality. There is great interest in mortality of large and old trees from wildfires and prescribed burns due to their importance to key wildlife species and old growth forests, yet few studies have focused on these trees. The Star Fire burned

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16,800 ac on the Tahoe and Eldorado National Forests in California (*fig. 1*), within the western Sierra Nevada Mountain Range, late summer of 2001. Fire intensities ranged from high to moderate and low. The objectives of this study are to 1) determine post-fire mortality rates in large trees ($\geq$76 cm diameter at breast height-DBH) in areas burned at low to moderate intensity; and 2) develop statistical models to predict post-fire mortality from fire damage to crown, species, and diameter.

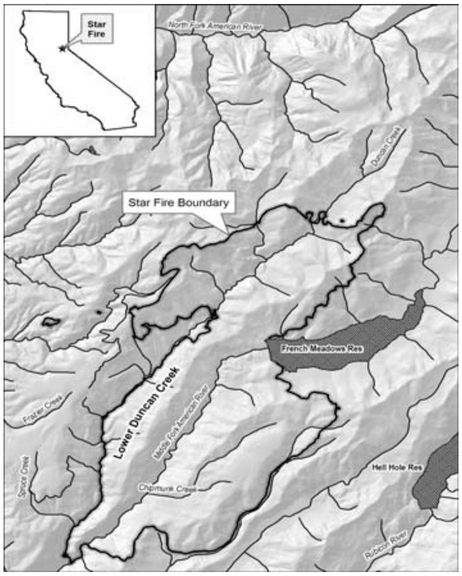

Figure 1—Project Location in California.

Methods

We made a complete census of large trees ($\geq$76 cm DBH) across a portion of the Star Fire. The lower portion of Duncan Canyon, which burned primarily at moderate to low intensities, was the focus landscape area (*fig. 1*). In early fall 2001, shortly after the fire, 287 trees were tagged, mapped (using Global Positioning System (GPS)), and crown damage, species, and size of trees recorded. The mixed conifer forest is located west of Lake Tahoe and was dominated by *Psuedotsuga menziesii* (Douglas-fir), *Abies concolor* (white fir), *Pinus labertiana* (sugar pine), and *Libocedrus decurrens* (incense cedar). Only two *Pinus jeffreyi* (Jeffrey pine) and three *P. ponderosa* (ponderosa pines) were sampled; therefore, these individuals were only included in the analysis that addressed all species. Mortality rates of other species were considered both together and separately.

Initially, we measured crown length, height of crown scorch, and bole scorch using an Impulse Laser apparatus. Species, DBH, and location of each tree were recorded. One year later, we visited each tree again and recorded its status as dead or alive.

Tree crown volumes were calculated based on crown height and crown width (measured along two axes). We assumed crown shape was a truncated ellipse (volume=0.75*height * PI*r^2) for all species except sugar pine, which we assumed had a general cylinder form (volume=height * PI*r^2; Van Pelt and North 1996, 1999). Percentage of live crown volume was calculated as the ratio of calculated live over calculated total crown volume (based on field measurements of total height, scorch height, and radius) times one hundred.

We performed binary logistic regressions to determine the relationship between mortality and the dependent variables percent live crown volume, percent live crown height, species, and DBH. Autocorrelation among variables was tested prior to analyses. A maximum likelihood fitting procedure was applied to estimate coefficients of the linear predictor, and fitted values were back-transformed via the logit link to predict the probability of mortality. Mortality was coded as 1 (dead) or 0 (alive). Statistical Package for the Social Science (SPSS) 10.0 for Microsoft Windows was used for all data analyses.

Results and Discussion

The descriptive statistics for trees greater than or equal to 76 cm DBH were measured shortly after the Star Fire (*table 1*). Data from the trees sampled, by species, were used to calculate group means and standard errors for DBH, meter and percentage of live crown height, and meter and percentage live crown volume. All trees measured were within low to moderate fire intensities from the Star Fire.

Table 1—*Descriptive statistics for trees (≥76 cm DBH) in the study area within the Star Fire, Tahoe National Forest. Measurements are of group means and standard errors.*

Species	No. of trees	DBH (cm)	Live crown		Pct live crown	
			Height (m)	Volume (m^3)	Height	Volume
White fir	81	96.1 + 2.2	17.7 + 1.1	597.4 + 54.6	63.2 + 3.4	49.1 + 4.0
Incense cedar	37	98.4 + 2.7	10.0 + 1.0	252.5 + 38.3	55.1 + 5.9	43.0 + 6.3
Sugar pine	106	117.4 + 2.9	17.6 + 0.9	1998.2 + 210.8	68.1 + 3.2	43.0 + 2.9
Douglas fir	58	97.3 + 2.0	17.2 + 1.4	1067.7 + 128.4	67.7 + 4.8	59.1 + 5.5

Mortality Rates

After one year, most of the trees had survived the low to moderate intensity fire in this portion of the Star Fire (*table 2*). Ninety-two percent of trees were still alive, although 8 percent appeared in poor condition. Four percent had died and were standing and a little more than one percent had died and fallen. Two percent were cut as hazard trees.

Table 2—*Mortality numbers and percentages of dead trees ≥76 cm DBH 1 yr following low to moderate fire in northern Sierra Nevada mixed conifer forest.*

Species	No. of trees alive (dead)	Pct mortality
White fir	79 (2)	2.5
Incense cedar	34 (3)	8.1
Sugar pine	100 (6)	5.7
Douglas-fir	54 (4)	6.9

Predictive Mortality Models

Based on the logistic regressions run, significant ($p<0.05$) models were identified for all species combined and for sugar pine, Douglas-fir, and white fir species independently. None of the variables measured were significantly related to incense cedar mortality. Independent variables with the best fit are presented in *table 3*. Based on the models, percentage of live crown height, crown height, and percentage of live crown volume were the best indicators of tree mortality. Although DBH has been found to be a significant variable by others (e.g., Mutch and Parsons 1998), it was not found to be true in this data set, perhaps due to the restricted range of tree sizes included in this study.

After the Star Fire, remaining live tree crown heights ranged from 0 to 100 percent of total crown height. Predicted probability of mortality never exceeded 40 percent in any of the four models (*fig. 3*), regardless of the amount of remaining live crown. This is a reflection of the very low mortality rates observed in the field for all species one year following the fire. For three of the four species models, live crown height (either percentage of or actual) was used as the independent variable to predict probability of mortality. Similarly, Bevins (1980) used scorch height to predict mortality in Douglas-fir trees following fire. However, few others have reported live crown height as an effective predictor variable for post-fire tree mortality. The sugar pine model was the only one for which live crown volume was the best predictor variable for tree mortality. Others have also found percentage of crown scorch volume to be the best predictor variable (Wagener 1961, Methven 1971, Dieterich 1979, Peterson 1983, Mutch and Parsons 1998). One possible reason for the difference in significant variables among species could be the accuracy of volume estimates among species. Peterson (1985) found that calculated crown volumes were more accurate for species classified as having cylindrical (e.g., sugar pine) rather than paraboloid or cone crown shapes (e.g., the other species in this study).

Table 3–*Logistic regression parameters for models with all species and individual species. Only parameters that were significant predictors of mortality are included.*

Model	Variable	Estimated regression coefficient	Wald χ^2	Sig.
All species (n=287)	pct live crown height	0.966	18.271	<0.001
Sugar pine (n=106)	pct live crown height	1.158	4.678	<0.001
Douglas fir (n=58)	pct live crown height	0.971	4.262	0.039
White fir (n=81)	live crown height	0.787	4.813	0.028

[1]The exp (B) reflects the change in odds of the dependent variable (e.g., tree mortality) for a unit change in the independent variable (e.g., pct live crown height).

The odds ratio (exp (B); Hosmer and Lemeshow 2000) for percentage of live crown volume in the sugar pine model indicates a rapid increase in predicted probability of mortality with every unit decrease in percentage of live crown volume below 20 percent (*table 3, fig. 3b*). The odds ratios for the other models are not as high as that for the sugar pine (*table 3*). Results from the white fir model similarly indicate that a possible threshold exists for the amount of scorch damage a crown can sustain and survive. The predicted probability of mortality for white firs, with less than 10 m of live crown remaining, increases rapidly for every unit decrease in remaining live crown height (*fig. 3d*). The more rapid predicted mortality response in the sugar pine and white fir could reflect these species' greater sensitivity to fire than

Douglas-fir or ponderosa and Jeffrey pine (Minore 1979). Similar thresholds are not indicated from the Douglas-fir model; instead this model suggests a fairly steady increase in predicted probability of mortality for every unit decrease in percentage of live crown height (*fig. 3c*). Data collected on tree mortality on these sites in coming years will be important in determining longer term mortality response.

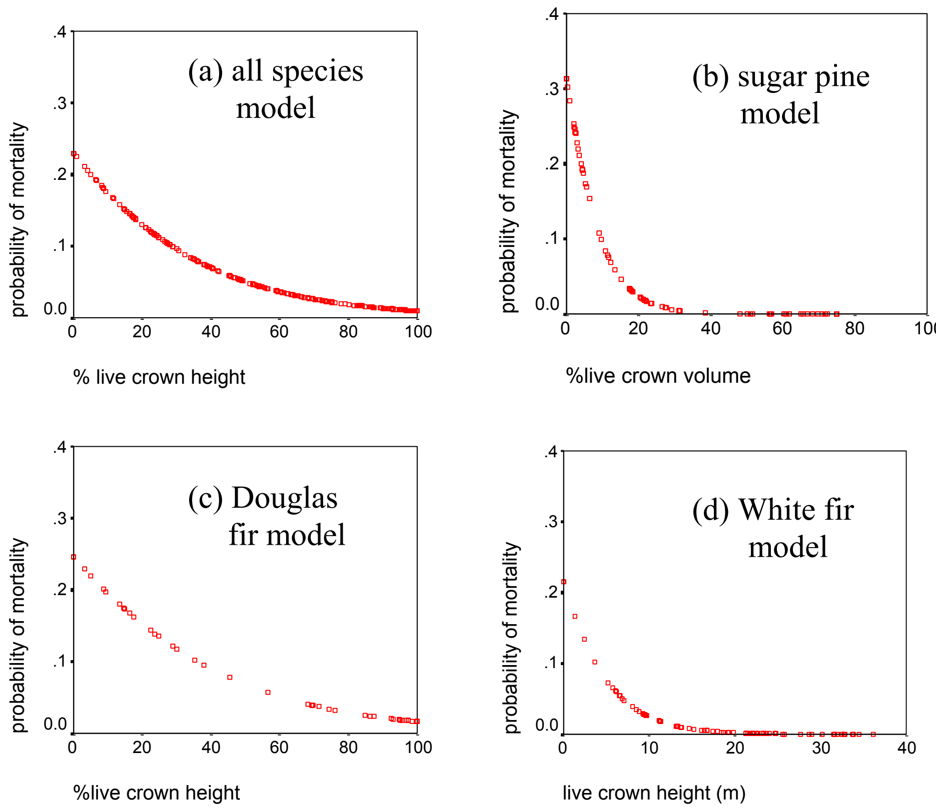

Figure 3—Scatter plots of model-predicted mortality and live crown volume or height. The best predictor variables varied based on which species were included in the model.

Conclusions

Models predicting mortality rates among conifer species in the Sierra Nevada can be used to develop silvicultural prescriptions related to prescribed burns, and can be used to mark high-risk trees for salvage directly after wildfires or prescribed burns. Variations among species and geographic location affect mortality from fires. Empirical data that covers a range of species, sizes, and physical settings should be used to develop models that advise management decisions. In this study, post-fire mortality rates among three conifer species common to mixed conifer forests of the California Sierra Nevada were significantly related to percent live crown height and percent live crown volume.

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Estimating Forest Fuels in the Southwest Using Forest Inventory Data[1]

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Abstract

Catastrophic wildfires occurring over the last several years have led land management agencies to focus on reducing hazardous fuels. These wildland fuel reduction projects will likely be concentrated in shorter interval, fire-adapted ecosystems that have been moderately or significantly altered from their historical range. But where are these situations located? What are their fuel characteristics? Who owns them? Describing fuel characteristics on these lands is not simple, but Forest Inventory and Analysis (FIA) data may be helpful. One objective of this study was to demonstrate the linkages between forest inventory data and hazardous fuel characteristics and to identify information gaps and needed relationships. A second objective was to estimate and contrast overstory and understory biomass, especially in high fire-risk areas. Restricting analysis to Arizona, New Mexico, and Utah, we estimate that understory biomass accounts for 4 to 8 percent (20 to 42 million tons) of total forest biomass. Additionally, we estimate that around 57 percent (619 million tons) of the estimated 1.08 billion tons of biomass is found on high fire-risk forest lands. Of these 619 million tons, approximately 434 million tons is associated with larger diameter ($\geq$10 inches) overstory trees, both live and dead, and most is found on non-reserved forestlands administered by the USDA Forest Service. Some of the problems we encountered included a lack of widely-applicable understory biomass equations, no equations for estimating tree seedling biomass using percent cover, and many biomass equations for shrubs that use diameter, a measurement that is not collected by FIA.

Introduction

Years of fire suppression have led to forest densification and unnatural buildups of brush and small diameter trees. Consequently, a major focus of the National Fire Plan (USDA Forest Service and U.S. Department of Interior 2000) is to reduce hazardous fuels. According to the Cohesive Strategy (USDA Forest Service 2000), wildland fuel reduction projects will be concentrated in shorter interval, fire-adapted ecosystems that have been moderately or significantly altered from their historical range of fire frequency, severity, and density of understory vegetation. But where are these hazardous situations located? What are their fuel characteristics? Who owns them? This information is needed for broadscale assessments and fuel treatment planning.

Describing fuel characteristics on these lands is not simple, but forest inventory data may be helpful. The USDA Forest Service's Forest Inventory and Analysis (FIA) program is administered through five regional centers (Southern, Northeastern, North Central, Pacific Coast, and Interior West), each of which conducts resource

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inventories on all forested land within their geographical area. This system of inventory collection allows for a consistent set of data that can be used to produce standardized tables describing a wide array of forest measurements, such as acres of forest and timber land, growing stock volume, and biomass, along with a wide range of resource characteristics (ownership, species, size, etc.). Consistency in data collection allows for nationwide comparisons of forest attributes. If one wishes to estimate, for example, the volume of small-diameter timber for various states, differences found between states can be attributed to actual, physical differences, not to inconsistencies in what data are available or how data are collected or measured. Additionally, FIA data include forest inventory information on all ownerships, allowing comparisons among agencies or between Federal and private lands.

This paper discusses the linkages between forest inventory data and hazardous fuel characteristics associated with high-risk forest situations. Forest fuel, measured as tons of biomass, will be described for both forest overstory and understory vegetation. The overstory contains large and small diameter trees. Understory vegetation focuses on three layers. The lowest layer, from ground to the knee, contains grasses, forbs, and low shrubs. The next layer, between the knee and eye level, contains forbs and medium shrubs. The highest layer includes plants above eye level, usually seedlings, saplings, and tall shrubs only. Information gaps and needed relationships are identified.

Methods

Our main objective was to assess the extent to which FIA data could be used to describe forest fuels on a broad scale. In particular, we wanted to use FIA data to describe forest fuels, not only in the overstory, but also in the understory. To do this, we needed to determine the types of information collected by FIA and how that information could be used to describe forest fuels. We decided upon biomass as a measure that could be used to estimate amounts of forest fuel and allow us to compare the overstory and understory. Analysis was restricted to Arizona, New Mexico, and Utah, the three southwestern states for which we could obtain relatively recent and complete FIA data. Our major focus was on forest lands that are at special risk for catastrophic wildfires.

Data Compilation

We began data compilation by investigating the types of FIA data available. FIA collects and estimates a large variety of information on the forest overstory (trees greater than 1" diameter) including biomass for both live and dead trees. (For a listing of the types of information collected on field plots by FIA, see http://www.fs.fed.us/rm/ogden/mission/iwrime.htm). However, the only widely-available understory information is percent cover by vegetation layer (0 to 1.5 ft, 1.6 to 6 ft, and 6.1 ft or greater) and life form (trees, shrubs, grasses, forbs), and this information is collected for live vegetation only. Within the last several years, information on down woody debris has begun to be collected, but the collection of this information is in its infancy and, therefore, was not available for any State in its entirety. This information is collected on what are called "P3" or Phase 3 plots, which are a by-product of the Forest Health Monitoring/FIA merger in 2000. Approximately one in 16 FIA plots is also sampled for P3 data using one to two additional field personnel to collect data items and samples, including information on down woody debris (Rogers, personal communication).

Once we determined availability of fuels information in the FIA data, we selected a measurement metric of forest fuels that would allow us to compare fuels in the overstory to those in the understory. We decided on biomass for several reasons: 1) fuel loadings are often measured in terms of biomass, 2) biomass was already estimated for the overstory by FIA, 3) biomass is a useful measure of potential material for bio-based products, and 4) we knew of some equations for computing biomass of forbs and grasses using percent cover. We then obtained overstory data from the Interior West Forest Inventory and Analysis (IWFIA) unit in Ogden, Utah. For the understory data, IWFIA calculated average percent cover by layer and life form for each inventory plot, averaged across all subplots on which information was collected. Understory data were then merged with overstory plot data.

We were particularly interested in assessing the amount of forest fuels on lands at risk for severe wildfires. The Cohesive Strategy states that its aim is to "reduce losses and damages from wildland fires by concentrating treatments where human communities, watersheds, and species are at risk" (USDA Forest Service 2000). To determine the location of these areas, we used a map of coarse-scale spatial data developed by Hardy and others (1999). These data designate both Historic Natural Fire Regime (historic fire frequency and severity) and Current Condition Class (an indication of the degree of departure from the historic fire regimes). This map was sent to IWFIA where it was overlaid with FIA plots, and each plot was assigned a Fire Regime/Condition Class depending upon where the plot fell on the map. We designated plots falling in Fire Regime I or II and Condition Class 2 or 3 as "high risk." Almost all of the lower elevation zones in the United States, which are the areas most affected by human intervention and where resources and communities are at highest risk, fall under Fire Regimes I and II. In addition, Condition Classes 2 and 3 have been moderately or significantly altered from their historical range and, therefore, are more at risk for severe wildland fires and generally require some degree of mechanical treatment before prescribed fires can be successfully used to control fuels (Schmidt and others 2002).

We encountered several problems during data compilation, including the lack of useable information on down woody debris and no dead understory vegetation information. Additionally, several plots were missing understory vegetation information due to snow cover at the time the plot was visited. For these plots, we used the average percent cover by layer and life form for the appropriate forest type.

Calculating Biomass

To calculate the biomass of live and dead overstory vegetation (trees≥1" diameter), we used the algorithms designed by FIA for use with their data (Miles and others 2001); this was a straightforward process. However, for understory vegetation, we had to use equations relating biomass to percent cover. This turned out to be much more difficult than originally envisioned.

Several problems were encountered in converting FIA understory vegetation attributes of percent cover by layer and life form to biomass. First, we were unable to locate any equations for estimating the biomass of tree seedlings based on percent cover or the number of seedlings. This meant we were unable to estimate biomass of tree seedlings (trees <1 in. diameter), thus leaving a gap in our analysis. Second, many of the equations for estimating biomass of shrubs or other woody plants used diameter as an estimation parameter (Alaback 1986, Reeves and Lenhart 1988). Shrub stem diameters were not collected by FIA. Third, most biomass equations for the understory were developed for very specific geographical areas and specific species (Alaback 1986, Alexander 1978, Means and others 1996, Reeves and Lenhart 1988). Little work has been done to develop more generally applicable equations or to see how well equations developed for a specific species estimate biomass of other species.

In the end, we used understory biomass equations that were based on a mix of species, rather than equations developed for one specific species. These mixed-species equations had been developed for other regions of the country and their applicability to the southwest is questionable; however, we felt these equations could be used for illustrative purposes and give some idea of the relative understory biomass of the three states. For grasses and forbs, we used equations developed by Mitchell and others (1997) for estimating biomass based on percent cover. For shrubs, we adapted an equation developed by Olson and Martin (1981) that used two parameters, percent cover and height (*table 1*). Although FIA does not specifically measure height of understory plants, percent cover by vegetation layer is recorded. We used the midpoints of the first two layers (0.75 ft, 3.8 ft) as an estimate of the height of the understory vegetation. For the third layer ($\geq$6.1 ft), we assumed that, on average, shrubs ranged from 6.1 ft to around 15 ft, so we used a midpoint of 10.5 ft.

Table 1—*Regression equations used for estimating biomass of forest land understory vegetation in Arizona, New Mexico, and Utah*

Life form	Parameters	Equation[1]	Source
Shrubs	Pct cover (x_1) Height (x_2)	G $0.5m^{-2}$ = -.62689 + (0.05778* x_{1*} $x_{2)}$	Olson and Martin 1981
Forbs	Pct cover (x_1)	kg ha^{-1}=13.66* x_1	Mitchell and others 1987
Grasses	Pct cover (x_1)	kg ha^{-1}=8.17*x_1	Mitchell and others 1987

[1]Dependent variable y=oven-dry plant weight. Units varied, but all results were converted to lb ac^{-1}.

Results

The results presented in this section are largely illustrative of the types of forest fuel information that can be extracted from FIA data. Estimates of area and overstory biomass are standard outputs from the FIA database. Understory estimates of biomass involve more extrapolation. We also provide information on the relative amounts of biomass found on lands at risk for severe wildland fires by merging the FIA data with current condition class and historic natural fire regime information.

There are 51.8 million acres of forest land in Arizona, New Mexico, and Utah (*fig. 1*). The majority of this land (68 percent) is classified as woodland (forest land at least 10 percent stocked with woodland species trees of any size). Only one-fourth of the forest land is classified as timberland (forest land at least 10 percent stocked with timber species trees). Each of the three states has more than 10 million ac of woodland, with Arizona containing both the largest area of woodland and hence, the largest area of forest land. None of the states have more than 5 million ac of timberland. Only 8 percent of the forest land has been withdrawn from tree utilization through statute or administrative designation (reserved land).

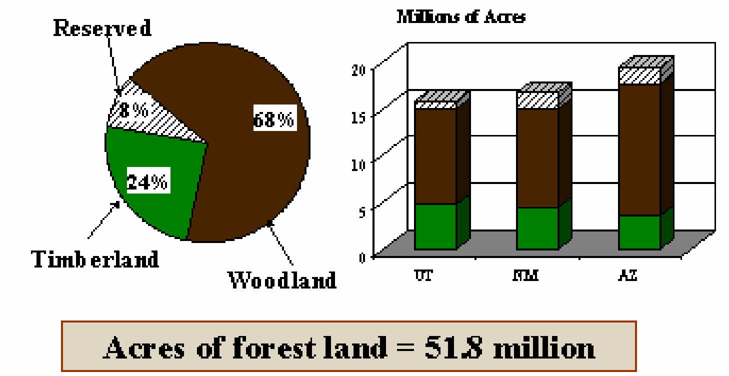

Figure 1—Acres of forest land in Arizona, New Mexico, and Utah.

Total biomass found on forest land in these three states comes to an estimated 1.08 billion tons (not counting tree seedlings) (*fig. 2*). Most of this, 96 percent, is found in the overstory, and the majority of the overstory biomass comes from live trees. Interestingly, although Arizona contains the largest area of forest land (*fig. 1*), Utah contains the largest amount of total biomass, more than 390 million tons, and also contains the largest amount of understory biomass.

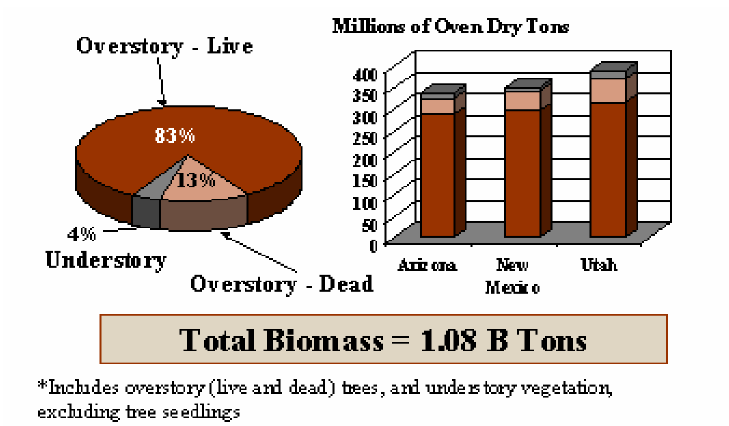

Figure 2—Estimated biomass of overstory and understory vegetation*on forest land in Arizona, New Mexico, and Utah.

Estimates of understory biomass do not include tree seedlings (trees <1 in. diameter) due to the lack of equations for estimating seedling biomass. Just how much did this likely affect our estimates? We found one reference that suggested that seedling biomass comprises anywhere from 5 to 50 percent of total understory biomass (Telfer 1971). To test how the missing seedling information affected our estimates, we recalculated our estimates by including tree seedlings as 5, 25, and 50 percent of total understory biomass (*table 2*). Assuming that tree seedlings account for 5 percent of understory biomass, there would be no noticeable affect on understory biomass estimates. If tree seedlings were 25 percent of understory biomass, the total understory percentage increased to 5 percent, as opposed to 4 percent with no tree seedlings included. When we assumed that tree seedlings make up 50 percent of understory biomass, the percentage of biomass accounted for by the understory increased to 8 percent. At this level, the importance of the understory began to vie with that of the dead overstory trees, and our overall biomass estimates increased from 1.08 billion tons to 1.13 billion tons.

Table 2—*Sensitivity of forest land understory biomass estimates to missing tree seedling information: Arizona, New Mexico, and Utah*

Tree seedling assumption	Estimated understory biomass using tree seedling assumptions	
Pct tree seedling biomass of total understory biomass	Total understory biomass	Understory biomass as a pct. of total biomass
	Millions of tons	Pct
As estimated w/o seedlings	20.8	4
5	21.9	4
25	27.8	5
50	41.7	8

The major focus in this study was forest lands at special risk for catastrophic wildfires because these are the areas where fuel treatments are most likely to occur. Using the maps developed by Hardy and others (2000) to classify FIA plots according to historic natural fire regime and condition class, we estimated the amount of biomass on these lands. Results show that of the estimated 1.08 billion tons of forest land biomass, approximately 57 percent (619 million tons) is found on lands we classify as high risk (Fire Regime I or II, and Condition Class 2 or 3); 23 percent is found in the highest risk category (Condition Class 3) (*fig. 3*).

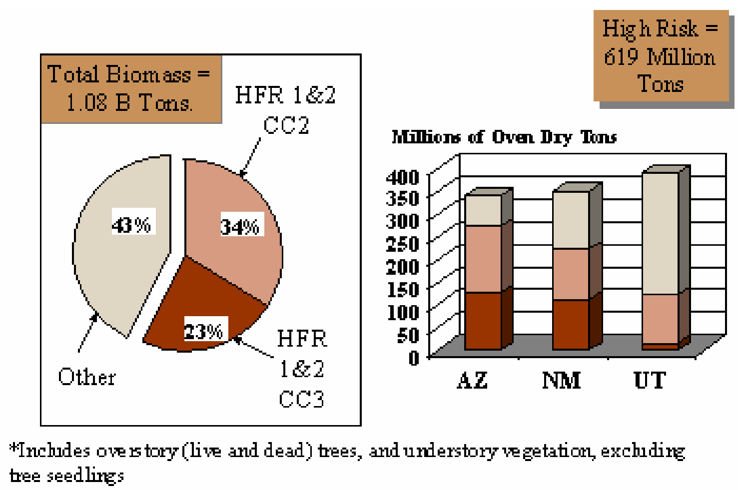

Figure 3—Estimated biomass of overstory and understory vegetation*on high fire risk forestland in Arizona, New Mexico, and Utah.

This is not surprising considering the amount of land falling in these categories (*fig. 4*). More than two-thirds of the forest land in Arizona and New Mexico falls into these high-risk categories. For Utah, the percentage is much smaller, with less than one-third of the forest land classified as high risk. In fact, the breakdown of biomass by state shows that Utah, although having the largest amount of forest land biomass, has only 30 percent of its biomass in these high-risk areas and only a very small percentage (less than 5 percent) in the highest risk category. Conversely, 81 percent of the forest land biomass in Arizona is found in these high-risk areas, with about an equal amount falling on Condition Class 2 and Condition Class 3 lands. In New Mexico, biomass is about evenly split between the two fire risk categories and all other lands.

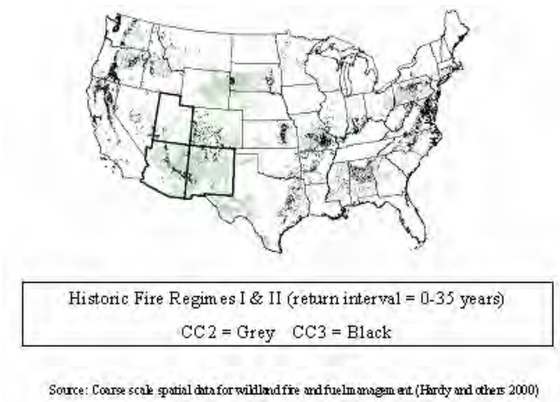

Figure 4—Historic Fire Regimes I and II; Condition Classes 2 and 3.

Next, we looked in more detail at biomass in these high-risk areas, focusing on overstory versus understory, ownership, and reserved status. Looking first at vegetation layers (*fig. 5*), the majority of the biomass found on these high-risk lands is in the overstory (96 percent) with 84 percent coming from live overstory trees and 12 percent from dead trees. Of the 4 percent of understory biomass, half is found in Layer 2 (1.6 to 6 ft). However, it is important to note that because this estimate does not include tree seedlings, the distribution of biomass among layers is affected. Yet, when we looked at the plot data, more plots reported tree seedlings in Layer 2 than the other two layers, so the relative ranking may be correct

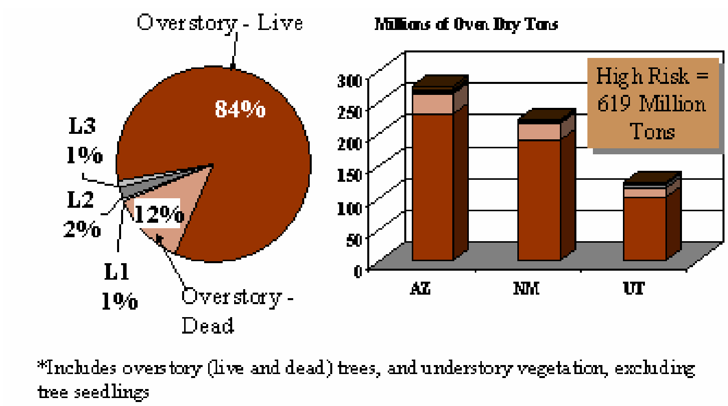

Figure 5—Estimated biomass of overstory and understory vegetation*on high fire risk forestland in Arizona, New Mexico, and Utah by vegetation layer.

Looking at the overstory separately (*fig. 6*), there are at least two points of note. First, 73 percent of the overstory biomass is found in trees (both live and dead) 10 in. or greater in diameter. Second, the relative percentage of dead to live trees does not change much with size. Twelve percent of the biomass of the larger trees comes from dead trees, while the percentage of dead tree biomass is 11 percent for the smaller trees. There is little difference in the distribution of the overstory in the three states.

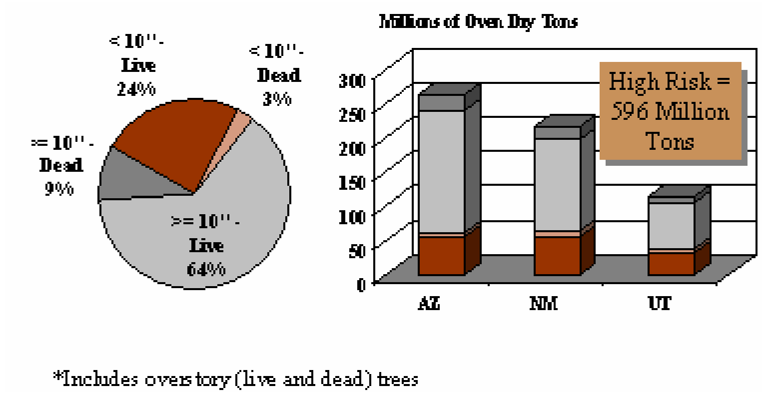

Figure 6—Estimated biomass of overstory vegetation* on high fire risk forestland in Arizona, New Mexico, and Utah, by size.

Turning to understory biomass, nearly 90 percent comes from shrubs, followed by grasses and then forbs (*fig. 7*). The breakdown by state shows, however, that the relative ranking of states has now changed. When looking at total biomass on these high-risk lands, or just overstory biomass (*fig. 5 and fig. 6*), the ranking from largest to smallest amount was Arizona, New Mexico, and Utah. When focusing on understory biomass only, Arizona still ranks first, but Utah overtakes New Mexico to rank second in terms of understory biomass on high-risk lands.

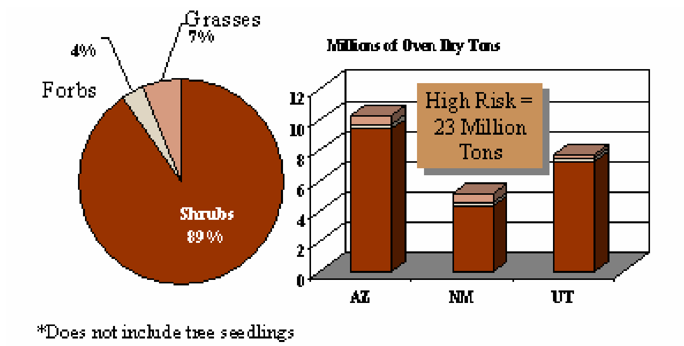

Figure 7—Estimated biomass of understory vegetation* on high fire risk forestland in Arizona, New Mexico, and Utah, by life form.

Aside from the physical characteristics of forest fuels on high-risk lands, other important dimensions in the management of these lands are ownership and reserved status. Nearly 60 percent of the biomass occurs on lands administered by the Forest Service, with another 28 percent being found on private land (*fig. 8*). A comparison of states shows that Arizona has a larger percentage of this material occurring on lands administered by Federal agencies other than the Forest Service or the Bureau of Land Management (mainly the National Park Service) than the other two states. In Utah, a larger percentage of biomass is found on lands administered by the Bureau of Land Management than in either Arizona or New Mexico. Most of the biomass in these three states (88 percent) occurs on lands currently available for timber utilization (non-reserved lands) (*fig. 9*).

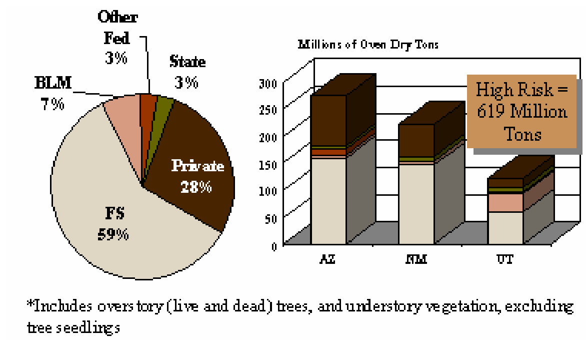

Figure 8—Estimated biomass of understory and overstory vegetation* on high fire risk forestland in Arizona, New Mexico, and Utah, by ownership.

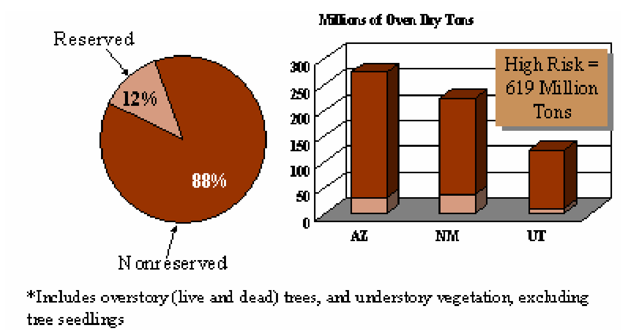

Figure 9—Estimated biomass of understory and overstory vegetation* on high fire risk forestland in Arizona, New Mexico, and Utah, by reserved status.

Conclusion

Broad scale assessments of forest fuels would be useful for fuel treatment and bio-based product planning. The general conclusion of this study, however, is that more research is needed to accurately estimate biomass from FIA understory vegetation data. While overstory biomass can be adequately estimated using FIA data, the understory biomass estimates presented here are only illustrative, giving a general impression of the amount of understory biomass in Arizona, New Mexico, and Utah. These understory estimates are not completely satisfactory. They are based on equations developed for other areas of the country, for different species, and do not include tree seedling biomass. Biomass of dead understory plants is also not available.

Given the fact that the understory appears to be a fairly small component of total forest biomass, FIA data can be used to provide reasonable estimates of amounts and location of forest fuels for broadscale assessments. However, to develop more complete estimates of biomass, including the understory, research is needed to develop equations for estimating the biomass of tree seedlings based on percent cover or the number of seedlings. Additionally, there is a lack of equations for estimating the biomass of shrubs based on percent cover; many of the available equations for shrub biomass use other parameters, such as diameter, which is not a measurement collected by FIA. Finally, and perhaps most importantly, research is needed to test the applicability of species and area-specific understory biomass equations to other species or areas and to develop more generally applicable understory biomass equations.

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The Application of FARSITE for Assessing a Mechanical Fuel Treatment in the Eastern Sierra Nevada[1]

Erin K. Noonan[2] and Paul T. Tueller[3]

Abstract

FARSITE fire area simulator was used to test the effectiveness of a small mechanical fuel treatment in a bitterbrush/sagebrush shrubland in the Eastern Sierra Nevada. Custom fuel models were created for input into FARSITE using a combination of data obtained from the Southwestern Fuels Photo Series and fuels data collected at the study site for treated and untreated areas. Fuels data were calibrated to expected fire behavior by running simulations in Behave Plus 4.4b. The resultant custom fuel model map combined with other GIS data necessary for FARSITE were used to simulate fire with moderate, high, and extreme weather parameters. Results showed a decrease of fire intensity, rate of spread, flame lengths, reaction intensity, and heat per unit area as the fire transitioned from an untreated to a treated fuel bed. These methods and results are important in displaying the potential benefits of fuel treatments adjacent to wildland urban areas and building support for small area fuel treatments.

Introduction

As the country faces another year of record acres burned in the United States, it is clear that high levels of fuels are contributing to increasing fire hazard (Skinner and Chang 1996, Graham and others 2004). Coupled with a westward demographic shift in the United States, the habitation of open space is increasing, creating a larger area of wildland urban interface. Evaluating fuel treatments in such areas is a beginning to tackling the nationwide problem of fuels and the resultant extreme fire behavior experienced during recent fire seasons.

FARSITE fire area simulator (Finney 1996) can be used to help quantify fire behavior after fuel treatments and thus is a useful tool in determining the effectiveness of these treatments. The FARSITE program simulates fires to assist land managers in predicting fire behavior. Using information on weather, wind, topography, fuels, and canopy characteristics, FARSITE displays an expanding fire perimeter at specific time intervals using a geographical information system or GIS (Finney 1996).

The limitations and assumptions of the FARSITE model are a result of the simplifications of complex fire processes necessary to decrease data requirements and excessive simulation times. The complexity of wind, weather, and fuel moisture inputs are reduced to facilitate use in the model. Hourly temporal variation for wind limits the ability to capture local wind events caused by topography (Andrews and

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others 2001). The resolution of raster based inputs are often determined by the 30 m digital elevation model (DEM), which assumes that conditions will remain homogenous for a minimum area of 30 m^2. As a result, fine scale variability that can change fire behavior outputs may not be captured at this resolution. Despite the limitations of FARSITE, it is still used frequently in fire operations, management, and research.

This experiment introduces a method for evaluating a mechanical fuel treatment in a sagebrush/bitterbrush shrubland. The objective was to detect a difference of fire behavior outputs between mechanically treated (fuel model 41) and untreated shrub fuels (fuel model 40) using FARSITE with various combinations of wind, weather, and fuel moisture. The simulations are evaluated by FARSITE outputs such as rate of spread (ROS), flame length (FML), fire line intensity (FLI), heat per unit area (HPA), and reaction intensity (RCI). The site is located next to a housing sub-division, approximately 28 ac in size, thus exemplifying a typical wildland urban interface area (*fig. 1*).

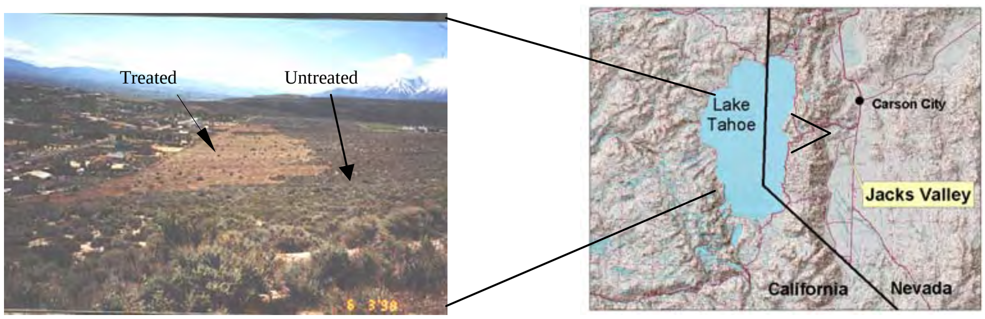

Figure 1–*Mechanical fuel treatment next to a housing sub-division.*

Methods

Simulation Area

The study site was located south of Carson City, Nevada in Jacks Valley. Elevation ranges from 4,800 to 5,000 ft (1,460 to 1,520 m). The average annual air temperature is 50°F (10ºC) while the average annual precipitation is 10 in. (25 cm) (Candland 1979). The study area is dominated by sagebrush shrublands (*Artemisia tridentata* Nutt) with antelope bitterbrush (*Purshia tridentata* (Pursh) DC and a sparse grass understory of squirreltail (*Elymus elymoides* (Raf.) Swezey) and cheat grass (*Bromus tectorum* L.).

Data acquisition

The fuels data representative of the 1:24,000 Genoa quadrangle were acquired from different sources. Initially, the fuel model map only represented the NFFL (Northern Forest Fire Laboratory) standard fuel models (Anderson 1982). These were classified as part of a fuel mapping project that encompassed 72 quadrangles in the eastern Sierra Nevada (Krauter 2001). The custom fuel models were created from fuels sampled during August of 2001 following methodology established by Brown and others (1982). This methodology was slightly altered to apply to shrubland areas. The fuels data were used to create custom fuel model 41 representing treated fuels. Fuel model 41 was composed of mainly 10 hr fuels that were masticated. Some regeneration occurred for the shrubs and herbs in this area; however, cheat grass, while present, did not compose the entire area. Due to the limitations of applying fuel sampling methodology designed for forested areas to shrublands, a photo series was used to represent the untreated fuel model 40. Data collected from field sampling and observations were used to guide the identification of the SWSB09 (SouthWest SageBrush) fuel model in the photo series (Ottmar and others). 1 hr fuels were increased to 0.75 t ac^{-1} to obtain acceptable spread rates and flame lengths for shrub fuels as predicted in BehavePlus (Andrews and others 2001). A series of simulations in BehavePlus were used to examine the effect of changes in 1 hr fuels, fuel depth, and surface area to volume on the outputs for the custom fuel models (*table 1*).

Table 1–*Standard and custom fuel models for the Jacks Valley simulations. Surface area to volume (1 ft^{-1}): 1hr (2002), Live woody and herbaceous (1500).*

Custom fuel model inputs	Custom fuels	
Fuel model number	40	41
1 hour fuels (t ac^{-1})	0.75	0.29
10 hour fuels (t ac^{-1})	0.09	1.97
10 hour fuels (t ac^{-1})	0.5	0.38
Live woody hour fuels (t ac^{-1})	2.23	0.23
Live herbaceous hour fuels (t ac^{-1})	0.13	0.09
Depth (ft)	2.2	0.8

Fuel moisture, wind, and weather values were obtained from the Fish Springs RAWS (Remotely Automated Weather Station) station, which was located 13.5 mi. south of the study site. Wind and weather were represented by average, 75th percentile and 95th percentile data. Fuel moisture was represented by low and average fuel moisture during the month of August for this area. Six separate simulations were done in FARSITE with different wind, weather, and fuel moisture variables to evaluate the effectiveness of the treated fuels (*table 2*).

Table 2–*Average output from the six simulations in Jacks Valley for mechanically treated (fuel model 41) and untreated shrub fuels (fuel model 40). Outputs: FLI=fire line intensity; RCI=reaction intensity; FML=flame length; HPA=heat per unit area; ROS=rate of spread.*

Simulation Number	Inputs	OUTPUTS *kW/m* FLI	*kW/m2* RCI	*m* FML	*kJ/m2* HPA	*m/min* ROS
1 treated	Weather 75th; average wind;	104.5	4074.7	0.4	4055.7	1.3
1 untreated	fuel moisture	230.2	4042.3	0.8	4629.4	2.5
2 treated	Weather 95th; average wind;	198.2	5491.4	0.7	5262.2	1.8
2 untreated	fuel moisture	526.3	6058.0	1.2	7290.5	4.1
3 treated	Weather 95th; average wind;	278.4	6236.1	0.8	5754.6	2.4
3 untreated	low fuel moisture	707.4	6467.5	1.4	8138.3	5.0
4 treated	Weather and wind 75th;	120.5	4542.4	0.5	4421.3	1.3
4 untreated	average fuel moisture	428.1	5182.2	1.0	6197.0	3.5
5 treated	Weather and wind 95th;	213.5	5423.6	0.7	5026.0	2.1
5 untreated	average fuel moisture	623.1	5658.3	1.2	6750.5	4.9
6 treated	Weather and wind 95th;	284.8	6140.9	1.3	5470.5	2.6
6 untreated	low fuel moisture	848.7	6652.6	1.5	7986.1	6.0

Results

The treated fuels resulted in lower average values for rate of spread, reaction intensity, heat per unit area, flame length, and fire intensity when compared to the untreated fuels. This was true for both average and low fuel moisture values; 75th and 95th percentile weather; and average, 75th and 95th percentile wind inputs. Furthermore, increasing the wind and weather percentile data and decreasing fuel moisture resulted in larger values for the fire behavior outputs (*table 2*).

Rate of spread values were 56 percent more for the untreated versus treated fuels. Similarly, fire line intensity values were 64 percent more for the untreated versus treated fuels. These two outputs displayed the largest difference on average between the treated and untreated fuels. Flame length was 38 percent more for the untreated versus treated fuels.

The six fire simulations displayed in *figure 2*, showed the results of fire behavior under the different wind, weather, and fuel moisture treatments. The light colored pixels show areas where the fire line intensity did not exceed 100 Btu/ft/s and flame length was less than 4 ft. At these levels, direct attack on a wildfire is feasible at the head or flanks (Andrews and Rothermel 1982; Pyne and others 1996). The dark colored pixels show areas that exceeded 100 Btu/ft/s and flame lengths greater than 4 ft. At these levels, wildfires are managed with equipment such as plows, dozers, pumps, and retardant aircraft.

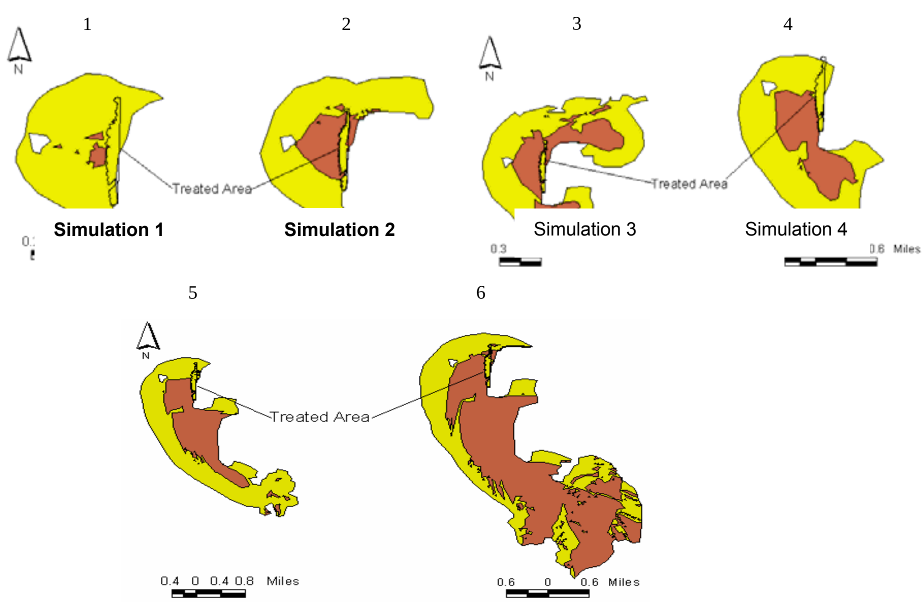

Figure 2–*Jacks Valley FARSITE simulations 1 to 6. Dark colored areas suggest indirect tactics (flame lengths greater than 4 ft and fire line intensity greater than 100 Btu/ft/s).*

Simulations1 to 3 represented a wind direction of 270 degrees, while simulations 4 to 6 represent a wind direction of 315 degrees. This difference accounts for the change in shape of the fire perimeters after the third simulation. Simulation six represents a larger area of more severe fire behavior while the first simulation is the least severe with a small area of approximately 12.9 ac. In all the simulations, the treated fuel area (fuel model 41) displayed fire line intensities below 100 Btu/ft/s and flame lengths less than 4 ft.

Discussion and Conclusion

The treated fuels were adjacent to a housing development on the east and were created to provide an area of defensible space against wildfire. The mechanical treatment occurred in June of 1998, allowing 3 growing seasons for the vegetation to recover from the treatment. Consequently, fuel model 41 represented a mechanically treated area 3 yr after the treatment, because these fuels were sampled and quantified in 2001.

The untreated area is a wildlife preserve. A cluster of antelope bitterbrush extends into the southern section of the treated fuels. This was done as a compromise with the homeowners in the area who suggested islands of vegetation should be included within the treated fuels for deer. However, simulations one and four are the only simulations that did not display dangerous fire behavior in this untreated section (*fig. 2*). These simulations had the least severe inputs and outputs compared to all the simulations. Patches of vegetation that exist within treated fuel breaks must be carefully evaluated and maintained. Otherwise, their existence may defeat the purpose of a fuel break as they vector fire to adjacent houses.

Generally, the treated fuels were successful at creating a buffer against dangerous fire behavior. The mechanically treated area resulted in less fuel 3 yr after the initial treatment because the vegetation was killed and had decayed. However, field sampling at this site occurred during Fall when cheatgrass (*Bromus tectorum)* was not abundant, which would result in a greater rate of spread for this treated area. Maintaining this fuel break would be one way to inhibit the increase of cheatgrass, thereby decreasing potentially higher rates of spread.

Low fuel loading for the treated area was reflected in the fuel model by decreasing the tons per acre for the 1 to 100 hr fuels, live herbaceous, and live woody fuels. The decrease of 1 hr fuels from 0.75 t ac^{-1} (untreated fuel model 40) to 0.29 t ac^{-1} (treated fuel model 41) significantly decreased the rate of spread and flame lengths. This occurred because there were not enough fine fuels to carry the fire (Burgan and Rothermel 1984). Low values for 1 hr fuels inhibit fire spread in FARSITE. Consequently, the lower fire behavior values for the treated area were not surprising.

Other factors that should be taken into consideration when explaining the results of these simulations are the limitations and assumptions of the model itself. These fuel models were created as objectively as possible; however, FARSITE intends the user to create fuel models that approximate observed fire behavior for the area at question. Realistically, fuel model 40 (untreated shrub fuel model) should have displayed flame lengths of 40 ft (Pers. comm., Forest Service fuels manager). However, even with the most extreme inputs represented in simulation 6, flame lengths did not exceed 19.9 ft. This was probably due to the low values of 1 hr fuels and high values for live woody fuels represented in fuel model 41, which act as a sink in the fire spread equation (Rothermel 1972).

The Jacks Valley treated zone was treated again in September 2001. The section of untreated fuels that extended into the treated zone was removed as well. With consistent management and policies coupled with a scientific, field-based monitoring program to evaluate the effectiveness of fuel treatments, these buffers can help to remedy the problems of urban interface, especially in the areas where houses reside with shrublands. FARSITE is a tool that can be used to display the effects of decreasing dangerous fire behavior in these fuel treatment areas, thus gaining public support and understanding. This allows for the continued expansion of the fuel treatment programs much needed in the wildland urban interface.

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Modeling Transitions in Shrubland Fire Behavior Using Crown Fire Modeling Techniques[1]

Joe H. Scott[2]

Abstract

Transitions from one type of fire behavior to another cause difficulty in modeling fire behavior in some shrubland fuelbeds. Under different environmental conditions, some shrub fuel complexes may burn benignly in only surface litter, grass, or low shrubs, or more intensely in the taller main shrub canopy as well. With current surface fire modeling methods, simulating fire behavior in these fuels requires *ex ante* knowledge of which fuels will carry the fire so that the appropriate fire behavior fuel model can be selected. A conifer crown fire simulation technique has been adapted to simulate the transition between surface and shrub-canopy fire. Simulations are based on the Rothermel surface spread model; however, the technique can be adapted to other spread models. Two surface fire behavior fuel models are specified—a "low" fuel model representing the grass, low shrub or litter fuels that support relatively benign fire behavior when the fire remains beneath the main canopy, and a "high" fuel model that represents fire behavior in the main shrub canopy. The method also requires an experience-based estimate of the flame length at which the transition to shrub-canopy fire begins. Standard or custom fire behavior fuel models may be used. Interpretation is similar to that for conifer crown fires.

Introduction

Selecting the best standard surface fire behavior fuel model (FBFM; Albini 1976, Anderson 1982) for some fuel complexes can be difficult because one fuel model often does not fit all environmental conditions. For example, Keane and others (1998) made two FBFM spatial data layers for use in FARSITE (Finney 1998). One layer specifies the most appropriate FBFMs for moderate environmental conditions, and another layer specifies a FBFM to use for extreme conditions (e.g., severe drought). There are no objective guidelines for deciding which FBFM data layer to use for a given set of environmental conditions; the user heuristically decides which layer to use based on quality of the fire behavior simulations relative to observation or expectation. In such a simulation system, ex ante knowledge of expected fire behavior is needed in order to predict exactly that–expected fire behavior.

These problems are most prevalent in shrub fuel complexes. In some cases, shrub and shrub-like fuelbeds exhibit transitional behavior, gradual or abrupt changes in fire behavior as the nature of the fire-carrying fuel stratum changes from grass and litter to the main shrub canopy, similar to that of conifer crown fires (Scott and Reinhardt 2001). For example, the Storm King fire spread beneath tall oakbrush during a period of moderate environmental conditions, then later crowned through the oakbrush canopy under the influence of a strong wind (Butler and others 1998).

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[1]An earlier version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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In the simulation technique presented here, shrubland fuel complexes may include short coniferous vegetation such as pinyon-juniper or conifer plantation. A shrubland fuel complex may exhibit two very different behaviors in response to a changing fire environment (fuel moisture and wind speed), especially in deep fuel complexes (with resulting separation of the shrub canopy and surface litter). One behavior corresponds to spread primarily through the grass and litter beneath the shrub canopy, the other to spread through the canopy itself. Another example is in patchy shrub or shrub-like fuel complex (e.g., pinyon-juniper) that has a horizontal mixture of two very different surface fuels. Under some conditions fire spreads through only the litter or grass layer, while under other conditions fire may spread through the whole fuel complex, resulting in higher spread rate and fireline intensity. Anecdotal observation of fire behavior in those fuel complexes suggests that fuel condition (fuel moisture), slope, and wind speed all play a role in determining whether the main canopy will support fire spread.

An automated method to scale fire behavior predictions between two FBFM selections might improve fire behavior simulations when knowledge of which fuel stratum is carrying the fire is not available. Such a method should be sensitive to site and environmental conditions (slope steepness, wind reduction to mid-flame level, fuel moisture, and windspeed), as well as fuel characteristics (such as FBFM and available shrub canopy bulk density). This paper describes a “stacked” fuel model concept for use with the Rothermel surface fire spread model.

Modifying NEXUS

The NEXUS crown fire hazard assessment system (Scott 1999) is a Microsoft® Excel spreadsheet that simulates the transition between surface fire and crown fire behavior, based on Van Wagner’s (1977) transition criteria. The Southern Utah Fuels Management Demonstration Project identified the need for a similar transition model for shrub-canopy fires as well as conifer crown fires, especially in pinyon-juniper (P-J) fuels. NEXUS has been modified to accommodate shrub-canopy fire simulations and converted to a stand-alone program, called NEXUS 2.

Crown fire simulation

The original NEXUS conifer crown fire simulation method is described in detail in Scott and Reinhardt (2001). Predicted surface fireline intensity is compared with the critical fireline intensity needed for transition to crown fire using Van Wagner’s (1977) transition criterion. If the threshold is not met, then the model predicts surface fire and the final spread rate and intensity are those of the surface fire. If the initiation criterion is met, then Van Wagner’s second criterion, mass-flow rate, is checked to see if it meets the minimum required for sustained active crown fire spread. Mass-flow rate is the product of crown fire spread rate (predicted in NEXUS using Rothermel’s (1991) correlation) and canopy bulk density. If this second criterion is met as well, the model predicts active crowning and final spread rate and intensity are those of the crown fire; if not, the model predicts passive crowning.

The simulations use information about the fire environment to determine which type of fire is likely to occur: surface fire, passive crown fire, or active crown fire. Spread rate and intensity in a passive crown fire are scaled between separate predictions for surface and active crown fire.

Shrub-canopy fire simulation

NEXUS 2 has been modified to simulate shrub-canopy fires as well as conifer crown fires. No predictive model of transition from surface to shrub-canopy fire is available, so the user must input the flame length at which the transition from surface to shrub-canopy fire is expected to take place. Using Byram's (1959) flame length model, NEXUS 2 converts this surface fire flame length to fireline intensity, which is then used in place of Van Wagner's initiation criterion (minimum required fireline intensity). For plotting on a shrub-canopy fire hazard assessment chart (*fig. 1*), fireline intensity is divided by heat per unit area to estimate its equivalent in terms of spread rate. Using Rothermel's model, heat per unit area is a function of FBFM and fuel moisture, but not slope or wind speed.

Van Wagner's mass-flow rate criterion for conifer crown fires is used in modeling shrub-canopy fires as well. The available shrub canopy bulk density must be provided to NEXUS 2. This is a rarely measured variable in the fuel types to which it may be applied, such as pinyon-juniper or oakbrush. Until research provides reliable values for available shrub canopy bulk density, this method must be applied with caution.

The shrub-canopy fire model uses information about the fire environment to determine if fire behavior will be determined by the low or the high fuel model, or if behavior will fall in the transition zone in between.

Shrub-canopy transition indices

In modeling the hazard of conifer crown fires, NEXUS computes two indices of crown fire potential: Torching Index (TI) and Crowning Index (CI). The TI is the 20-ft windspeed at which some kind of crowning is possible; it is the point at which the predicted surface fireline intensity equals the critical value needed for crown fire initiation. The CI is the 20-ft wind speed at which active crown fire is possible; it is the point at which the potential crown fire spread rate equals the critical value needed to produce the minimum mass-flow rate for maintaining solid flame.

The indices have similar meaning and interpretation when computed for shrub-canopy fires. The TI is the 20-ft wind speed at which flame length predicted for the low FBFM equals the transition flame length value entered by the user. The CI is the 20-ft wind speed at which the spread rate predicted for the high FBFM equals that needed to produce the minimum mass-flow rate for maintaining solid flame, a function of the available shrub canopy bulk density.

The CI can be lower than the TI in many circumstances. In those cases, just as for conifer crown fires, there is no transition region–the fire goes from the low FBFM to the high FBFM with an abrupt jump rather than a smooth transition.

Estimating transitional fire behavior

For the cases where TI<CI, final fire behavior for the wind speed region between TI and CI is calculated by scaling between the predictions for the low and high FBFMs. The scaling is based on a transition function (TF) similar to Van Wagner's (1977) crown fraction burned, which produces a value between 0 and 1.

$$\text{SPRTfinal} = \text{SPRTlow} + \text{TF}\ (\text{SPRThigh} - \text{SPRTlow})$$

When TF is 0, the formula evaluates to SPRTlow. When TF is 1, the formula evaluates to SPRThigh. For TF=0.5, the formula estimates SPRTfinal will be half-way between SPRTlow and SPRThigh.

Details of the TF calculation are provided in Scott and Reinhardt (2001 [Appendix A, equation 28]). Briefly, TF is the fractional amount by which the difference between SPRTlow (at the given wind speed) and the transition spread rate exceeds the difference between SPRTlow (at CI) and the transition spread rate, with bounds between 0 and 1.

Example Simulation

Inputs

The following stylized example illustrates the application of this modeling concept for a pinyon-juniper fuel complex with a significant component of short grass in the areas between the pinyon and juniper plants. When the fire environment is moderate, only the grass burns, and the pinyon-juniper canopy is not involved in fire spread or intensity. The FBFM that best describes fire spread in the grass is model 1 (short grass). However, local knowledge suggests that spread rate will be approximately 75 percent of that predicted by FBFM 1, so we will use a multiplier of 0.75 in NEXUS. Wind reduction factor is 0.2 for these fuels, because the P-J "overstory" blocks the 20- ft wind from reaching the grass fuels.

For this example, available shrub canopy bulk density is set to 0.10 kg m^{-3} (0.0062 lbs ft^{-3}), and the transition flame to 2 feet.

Table 1—*Inputs for shrub-canopy fire simulation in NEXUS.*

	Low FBFM[1]	**High FBFM[1]**
Fire behavior fuel model (FBFM[1])	1 (short grass)	4 (chaparral)
Spread rate multiplier[2]	0.75	1.00
Fuel load and depth multiplier[3]	1.00	0.75
Wind reduction factor[4]	0.2	0.4

[1]See Anderson (1982) for a complete description.
[2]Affects rate of spread and fireline intensity linearly
[3]Affects rate of spread linearly, fireline intensity with square of multiplier
[4]Ratio of mid-flame to 20-ft windspeed

When the fire environment is conducive (stronger winds, steeper slopes, drier fuels), the P-J canopy dominates fire spread and intensity. Local experts indicate that spread rate and fireline intensity in the P-J canopy are best modeled with FBFM 4, but only after reducing the load and depth to 75 percent of the original value. (This has the effect of reducing spread rate to 75 percent and fireline intensity to 56 percent [$.75^2$] of the value predicted by the unadjusted FBFM.) The adjusted spread rate is used to determine if the critical mass-flow rate is achieved. Wind reduction factor is 0.4 for the P-J fuels, because they will produce taller flames and have no blocking overstory. Fire environment inputs represent moderately dry conditions (*table 2*).

Table 2—*Fire environment inputs for the stylized example.*

	Variable
1-hr timelag moisture content (pct)	5
10-hr timelag moisture content (pct)	6
100-hr timelag moisture content (pct)	7
Live woody moisture content (pct)	85
Slope (pct)	0
20-ft wind speed (mph)	0-40

Outputs

The simulation outputs are best displayed on a shrub-canopy fire hazard assessment chart (*fig. 1*). Plotted over a range of 20-ft wind speeds, the low FBFM spread rate predictions cross the transition spread rate (computed from the transition flame length of 2 ft) at a 20-ft wind speed of 11 mph; that is the Torching Index, where the transition toward a full shrub-canopy fire begins. The high FBFM predictions cross the critical spread rate for full shrub-canopy fire at 16 mph; that is the Crowning Index, where the transition to shrub-canopy fire is complete. The region below the TI indicates fire spread in the low fuel model, using the wind reduction factor and multipliers specified for that model. The region above the CI represents fire spread in the high fuel model, using its wind reduction factors and multipliers. The region between the TI and CI represents transitional fire behavior, analogous to passive crowning in conifer crown fires. Spread rate and intensity rise through the transition zone as more of the shrub canopy becomes involved.

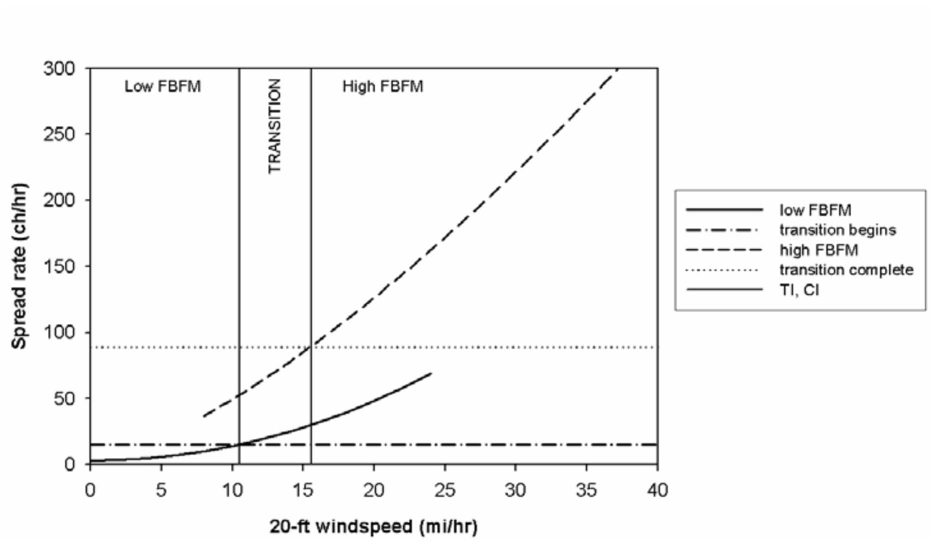

Figure 1—Shrub-canopy fire hazard assessment chart. Lines for low FBFM and transition-begins spread rate cross at the Torching Index, about 11 mi hr^{-1}. The high FBFM spread rate and transition-complete spread rate cross at the Crowning Index, about 16 mi hr^{-1}. Final spread rate follows the low FBFM up to the TI, then bridges the gap between models in the transition region, and finally follows the high FBFM above the CI (*fig. 2*). The steepness of the transition from the low to high FBFM depends on how close are TI and CI; if TI is greater than CI, the change is instantaneous.

Final flame length exhibits similar behavior, but the flame-length gap is larger than the spread-rate gap, so the impact of transition is more severe. Above the CI, final flame length exceeds high FBFM flame length by a small amount; this is because the final flame length is computed from the combined low and high FBFM heat per unit area values rather than just the high.

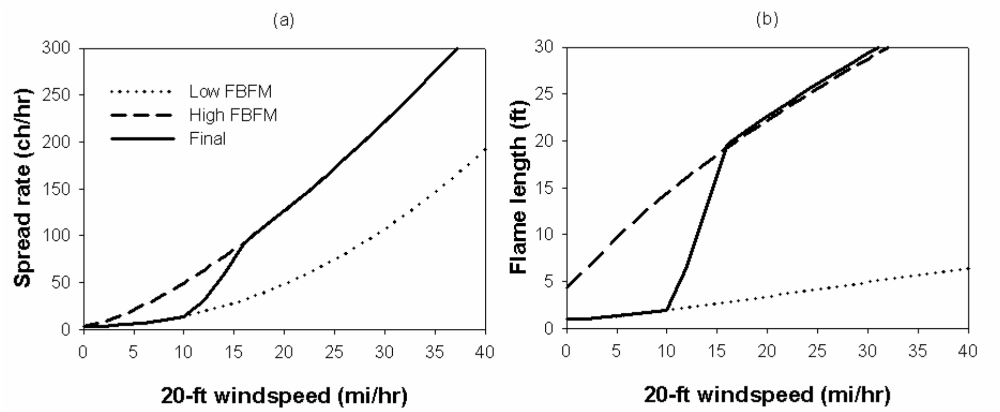

Figure 2—Simulation results for spread rate (a) and flame length (b).

Discussion

The modeling process for simulating shrub-canopy fires in NEXUS 2 appears similar to the two-fuel-model concept of BEHAVE (Andrews 1986) and BehavePlus2 (www.fire.org), but there are important distinctions. The two-fuel-model concept simulates spread rate through a horizontally arranged matrix of two fuel models; overall spread rate is a function of the relative coverage of each fuel model and their respective spread rates. In BEHAVE for DOS there was only one calculation method: area-weighting. In BehavePlus2 there are three methods to choose from: area-weighting, harmonic mean (Fujioka 1985, Martin 1988), and two-dimensional expected spread (Finney 2003). By contrast, NEXUS 2 uses two fuel models in a vertical, or “stacked”, arrangement. The low and high fuel models apply to the same area. The calculation predicts whether predictions for the low or the high fuel model will prevail, for the given set of environmental conditions, and scales the output between them if necessary.

Not all shrubland fuelbeds are appropriate for this simulation method. Shrub fuel complexes in which fire is carried by only one type of fuel (e.g., uniform, low sagebrush) can be adequately modeled with current methods using only one FBFM. The stacked fuel model simulation technique is suited for shrub (and shrub-like) fuel complexes, typically modeled with Rothermel’s surface fire spread model, that exhibit a such a degree of fuel separation that two distinct behavior responses may occur. For example, pinyon-juniper and tall oakbrush or chaparral fuel complexes are appropriate for consideration.

Sufficient fire behavior observation data do not exist to validate (or invalidate) this modeling approach. The availability of this modeling method may encourage the collection of data to test its validity.

The stacked fuel model concept can be used with standard or custom surface fire behavior fuel models. The only limitation is that the simulations must predict higher spread rate for the high FBFM than the low under all wind and fuel moisture conditions. A new set of standard FBFMs has been developed[3]. The stacked fuel model concept will work with those models as well.

While this paper describes fuel model stacking to model transitions in shrubland fire behavior, the concept is equally applicable to other fuel types. For example, in forested fuel complexes the surface litter often carries the fire under moderate environmental conditions while under more extreme conditions the fire may be carried by combined litter and shrub fuels. This can be simulated by specifying a low FBFM for the moderate conditions (e.g., FBFM 8, compact timber litter), a high FBFM for the more extreme conditions (e.g., FBFM 10, timber litter and understory), and the flame length at which the transition begins.

The stand-alone version of NEXUS capable of making these stacked shrub-canopy fire simulations will be available for beta testing in fall 2004. Visit www.fire.org/nexus/nexus.html for more information and to download the program when available.

Conclusions

Transitions in fire behavior similar to crown fires occur in many shrub fuel complexes. Standard fire behavior fuel models and modeling systems do not simulate such transitions; custom models can be built to approximate such abrupt changes, but usually with poor results.

NEXUS 2 has been modified to simulate shrubland fire behavior as a transition between two selected fire behavior fuel models.

Interpretation of the transition is similar to that for conifer crown fires. The Torching Index is the 20-ft windspeed at which transition from the low to the high FBFM begins, given the site and fire environment variables. The Crowning Index is the 20-ft wind speed at which the transition to the high FBFM is complete. The relative steepness of the transition from the low to high FBFM depends on the separation of TI and CI. When CI is less than TI the transition is instantaneous, and a hysteresis exists (Scott and Reinhardt 2001).

This method relies heavily on user-provided inputs that are not well know or easy to estimate. To be useful, we need data on canopy bulk density of fuel complexes like pinyon-juniper, a model of transition for shrub fuel complexes, and validation data.

Acknowledgments

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Assessing the Effectiveness of Landscape Fuel Treatments on Fire Growth and Behavior in Southern Utah[1]

Rick D. Stratton[2]

Abstract

This paper presents a methodology for assessing the effectiveness of landscape fuel treatments on fire growth and behavior. Treatment areas were selected by fire managers from the Bureau of Land Management (BLM) based on the threat of fire to communities and the need for range and wildlife improvement. A fire density grid was derived from the BLM's fire start layer to identify historically high ignition areas. Fire Family Plus was used to summarize and analyze historical weather and calculate seasonal severity and percentile reports. Information from Fire Family was used in FARSITE and FlamMap to model pre- and post-treatment effects on fire growth, spotting, fire line intensity, surface flame length, and the occurrence of crown fire. This procedure provides managers with a quantitative measure of treatment effectiveness as well as spatial output that can be used for analyzing fuel treatment effectiveness, burn plan development, NEPA documentation, public education, and etcetera.

Introduction

Fuel modifications are receiving renewed interest as protection strategies, particularly in wildland-urban areas (Agee and others 2000). This is a result of costly fire seasons like 2000 and 2002, new national directives with increased funding (USDA Forest Service and USDI 2000), recognition of a change in fuel composition, structure, and loading, and fire manager's desire, yet limited ability, to control large fires. The primary purpose of a fuel treatment is to change the behavior of a fire entering a fuel-altered zone, thus lessening the impact of that fire to an area of concern. This is best achieved by fragmenting the fuel complex and repeatedly disrupting or locally blocking fire growth, thus increasing the likelihood that suppression will be effective or weather conditions will change (Finney 2000).

Recent research suggests that landscape-scale fuel modifications, such as prescribed fire, are the most effective way to modify the behavior and growth of large fires (Finney 2001). However, the effectiveness of fuel treatments remain a subject of debate due in part to the weather conditions they will or will not perform under, treatment method, completeness of the application, treatment design (i.e., placement, pattern, size), and the difficultly in evaluating the effectiveness of the proposed treatment. Simulation modeling allows the user to partially address these issues under various weather and fuel scenarios and provides a "tested" outcome for field application. This paper presents a methodology for assessing the effectiveness of landscape fuel treatments on fire growth and behavior by utilizing previous fire locations, historical weather, and fire growth and behavior models.

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[1]An earlier version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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Analysis Area

Ash Creek is located approximately 20 miles south of Cedar City, Utah and is adjacent to the communities of New Harmony and Harmony Heights. The project area (approximately 2,000 ac; 5,300 ft) is on Bureau of Land Management (BLM) administered land and bounded tightly by private ownership to the north, Interstate 15 (I-15) to the east, the Dixie National Forest and Pine Valley Mountain Wilderness Area to the west, and BLM, State, and private in holdings to the south. The area has seen an increase in urban development due to its rural setting and views of the Kolob Fingers (Zion National Park), inviting climate, and close proximity to various recreational sites and metropolitan areas.

Located on a relatively flat, east, southeast bench, understory vegetation is primarily crested wheatgrass (*Agropyron cristatum*), bluebunch wheatgrass (*Elymus spicatus*), junegrass (*Koeleria macrantha*) (1 to 2 ft), sage brush (*Artemisia tridentata*) (1 to 3 ft), oak (*Quercus turbinella*; *Quercus gambelii*) in some draws (4 to 15 ft), and smaller amounts of Utah serviceberry (*Amelanchier alnifolia*), bitterbrush (*Purshia tridentata*), and true mountain mahogany (*Cercocarpus montanus*) (2 to 15 ft). Utah juniper (*Juniperus osteosperma*) and scattered pinyon pine (*Pinus edulis*), in varying density, is the dominate overstory species (10 to 35 ft).

Summer cold fronts contribute to strong winds that are channeled through the I-15-Black Ridge corridor and into the project area. The effect of these winds on fire shape is evidenced in the Ash Creek Fire of 1996 (approximately 500 ac). The area has a history of fires attributed to recreational use, I-15 through traffic, and lightning on Black Ridge (6,400 ft) to the east and Pine Valley Mountains to the west (10,000 ft).

The objectives of the project are to reduce fire intensity, occurrence of crown fire, and mid- to long-range spotting and to increase native plant diversity and enhance wildlife forage. This was accomplished through herbicide application and fuel reduction. Treatment boundaries were delineated by ownership, previously chained areas (1960's), and wildlife requirements and is reflected in an asymmetrical, amoeboid design. Sage-dominated areas were applied with several applications of an herbicide (Tebuthiuron or "Spike"). Encroaching juniper was manually cut (lop-and-scatter) and is being followed up with pile and broadcast burning.

Methods

Specific information about the project area, such as objectives of the proposed treatment (e.g., wildfire control, wildlife enhancement), type of treatment (e.g., prescribed fire, manual thinning), pre- and post-treatment condition of the entire fuel complex, and supporting Geographic Information System (GIS) data were obtained from the BLM. A thirty-three year fire ignition layer for the BLM and Forest Service was used to derive a fire density GRID, using ArcView/Spatial Analyst (ESRI 2000) [When local fire data is unavailable, this information can be retrieved for Forest Service units from the National Interagency Fire Management Integrated Database (NIFMID) (USDA Forest Service 1993), using the Kansas City Fire Access Software (KCFAST) (USDA Forest Service 1996), fire occurrence information retrieval site; fire information for the Department of Interior is available on a yearly basis via a CD-ROM (USDI Park Service 2001)]. The locations of the nearest Remote Automated Weather Stations (RAWS) were identified and reporting history and site characteristics were analyzed to determine the most adequate station for the project area. Due to the channeling effect of the winds through the project area, one station was used to obtain wind speed and direction (White Reef; 16 yr history), and another was used to for the weather (Enterprise; 29 yr). Historical weather information was downloaded from NIFMID/KCFAST, fire occurrence information retrieval site and imported into Fire Family Plus (Bradshaw and McCormick 2000).

Fire Family Plus

Fire Family Plus is a fire climatology and occurrence program that combines and replaces the PCFIRDAT (Cohen and others 1994; Main and others 1990), PCSEASON (Cohen and others 1994; Main and others 1990), FIRES (Andrews and Bradshaw 1997), and CLIMATOLOGY (Bradshaw and Fischer 1984) programs into a single package with a graphical user interface. It allows the user to summarize and analyzing weather observations and compute fire danger indexes based on the National Fire Danger Rating System (NFDRS) (Bradshaw and others 1983; Burgan 1988).

Fuel moistures (i.e., 1-, 10-, 100-hr, live herbaceous, live woody) were obtained from a Fire Family Percentile *Weather Report*. Calculated fuel moistures were compared with local field sampling to validate and adjust the values. Wind speed, temperature, and relative humidity were obtained from a S*easonal Severity Report*; wind direction was obtained from a *Wind Speed vs. Direction Report*. Wind speeds were modified to account for persistent gusts (NOAA 2003) and directions were developed based on actual hourly RAWS data that adequately represented the appropriate percentile weather.

All weather and fuel moisture information was recorded at the 75th (moderate), 85th (high), and 95th (very high) percentile (*table 1*). In other words, weather occurring during the reporting period (June 1-September 30) 25 percent of the time is represented by the 75th percentile, 15 percent of the time for the 85th percentile, and so forth. All climatological and fuel variables were then used to develop the required weather and wind files/inputs for FARSITE and FlamMap.

Table 1—*Weather and fuel moisture information for the 75th, 85th, and 95th percentile as reported by Fire Family Plus and modified as noted.*

	Hour pct			**Live pct**		**Temp.°F**		**RH pct**		**Wind**	
	1	10	100	Herb.	Wood.[1]	Min.	Max.	Min.	Max	mph[2]	Direction[3]
75	4	6	9	90	110	56	87	16	47	17	190-235
85	4	5	7	80	100	59	89	14	40	19	190-235
95	3	5	6	60	90	64	92	10	28	23	190-235

[1]Adjusted from the Seasonal Severity Report based on local field sampling.
[2]Adjusted from the Seasonal Severity Report to account for wind gusts (NOAA 2003).
[3]During the burn period (1100 to 1900 hr).

FARSITE (Fire Area Simulator)

FARSITE (Fire Area Simulator) is a two-dimensional deterministic model for spatially and temporally simulating the spread and behavior of fires under conditions of heterogeneous terrain (i.e., elevation, slope, and aspect), fuels, and weather (Finney 1998). To do this, FARSITE incorporates existing fire behavior models of surface fire spread (Roth 1972; Albini 1976), crown fire spread (Van Wagner 1977, 1993), spotting (Albini 1979), point-source fire acceleration (Forestry Canada Fire Danger Group 1992), and fuel moisture (Nelson 2000) with GIS data. Simulation output is in tabular, vector, and raster formats.

FlamMap

FlamMap (Finney, in preparation) is a spatial fire behavior mapping and analysis program, which requires a FARSITE landscape file (*.LCP), as well as terrain, fuel, and weather data. However, unlike FARSITE, FlamMap assumes that *every* pixel on the raster landscape burns and makes fire behavior calculations (e.g., fire line intensity, flame length) for each location (i.e., cell), *independent* of one another. That is, there is no predictor of fire movement across the landscape and weather and wind information can be held constant. By so doing, FlamMap output lends itself well to landscape comparisons (e.g., pre- and post-treatment effectiveness) and for identifying hazardous fuel and topographic combinations, thus aiding in prioritization and assessments.

Vegetation and Fuel Models

Spatial vegetation data for the project area was extracted from a larger 15 million acre study area (Long and others, in preparation). A supervised classification of LANDSAT Thematic Mapper data—path 33 and rows 37 and 38—was used with ERDAS IMAGINE software (ERDAS 1999), incorporating polygons created by the IPW image processing program (Frew 1990). A maximum likelihood algorithm in ERDAS was used to classify the imagery based on a statistical representation of spectral signatures for each vegetation class created from field sampling. Ancillary layers, including land use and land cover, were used in combination with the classified imagery to assign polygons to one of 65 final vegetation classes.

The vegetation classes were cross-walked to 44 fuel models (including barren and water), 35 of which were "customized" models (i.e., the standardized model parameters (Anderson 1982) were altered to reflect a condition not adequately represented by the fire behavior models) and two, were custom models (i.e., 14: sparse grass-forb; 35: sparse shrub). Canopy cover, stand height, crown-base height, and crown bulk density were developed based on field data, anecdotal observations, and previously published work. Moderate and severe custom fuel files (*.FMD) were built to reflect the differences in fire behavior between moderate and high/severe conditions.

Terrain, fuel model, and canopy information was used to construct two modeling landscapes: pre-treatment and post-treatment. Sage-dominated areas were assigned either a fire behavior model (2, 6) or a customized model (e.g., 2-, 5+, 6-, etc; where the "-" or "+" represents a 20 percent change in the loading and depth). To simulate the effect of the Tebuthiuron, treated areas were reassigned a fuel model representing a 10-30 percent reduction in the shrub component. In some areas an adjustment factor (*.ADJ) was used to change the rate-of-spread without affecting other fire behavior outputs.

Pre-treatment stands of pinyon-juniper were assigned a standardized fuel model (4, 6) or a customized model (4-, 6-, 6--), *each with varying canopy characteristics*. Lop-and-scattered pinyon-juniper that was later pile and/or broadcast burned was reassigned a fire behavior fuel model (2, 5, 6, 11, 12), a customized model (4-, 5+, 6-, 6--), or a custom model (i.e., 14, 35); in general stand height, canopy cover, crown bulk density, and crown base height was eliminated, or reduced substantially.

Calibration

To produce fire growth and behavior output consistent with observations, model checking, modifications, and comparisons are done (i.e., calibration) with known fire perimeters and weather conditions (Finney 2000). Two fires were used to calibrate the model output, the Sanford Fire (April-June 2002; 78,000 ac) and the Langston Fire Use (August 2001; 600 ac). The Sanford Fire (Panguitch, UT) was useful in modeling low to extreme climatic conditions, with substantial elevation, topographic, and vegetative variation. Most fuel models were represented in the fire area, and canopy characteristics and their influence on crown fire transitions, spotting, and spread was analyzed. The Langston Fire Use (Zion National Park, UT) allowed testing of flanking and backing surface rates-of-spread in moderate weather conditions, on relatively flat terrain, and in fuel models 5, 8, 9, and 10.

Modeling Fire Growth and Spotting; FARSITE

To model fire growth and spotting potential, a single-source ignition in FARSITE was started in a historically high ignition area, as identified by the fire density grid. I-15 was imported as a barrier to surface spread, but was not impermeable to spotting. All fire simulations were modeled *without* suppression. One-day simulations, with a burn period of 1100 to 1900 hr, were run representing the 75th, 85th, and 95th, percentile weather and fuel conditions. The simulation process was repeated multiple times—with the same ignition point, as well as in other high ignition areas—to sample the variation in predicted fire size, shape, common spread pathways, spotting frequency, distance, and etc. Based on multiple runs, two of the “most representative” simulations were selected for each percentile level (six in all).

Calculating Fire line Intensity, Flame Length, and Crown Fire Activity: FlamMap

To calculate pre- and post-treatment fire line intensity, surface flame length, and crown fire activity, FARSITE terrain, fuel, and weather information was imported into FlamMap. Weather and fuel moisture conditions representing the 75th, 85th, and 95th percentile were used to generate the fire behavior data (18 output grids).

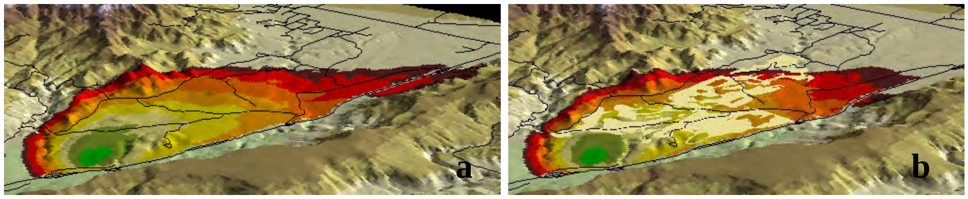
Results

Figure 2 are pre- and post-treatment FARSITE simulations for the 85th percentile draped over a 3-D landscape. Each color represents a 1 hr time-step or progression of the fire. *Table 2* summarizes fire size and spotting for each of the three percentiles, pre- and post-treatment.

Figure 3 displays FlamMap *area maps* of the 85th percentile pre- and post-treatment for flame length, fire line intensity, and crown fire activity. Tabular data for these fire behavior outputs are displayed in *table 2.*

Fire Size and Perimeter Growth

A modest reduction in fire size is apparent for each of the percentile weather and fuel conditions. The 85th showed the greatest percent change from the untreated condition (approximately 18 percent), which is likely do to the removal of most of the pinyon-juniper (i.e., fuel model “4s”/“6s”), thus reducing the spotting distance and the number of embers lofted. The 75th percentile simulation shows little change due to the similar surface rates-of-spread in dense, yet sparse pinyon-juniper stands and in recently burned/residual slash areas. As the weather conditions got more severe (95th percentile) and the fire size increased, the effectiveness of the treatments on fire growth diminished.



Figure—2 Eight-hr FARSITE simulation for the 85th percentile weather and fuel condition, pre-treatment (a) and post-treatment (b). Each color represents a 1 hr progression of the fire overlaid with roads (black) and the treated landscape (light yellow). Black Ridge is in the foreground and the Pine Valley Mountains in the background (NW). Fuel modifications reduced the size of the fire by approximately 1,500 ac (18 percent).

Table 2—*FARSITE and FlamMap fire growth and behavior output for 75th, 85th, and 95th percentile weather and fuel moisture conditions.*

[1] Percentile		Size ac	Perimeter mi	Spot fires	Flame lgth ft	Intensity[2] BTU $ft^{-1}\cdot sec^{-1}$	Crown[3] ac
75th	Pre	5,880	18	326	2.6	72	12,883
	Post	5,297	18	228	2.43	68	9,242
	Pct change	-9.91	0.00	-30.06	-6.54	-5.56	-28.26
85th	Pre	8,588	28	434	13.96	1,885	27,600
	Post	7,056	22	301	10.77	1,262	22,093
	Pct change	-17.84	-21.43	-30.65	-22.85	-33.05	-19.95
95th	Pre	24,881	59	1,139	16.12	2,588	27,600
	Post	23,202	60	1,054	13.4	1,992	22,093
	Pct change	-6.75	1.32	-7.46	-16.87	-23.03	-19.95

[1]Number of spot fires initiated in the treatment area during a 6 hour period
[2]Mean flame length and intensity
[3]Passive and active crown fire

Although reductions in fire size are evident in all three percentiles, a decline in perimeter growth was only predicted in the 85th percentile. In the case of the 75th percentile, while the treatment reduced surface fuel, the effective wind speed was increased due to the removal of the pinyon-juniper, thus increasing the perimeter and rate-of-spread of the fire—this is also likely the case for the 95th percentile. In the 85th pre-treatment simulation, crown fire runs and spotting in fuel model “4s” and “6s” outpaced the increased effective wind speed.

Spot Fires

A reduction in new ignitions ahead of the main fire front is evident under all three weather conditions. This is due largely to the removal of the pinyon-juniper. It is worthy to remember that spotting in FARSITE is stochastic and the numbers of embers lofted and burning when they reach the ground are dependent on the spotting model (Albini 1979) and largely influenced by the ignition frequency and canopy characteristics. Thus, this information is imprecise and more emphasis should be given to the percent change, rather than the actual number of fires.

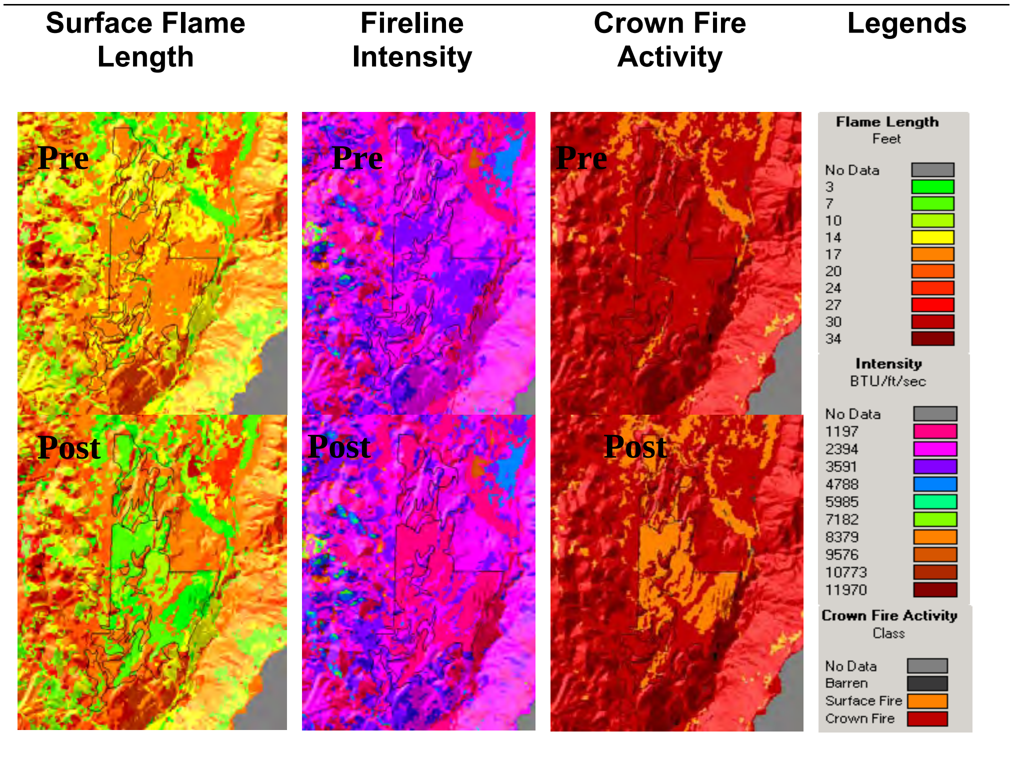


Figure 3—FlamMap output for the 85th percentile condition, pre-treatment (top) and post-treatment (bottom). The project area boundary is overlaid in black and runs north to south—about 4.5 miles.

Crown Fire

Although FlamMap differentiates between passive and active crown fire, *table 2* summarizes both types of crown fire as one. This was done due to the under prediction of active crown fire in FlamMap and FARSITE as compared to observed conditions (Cruz and others 2003; Fulé and others 2001; Scott and Reinhardt 2001). For example, in the 85th percentile condition, all crown fire was termed "passive;" in the 95th, only a slight amount (190 ac) had transitioned to an active crown fire, thus the identical values between pre- and post-treatment landscapes.

Discussion

Modeling Assumptions and Limitations

There are several assumptions and limitations to the methodology presented in this paper. FARSITE and FlamMap, as well as the models utilized by these modeling systems (e.g., surface fire spread, crown fire spread), operate under a broad range of assumptions and have specific limitations. Spatial data has resolution and accuracy limits inherit to mapping of heterogeneous surface and canopy fuels and terrain. Vegetation cross-walked to fuel model and fuel model assignments of treated landscapes are occasionally problematic and model output is largely a reflection of these "conversions." Moreover, RAWS information can be incorrect, unavailable, or influenced by local factors not known to the end-user. It is important that users understand model constraints, and more importantly utilize models and output within accepted bounds.

FARSITE or FlamMap?

FARSITE was used to simulate fire spread and spotting potential, although several other outputs are available, including fire line intensity, flame length, and crown fire activity. Instead, FlamMap was used to calculate these fire behavior outputs, for a number of reasons, including: 1) FlamMap calculations are near instantaneous where as FARSITE simulations can oftentimes take several hours; 2) FlamMap's primary design is to distinguish hazardous fuel and topographic conditions, making pre- and post-treatment comparisons and contrasts across landscapes much easier and more suitable than in FARSITE; 3) Although historical fire occurrence was used in this analysis, there is no guarantee future fires will occur in these areas. While a pattern is often evident, demographics, human activities, and climatic conditions can change. Therefore, selecting a specific fire start is often subjective—particularly with little or no ignition data—yet tremendously significant to the outcome of the simulation(s), thus not requiring this input (FlamMap) is advantageous; 4) Other parameters such as determining the distance to the treated area, developing the wind file, specifying the simulation duration, and setting fire behavior parameters, are largely at the discretion of the modeler and difficult to fully substantiate, whereas fewer parameters are required in FlamMap; 5) Many fires that often impact an area of concern, such as a community like Harmony Heights, start considerable distances away from the area they threat, so assessing an area with a single, localized run is limiting.

Modeling Discussion

A great deal of information can be obtained by modeling the effect of fuel treatments on fire growth and behavior and analyzing model outputs. Ideally, modeling will be done *before* the actual treatment is implemented so model findings can be incorporated to modify the treatment pattern, size, methods, etc. However, post analysis of fuel breaks, as in this case, can substantiate management decisions, yield useful findings for future projects, and identify weaknesses in treatment design and application.

At first glance, fuel modifications seem to have had little effect on the fire (*fig. 2*). In respect to fire growth, this is the case under certain weather conditions. Indeed, some modifications may have even *increased* the rate of spread, by exposing previously sheltered fuels. However, changes in other fire behavior characteristics are considerable, thus accomplishing the objectives of the treatment (*table 2*).

An area where modeling suggests additional landscape treatment may be beneficial is along the southeast corner of I-15. The large, southeastern most treatment polygon stands alone if a fire approaches from the south. This is due to private ownership directly to the north. Previous modeling indicates the most effective treatment design tends to be those that have fuel modifications in succession and distributed across the landscape (Finney 2001). Moreover, the *sooner* a fire encounters a fragmented fuel complex the greater will be the effectiveness of that treatment on disrupting or locally blocking fire growth. Therefore, a second phase of this project might consider additional polygons to the south, like those in succession to the west. By so doing, a fire spreading to the north would encounter several fuel breaks before reaching the public land to the north, thus reducing the forward fire spread rate and assisting firefighting efforts.

Finally, modeling allows for hypotheses testing. For example, "what is the 'breaking point' of the Ash Creek treatment when the weather and fuel conditions are such that treatment effectiveness is minimized in respect to fire growth?" Through multiple simulations with varying weather scenarios, this question can be theorized at the 88^{th} to 92^{nd} percentile.

Conclusion

Managers have a growing need to assess the effectiveness of landscape fuel treatments, however this need has outpaced the development of spatial models to accomplish the task. FARSITE, although not originally intended to do so, has been used to assess treatment effectiveness on fire growth and behavior (Stephens 1998; van Wagtendonk 1996). The methodology presented in this paper uses FARSITE, but also incorporates FlamMap, Fire Family Plus, and previous ignition history to assess fuel treatment effectiveness. Although the approach has limitations, model outputs yield useful information for planning, assessing, and prioritizing fuel treatments. In the future, planned enhancements to Flammap will enable users to evaluate landscape alterations on fire spread utilizing minimum travel time methods (Finney 2002) and eventually aid in optimizing treatment design to mitigate fire behavior and spread.

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The Effects of Fire on Serpentine Vegetation and Implications for Management[1]

Hugh D. Safford[2] and Susan Harrison[3]

Abstract

We summarize the results of two studies comparing fire effects on adjacent serpentine and non-serpentine soils in two vegetation types, grassland and chaparral. In both vegetation types, serpentine soils were less fertile and supported less biomass. Mean time-to-last-burn was longer, and fire severities lower, in serpentine vs. non-serpentine chaparral. Positive effects of fire on species richness, diversity, species turnover, and exotic invasion were most pronounced in the more productive non-serpentine vegetation. Management of unique, low productivity habitats like serpentine must take into account differing responses and lower resilience to fire that often characterize these environments.

Introduction

Ultramafic “serpentine” substrates underlie approximately 5,000 km² of the State of California. Although this amounts to less than 1.5 percent of California’s total area, fully 10 percent of the state’s endemic plant taxa are restricted to serpentine soils (Kruckeberg 1984). Two and a half percent of USDA-Forest Service managed lands in California are found on serpentine soils, and in the state’s northwestern corner, about 14 percent of the land area managed by the Klamath, Six Rivers, and Shasta-Trinity National Forests is underlain by serpentine substrates. Thirty percent of the rare plant species managed as “Sensitive” in these three forests are serpentine obligates, and a further 10 to 15 percent are primarily found on serpentine soils (J. Nelson and L. Hoover, pers. comm.). Further extensive occurrences of serpentine occur on the Mendocino, Plumas, Tahoe, and Los Padres National Forests. Because of the botanical and geological resources associated with ultramafic outcrops, a large number of Research Natural Areas (RNAs) and Special Interest Areas (SIAs) in California’s National Forests have been designated to protect serpentine habitats. The U.S. Bureau of Land Management and California State Parks also manage a number of conservation units composed primarily or entirely of serpentine vegetation.

Although the ecology of serpentine in California has been the subject of many dozens of scientific studies, the fire ecology of serpentine habitats has remained largely unexplored, and the role of fire in serpentine ecosystems is poorly understood. This fact is a serious impediment to management of serpentine habitats in California (McCarten and Rogers 1991, Six Rivers NF 1998). As McCarten and Rogers (1991) note, “predictions about the response of serpentine vegetation to fire must (presently) be based on studies in non-serpentine vegetation, but substantial differences between the two habitats may render such predictions false.”

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Differences in the ecology of serpentine and non-serpentine habitats are based fundamentally in the chemical idiosyncrasies of ultramafic rocks and the soils that develop on them. Ultramafic rocks are composed primarily of ferromagnesian minerals high in Mg- and Fe-oxides and heavy metals, and low in silica and oxides of Ca, K and P. Soils which develop on these substrates are thus critically low in most of the basic macronutrients essential to plant growth (Proctor and Woodell 1975). Compared to most other soil types, very low levels of soil fertility in serpentine soils lead to 1) low rates of plant growth and low levels of community productivity, 2) a very heterogeneous vegetation structure, often characterized by thin vegetative cover and large extents of bare ground, 3) high ratios of native to exotic species, and 4) a high number of stress-tolerating, edaphic-endemic taxa (Baker and others 1992). In addition, serpentine vegetation typically supports a much larger component of perennial herbs (grasses and forbs) than its non-serpentine counterparts.

Ecological theory predicts that the amount of resources available in an ecosystem should affect its response and sensitivity to disturbances like fire. Plant stature, biomass, rates of growth and fuel accumulation are all influenced by habitat productivity, and all of these factors play an important role in determining fire frequency and intensity (Pickett and White 1985, Bond and van Wilgen 1996). Site quality may also be related to rates of competitive displacement, with relatively unproductive (and therefore less competitive) environments less reliant on disturbance for diversity maintenance, on both spatial and temporal scales (Grime 1979, Huston 1994). Since the most significant direct effect of fire is to increase available amounts of space and light, the effects of fire in a given vegetation type should relate positively to productivity, since more productive plant communities are more strongly limited by above-ground competition (Tilman 1982).

Here, we summarize results of two independent studies carried out in California's northern Coast Range, comparing effects of an October, 1999 wildfire on serpentine and non-serpentine grassland and chaparral (Harrison and others 2003, Safford and Harrison 2004). Due to strong differences in soil fertility and community productivity between serpentine and non-serpentine habitats, our basic hypothesis in both studies was that the effects of fire (on, e.g., species richness, diversity, composition, exotic invasion) would be stronger on non-serpentine soils than the less productive serpentine soils. In the Discussion, we explore management implications of our results.

Methods

Study Site

Our study site was located at the junction of Napa, Lake, and Yolo Counties (38° 51' N, 123° 30' W), California, USA. 16,000 ha of this area were burned in the Sixteen Fire in mid-October, 1999. Field sampling was carried out on properties managed by the University of California (Donald and Sylvia McLaughlin University of California, Natural Reserve) and U.S. Bureau of Land Management. Elevations in the study area range from 370 to 945 m; climate is Mediterranean. Annual precipitation averages 725 mm, and mean annual temperature is approximately 18.1° (11.4° January, 27.8° July). Geology, flora and vegetation of the area are described in UCD-NRS (2000). Botanical nomenclature in this paper follows Hickman (1993).

Sampling Methods

In grassland, we took advantage of a series of long-term study plots which were partially burned by the Sixteen Fire (Harrison and others 2003). Species richness had been previously sampled at 80 sites in 1998 and 1999, using five 1 m^2 quadrats subsampled along a 40 m transect; 35 of these sites burned. After the fire, we added 10 burned serpentine sites, and 10 unburned sites (mostly non-serpentine) to more evenly balance the sampling design. Our final sample included 45 burned sites (18 serpentine, 27 non-serpentine) and 55 unburned sites (33 serpentine, 22 non-serpentine). We continued sampling for 2 yr after the fire (in 2000 and 2001). At each site we measured total, native, and exotic species richness and frequency; we collected soil samples, and (in 1998 and 2001) we clipped, dried and weighed aboveground biomass.

In chaparral we sampled from a design pairing burned sites with neighboring unburned controls (Safford and Harrison 2004). In spring and summer of 2000, 2001 and 2002, we sampled 40 pairs of burned and unburned sites, 20 in serpentine chaparral and 20 in non-serpentine chaparral. At each site we sampled one 250 m^2 macroplot, subsampling for percent cover of all species, numbers of shrub seedlings, and a suite of environmental variables. We estimated fire severity in burned quadrats by measuring carbonized stem termini on individuals of chamise (*Adenostoma fasciculatum*; in a few cases *Quercus durata* was substituted). We estimated time since last fire by counting growth rings from 3 to 5 fire-killed individuals of non-sprouting species of *Arctostaphylos, Ceanothus,* and *Cupressus.* In 2001 and 2002, we sampled woody and herbaceous biomass, and assessed regrowth of the vegetation by measuring the heights of the most common shrub species within each burned site. In some analyses, we used unburned sites as surrogates for "pre-fire" conditions; their paired burned sites represent "post-fire" conditions.

Analyses

In grassland, we used ANOVA and MANOVA to test the effects of year, soil, fire and their interactions on species richness and composition, and to compare bunchgrass frequency before and after fire on serpentine and non-serpentine sites. In chaparral, we used ANOVA/MANOVA to test for differences between soil types in 1) site and vegetation variables independent of fire; 2) fire severity and time since last fire; 3) post-fire changes in species richness, diversity and composition; and 4) rates of post-fire recovery in biomass, species richness, and seedling densities. For each soil type, we used regression to test the relationship between post-fire change in richness, diversity and composition and a suite of predictor variables, including fire severity, time since last fire, and variables related to productivity and soil fertility. Repeated-measures ANOVAs were used where appropriate (Proc Mixed; SAS 8.0).

Results

Grassland

Serpentine grasslands were significantly higher in soil Mg, and lower in soil N, P, and Ca than non-serpentine grasslands; biomass and cover were significantly higher in non-serpentine grasslands (Harrison 1999). Pre-fire, non-serpentine sites were less species rich on average than serpentine sites. Fire significantly increased the mean number of species in burned non-serpentine grassland, such that richness was comparable to that in serpentine sites, but this effect lasted only one year. On the average, non-serpentine richness dropped significantly between 2000 and 2001, but there was no significant difference in mean serpentine richness between 2000 and 2001 (*fig. 1*). The relative proportions of native and exotic species in serpentine sites showed no significant effect of fire. The mean number of exotics in non-serpentine

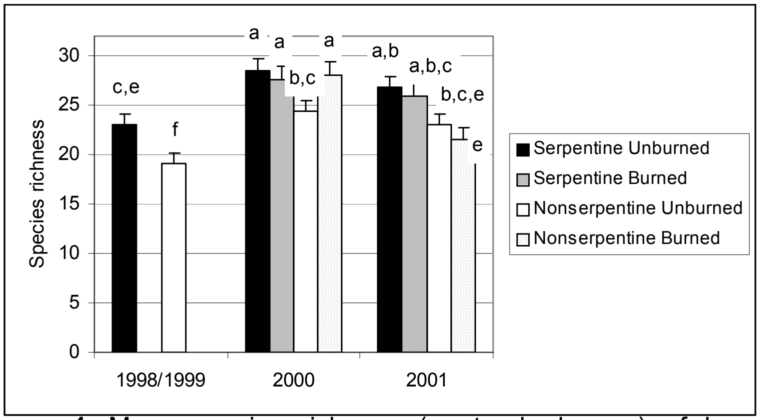

Figure 1—Mean species richness (+ standard error) of burned and unburned grassland sites, from 1998/1999 (two pre-fire years averaged) to 2001. Bars with different letters are significantly different at *P*<0.05.

sites increased significantly after fire (9.7 to 13.2; *P*<0.005), but there was no effect on mean native species richness. Neither species richness nor post-fire change in richness (i.e., post-fire minus pre-fire richness) were significantly related to biomass on either soil type. Species turnover between 1999 and 2000 did not differ significantly between burned and unburned sites in serpentine grassland (54 percent vs. 49 percent), but fire had a significant positive effect on turnover in non-serpentine grassland (63 percent burned vs. 56 percent unburned, *P*=0.01).

Perennial grasses occurred in 7 to 14 percent of our non-serpentine quadrats (sub-samples), depending on the year, but there were no significant differences across years or between burned and unburned sites. Frequency of perennial grasses was much higher in serpentine grassland (4 yr mean=49 percent), with a marginally significant increase occurring in burned sites after fire (59 percent in 2000 vs. 45 percent in 1999, *P*=0.10).

Chaparral

Serpentine chaparral was significantly higher in soil Mg and pH, and lower in soil P, K, and Ca than non-serpentine chaparral (Safford and Harrison 2004). Unburned (“pre-fire”) cover (81.4 percent vs. 69.6 percent) and post-fire biomass (woody species: 660 g m^{-2} vs. 84 g m^{-2}; herbaceous species: 81.5 g m^{-2} vs. 25.3 g m^{-2}) were both significantly greater in non-serpentine than serpentine chaparral (all *P*<0.001). Fire severity, measured as the mean diameter of carbonized stem termini, was significantly greater in non-serpentine than in serpentine chaparral (5.1 mm vs. 3.3 mm; *P*<0.005). Mean time-to-last-burn was much greater in serpentine chaparral (73.7 ± 39 yr vs. 18.6 ± 3.1; *P*<0.001).

Total species richness, diversity (measured as 1-Simpson’s Index), and exotic species richness were all significantly higher on burned than unburned sites in the year following fire (all *P*<0.001). The positive effects of fire on richness were significantly greater in non-serpentine sites (mean=15 species unburned vs. 35.8 burned) than in serpentine sites (mean=21.5 unburned vs. 31.5 burned (*fig. 2*). This same significant Fire X Soil interaction was found for species diversity and exotic richness as well (the latter increased from 1.2 to 6.9 species on non-serpentine soils, and from 0.9 to 4 on serpentine soils). Species turnover due to fire was also significantly greater in non-serpentine than in serpentine chaparral (77 percent turnover vs. 55 percent; P<0.001).

"Post-fire" change in diversity (i.e. burned minus unburned diversity) was positively related to "pre-fire" (unburned) cover on both soil types (serpentine: r^2 = 0.382, P=0.004; non-serpentine: r^2=0.149, P=0.09), while post-fire change in richness was positively related to soil total N in non-serpentine chaparral (r^2=0.224, P=0.04), and to the soil Ca/Mg ratio (a reliable predictor of productivity in vegetation of ultramafic soils) in serpentine chaparral (r^2=0.157, P=0.08). Mean time-to-last-fire was not significantly related to post-fire changes in richness or diversity on either soil type. Fire severity showed a negative relationship to postfire change in diversity in nonserpentine chaparral (r^2=0.397, P=0.003), and a unimodal relationship to change in diversity in serpentine chaparral (r^2=0.367, P=0.021).

In burned plots, we found 11 species on each soil that are either documented as, or assumed to be, fire-dependent species (i.e., whose germination is stimulated by heat, charate, and etc.; Keeley 1991). These species contributed 40 percent of post-fire herb cover in non-serpentine chaparral, but only 4.4 percent in serpentine chaparral. Obligately seeding shrub species (species which do not regenerate by sprouting) contributed more than twice as much pre-fire cover in serpentine chaparral as in non-serpentine (57 percent vs. 26 percent).

In the 3 yr after fire, species richness in burned non-serpentine sites dropped rapidly in concert with a strong recovery of vegetative cover (and biomass). During the same period, post-fire growth in serpentine chaparral was much slower, and the decrease in species richness much less pronounced (*fig. 2*). Shrub seedling densities in 2000 were significantly higher in burned non-serpentine chaparral than in burned serpentine (122.8 m^{-2} vs. 62.7 m^{-2} P<0.001). Seedling densities in burned sites dropped strongly in 2001 and 2002, but remained higher in non-serpentine than in serpentine chaparral (P<0.01).

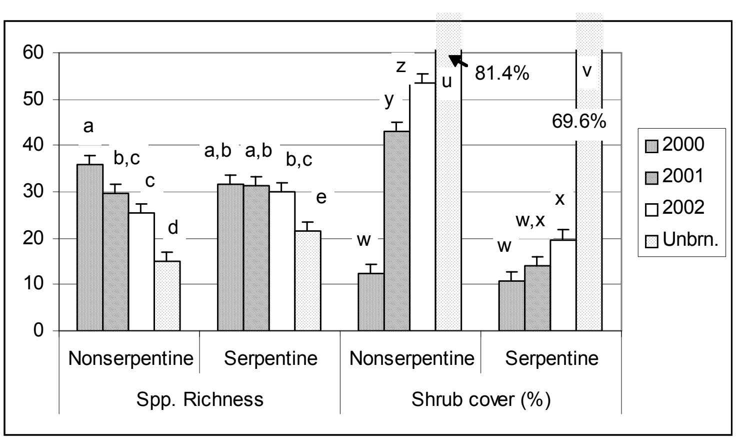

Figure 2—Means (+ standard error) of species richness and shrub cover in chaparral. Unbrn.=unburned controls in 2000. Bars with different letters are significantly different at P<0.05.

Discussion

The two studies summarized in this paper are among the first to explicitly treat the fire ecology of serpentine vegetation. In both studies, serpentine vegetation was much less productive than non-serpentine vegetation, and–at least in chaparral-fires were less severe and occurred less frequently on serpentine soils. Results of both studies showed strong effects of soil type on vegetation response to fire, but we were able to statistically link these effects to productivity only in chaparral. In general, the effects of fire appeared to be longer lasting in serpentine vegetation, with returns to pre-fire levels of richness and diversity, cover and biomass taking longer (in chaparral, much longer) than in similar vegetation types on non-serpentine soils.

In both chaparral and grassland, the richness of exotic species increased substantially after fire, and did so proportionately more on non-serpentine than on serpentine soils. These results are consistent with the prediction that invasibility should increase more in response to disturbance in productive than in unproductive environments (Grime 1979, Huston 1994). In grassland, although the effects of fire on species richness largely disappeared by the second year after fire, the number of exotics in non-serpentine grassland remained higher than before fire. At the same time, the post-fire increase in frequency of perennial grasses on serpentine soils was sustained for at least two years. Much evidence in the literature suggests that isolated fires, especially those occurring outside the growing season, have little long term impact on grassland ecosystems and are thus of little use in grassland restoration (Bond and van Wilgen 1996, Galley and Wilson 2001). However our results suggest that the thatch-removing effects of dormant-season fire can be at least ephemerally beneficial in low productivity grasslands already dominated by native perennials.

Lower fire severities and longer fire intervals in our serpentine chaparral sites are probably due primarily to the generally lower availability and continuity of woody fuels in these environments. However, dissimilarities in fire severity and time-to-last-fire were not the primary factors explaining differences in fire-induced changes in species diversity between the two soil types. Instead, it appears that the more open vegetation structure of serpentine chaparral allows for recruitment of opportunist/heliophilic herbs and shrubs in inter-fire years. Many of these same species occur in non-serpentine chaparral, but encounter opportunities to germinate and reproduce only after fire. In general, at the landscape-level, a disturbance-generated mosaic of early and late seral vegetation types seems less important to diversity maintenance in serpentine habitats than it is in most non-serpentine habitats.

Low fire frequencies in serpentine chaparral lead to the dominance of obligately seeding (non-sprouting) species in the shrub and tree layers (see also McCarten and Rogers 1991). These types of species require relatively long fire free intervals to reach reproductive maturity. Many rare "serpentine" species (and serotinous conifers like *Pinus attenuata* and *Cupressus spp*.) may be restricted to serpentine outcrops simply because of their intolerance to frequent and/or intense fire. Overall, it seems that although fire is inescapably an integral part of the ecology of any Mediterranean vegetation type, its importance to the functioning and persistence of low productivity habitats on substrates like serpentine is less pronounced than in habitats of higher productivity. Development of practical management strategies for vegetation of serpentine and other unusual soils demands that we take these fundamental ecological differences into account.

Acknowledgments

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Assessment of Emergency Fire Rehabilitation of Four Fires From the 2000 Fire Season on the Vale, Oregon, BLM District: Review of the Density Sampling Materials and Methods[1]

Jack D. Alexander III,[2] Jean Findley,[3] and Brenda K. Kury[2]

Introduction

In 2001 and 2002, Synergy Resource Solutions, Inc. (Synergy) sampled 58 study sites on four fires which burned during the 2000 fire season in the Vale, Oregon Bureau of Land Management District.

Methods

Synergy collected density data and photo-plot data to measure seeding success and compare treatments over time. Synergy selected 31 sites which were reseeded immediately following the fires with three native and four non-native seed mixes. Synergy collected photo-plot data on 11 sites which were burned but not seeded following the fires. This study was designed to note general success rates of seeding. Synergy evaluated treatments of Mulford's Milkvetch (*Astragalus mulfordiae*), a state-listed sensitive plant found in the Jackson fire, by sampling 16 paired plots (seeded vs. unseeded) to assess effects of seeding treatment on known populations.

Conclusion

Our poster examined what we learned about our monitoring techniques. Plot size and shape was critical to sampling efficiency for density studies. A moving plot rod was more efficient than a fixed plot frame.

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[1]A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2-5, 2002, San Diego, California.

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Growth of Regreen, Seeded for Erosion Control, in the Manter Fire Area, Southern Sierra Nevada[1]

Jan L. Beyers[2]

Introduction

The Manter Fire began July 22, 2000 in the Domeland Wilderness on the Sequoia National Forest, southern Sierra Nevada, California. It was declared controlled on September 6 after burning 32,074 ha (79,244 ac) of National Forest System, Bureau of Land Management (BLM), and private lands. The Burned Area Emergency Rehabilitation (BAER) team prescribed helicopter grass seeding along stream channels in high intensity burn areas to protect downstream values, which included golden trout habitat, willow flycatcher habitat, and the communities of Weldon and Onyx in the South Fork Kern River valley. Summer thunderstorms were expected to be the primary threat, and providing rapid revegetation near watercourses was hoped to prevent damaging sedimentation.

Because some of the proposed seeding would be in Wilderness areas, the BAER team chose to use Regreen, a sterile wheat-wheatgrass hybrid, in those watersheds to minimize impacts to native vegetation recovery while providing rapid ground cover. The Forest Service Pacific Southwest Research Station was asked to design a monitoring study because of the high cost of Regreen (\$435 ac^{-1} applied, compared to \$38 ac^{-1} for cereal grain), uncertainty about its effectiveness, and concern over its interactions with native plants.

Methods

This study was conducted in two drainages intersecting Long Valley Loop Road within the Chimney Peak Wilderness (BLM) at an elevation of 2,135 m (7,000 ft). Pre-fire vegetation consisted primarily of pinyon pine with an understory of sagebrush, rabbitbrush and herbaceous species. One drainage was seeded by helicopter with Regreen in December 2000, for approximately 45 m on each side of the channel, as part of the BAER implementation. The second drainage was omitted from aerial seeding. Ten pairs of plots were established in the unseeded drainage, and one of each pair was randomly chosen for hand seeding. Ten comparable plots were identified in the aerially seeded drainage.

Because of weather and logistical constraints, work on the study site did not begin until April 2001, just after snowmelt. A silt fence was constructed at base of the slope, just above the main channel, in each plot to measure erosion (fence about 5 m wide). Five 1 m^{-2} vegetation subplots were located alongside each silt fence contributing area to measure vegetation response. Regreen seed was applied with a hand spreader to achieve 215 seeds m^{-2} (20 seeds ft^{-2}, approximate aerial rate) in the

[1]A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2-5, 2002, San Diego, California.

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designated plots. Vegetation cover was estimated during mid-July in 2001 and 2002 by visual inspection of the 1 m^{-2} subplots.

Results

In April 2001, vegetation subplots in the aerially seeded drainage averaged 0.64 Regreen seedlings m^{-2}, with a range of 0 to 4.6 seedlings m^{-2} per erosion plot. Many of the seedlings appeared to be drying out, despite abundant soil moisture. The hand-seeded plots contained an average of 197 Regreen seeds m^{-2} (18.3 seeds ft^{-2}), with a range of 129 to 296 seeds m^{-2} (12.0 to 27.5 seeds ft^{-2}).

The study site received about 1.0 cm (0.4 in) of rain during May 2001. On July 6, 1.5 cm (0.6 in) of rain fell, followed by approximately 2.5 cm (1 in) in less than an hour at around 1:00 am on July 7 (weather data obtained from the Bear Peak RAWS and Long Valley GOES stations). The silt fences were overwhelmed with material and deep rills formed on slopes. No further rain fell during the summer.

In mid-July 2001, less than 1 Regreen plant m^{-2} was found in both the hand seeded and aerially seeded vegetation subplots. Regreen cover was very low (<0.1 percent). Total plant cover averaged less than 5 percent in all treatments, dominated by native species (table 1). In 2002, total plant cover was 20 to 25 percent. Regreen provided less than 0.5 percent cover, as plants present the previous year continued growth in 2002. Although Regreen plants formed inflorescences in 2001, no new Regreen plants were detected in 2002.

Table 1—*Dominant plant species (by percent cover) in vegetation plots.*

2001	2002
Gilia cana ssp. cana	*Lotus oblongifolius*
Gayophytum diffusum	*Sphaeralcea ambigua*
Lotus oblongifolius	*Gilia cana ssp. cana*
Phacelia spp.	*Lupinus adsurgens*

Discussion

Regreen established poorly after broadcast seeding (both aerial and hand), despite adequate seeding rates. The helicopter seeding was conducted just before snowfall, as recommended, and the hand seeding just after snowmelt, which should have been good times for successful establishment. The large seeds, similar to wheat, lying on the ground surface were undoubtedly attractive to birds and small mammals, and many may have been eaten. In April 2001, many seedlings from the aerial application were observed to have just one root barely penetrating the soil–most seedlings probably dried out before they could become established. Agricultural sources recommend drilling the seed, and, indeed, most successful plants from the 2001 hand seeding were growing in rebar stake holes or natural depressions. Many 2001 Regreen plants resumed growth in 2002–although considered annual, they can survive for several years. No new Regreen seedlings were observed (to be expected from a sterile hybrid); thus, although persistent, Regreen was not invasive. Seeded Regreen did not produce enough plants or ground cover to affect erosion or to negatively impact native vegetation. Because it produced so little cover, seeding with Regreen was not a cost-effective erosion control measure on this burn site.

Classification of Wildland Fire Effects in Silviculturally Treated vs. Untreated Forest Stands of New Mexico and Arizona[1]

Douglas S. Cram,[2] Terrell T. Baker,[2] Jon C. Boren,[2] and Carl Edminster[3]

Introduction

Heavy grazing and prolific conifer regeneration around the turn of the century coupled with 100 years of aggressive fire suppression and fire exclusion have combined to change forest structure, understory and overstory composition, and fuel biomass conditions in southwestern forests. Catastrophic stand-replacement fires, particularly in ponderosa pine forests (*Pinus ponderosa* Lawson), have displaced high frequency low intensity historical fire regimes. We hypothesized forest stands treated recently (<20 yr) using silvicultural practices were less likely to experience catastrophic fire compared to untreated stands.

Methods

We compared wildland fire damage in silviculturally treated vs. untreated forest stands in New Mexico and Arizona. Study sites ranged in elevation between 1,900 and 2,800 m. Silvicultural treatments included: lop, pile, burn; lop and scatter; harvest and burn; commercial thin; and shelterwood. Due to the unpredictability of how, when, and where wildland fire will burn, setting up elegant experimental designs pre-wildland fire is impractical. Nonetheless, we were fortunate to find suitable sites to replicate the first three treatments listed above, and a randomized complete block design was used for analysis of these treatments. We measured overstory and understory indices of fire damage and severity to determine stand conditions following wildland fire. It is noteworthy that three of the four fires studied for this project occurred under high winds and extended drought conditions.

Results

Preliminary results indicate fire severity in middle elevation (approximately 2,350 m) montane coniferous forests is allayed when the fuel leg of the fire behavior triangle is abridged. Under extreme conditions created by drought, high winds, and suitable topographical conditions, we observed treated forest stands that, although suffering less severe fire and ground char damage than adjacent untreated stands, were still subjected to near stand-replacement type damage. However, this illustrates

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[1]A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.
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that even under extreme conditions, fire severity can be mitigated by fuel reduction, and further that more aggressive treatments would likely have faired better. In particular, we observed prescribed fire in combination with mechanical thinning had the greatest impact toward mitigating fire severity.

Discussion

Silvicultural prescriptions designed to reduce stand susceptibility to catastrophic wildfire must consider slope and aspect, slash treatment, and residual tree and stand characteristics. Specifically, as density (stems ha^{-1}) and basal area (m^2 ha^{-1}) decrease and mean diameter at breast height (cm) increase, fire severity and ground char decrease. Further, a threshold in canopy bulk density (kg m^{-3}) on stands with 0 to 5 percent slope was identified beyond which initiation of a crown fire was possible and below which it did not occur.

The Potential for Smoke to Ventilate From Wildland Fires in the United States[1]

Sue A. Ferguson,[2] Steven McKay,[2] David Nagel,[3] Trent Piepho,[2] Miriam Rorig,[2] Casey Anderson,[2] and Jeanne Hoadley[2]

Introduction

To help assess values of air quality and visibility at risk from wildland fire, a spatial time series of ventilation potential for the United States was generated. The ventilation potential was determined as a product of model-generated surface winds and spatially interpreted mixing height observations. The surface winds (approximately 10 m agl) were generated from Danard's primitive-equation model (1977), using heights and temperatures at 850 hPa, 700 hPa, and 500 hPa from the NCEP Reanalysis as the upper boundary.

Methods

The mixing heights were calculated from radiosonde observations using Holzworth's parcel method (1972). In addition, we approximated the location of potential valley inversions with a GIS algorithm that considered terrain slope, curvature, and flow accumulation. Nights on which local inversions occurred were approximated by matching the hourly surface weather observations with Pasquill's stability criteria (1962) for representative neighborhoods.

Results and Discussion

The data represent a 40 yr time series, twice daily, at 2.5' latitude/longitude (about 5 km) spatial resolution. A map-based, data acquisition system is available on the World Wide Web (http://www.fs.fed.us/pnw/fera/vent) for use by land managers to help assess local, regional, and national ventilation potential. Periods of calm winds, low mixing heights, and resulting poor ventilation are seen in all areas of the country. The frequency and magnitude of ventilation potential, however, varies from place to place and time to time. The data available through this system is useful in planning for prescribed burning in order to minimize impacts from smoke. The data can also be applied during wildfires to assess the risk of impaired air quality at a specified location.

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[1]A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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Effects of Fire and Mowing on Expansion of Reestablished Black-Tailed Prairie Dog Colonies in Chihuahuan Desert Grassland[1]

Paulette L. Ford,[2] Mark C. Andersen,[3] Ed L. Fredrickson,[4] Joe, Truett,[5] and Gary W. Roemer[3]

Introduction

Black-tailed prairie dogs (*Cynomys ludovicianus*) once ranged from Canada to Mexico throughout the Great Plains and west to Arizona. During the last 100 years, public and private control programs, plague, and habitat loss have reduced the distribution of black-tailed prairie dog populations by 98 percent, causing localized extinctions. This species is now considered uncommon or extirpated in many areas of its former range. Black-tailed prairie dogs significantly alter grassland ecosystems and are considered a "keystone" species that require active conservation efforts (Kotliar and others 1999). Conservation measures for this species, including reintroduction, are underway in a number of areas.

The best practical indicator of habitat suitability for reintroduction is visible evidence of previous prairie dog occupancy (Jacquart and others 1986, Ackers 1992, Truett and Savage 1998, Truett and others 2001a). Due to vegetation changes following the absence of prairie dogs from a site, vegetation manipulation is often required to provide reintroduced prairie dogs with suitable habitat (Truett and others 2001a). Woody plants (McDonald 1993) or tall grasses (Osborn and Allan 1949) that encumber vigilance behaviors for predators may require remedial actions to remove shrubs or reduce vegetation height (Player and Urness 1982, Truett and Savage 1998, Truett and others 2001a). Prairie dog colonies will not expand into areas where ocular vigilance is hampered. Dense vegetation greater than about 15 cm tall must be reduced in height so newly established colonies can survive and expand. Where mid- and tall grasses prevail, prairie dogs thrive mainly where fire, bison or cattle shorten the grasses by burning, or grazing and trampling (Truett and others 2001a).

Prior to the late 1800s, bison (*Bison bison*) coexisted with and helped sustain a diverse assemblage of animal and plant communities in grasslands, including black-tailed prairie dog colonies and associated species (Truett and others 2001b). Bison grazing historically reduced grass stature around prairie dog colonies. However, the system under which prairie dog-bison coevolved is largely absent. Under current

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conditions the grazing intensity is insufficient to remove mature vegetation (*fig. 1*). Grass mowing to simulate the effects of grazing has been used to facilitate expansion of reintroduced prairie dog colonies, but large-scale mowing is generally not cost-effective. Another alternative to grazing and mowing is prescribed fire. Fire is a natural disturbance that regulates ecological processes in southwestern grasslands (Ford 2000). Fire reduces the height of vegetation and attracts foraging bison for at least the first few years post burn (Shaw and Carter 1990, Vinton and others 1993, Hartnett and others 1996, Knapp and others 1999, Truett and others 2001b). Use of fire best simulates the system in which these species evolved and may provide a more cost-effective method to promote the expansion and vigor of reintroduced black-tailed prairie dog colonies.

We report on the preliminary results of an experimental study evaluating fire vs. mowing for facilitating expansion of reintroduced prairie dog colonies in the northern Chihuahuan Desert. Our objectives were to 1) find a cost-efficient management tool for enhancing habitats for prairie dog reintroduction, and 2) to understand the use of fire as a tool for managing colony expansion of black-tailed prairie dogs in Chihuahuan Desert grasslands. Our long-term goal is to use fire as a catalyst to help sustain a long-term dynamic between bison and prairie dogs. There is a documented pattern of bison grazing following fire. If this occurs, bison can be expected to concentrate on the burned areas for 3 to 4 years following fire. Once prairie dogs expand into the burned areas, they can, with the help of bison grazing, maintain their habitat at low stature (less than 15 cm tall) by trimming and eating the vegetation.

Figure 1—Black-tailed prairie dog, bison, and Chihuahuan Desert grassland, Armendaris Ranch, southern New Mexico.

Study Area

Our study was conducted on the Armendaris Land Grant located at the northern extent of the Chihuahuan Desert in southern New Mexico. The Armendaris Land Grant was purchased by Turner Enterprises, Inc., with the primary objective of producing bison (introduced in 1995), and promoting wildlife biodiversity. Efforts to restore black-tailed prairie dogs to previously occupied habitat on the ranch commenced in 1995. The site is located on deep mixed alluvium formed at the base of alluvial fans and adjacent flood plain. Soils are characterized as a Mimbres silt loam. Average annual precipitation ranges between 200 to 250 mm (SCS 1984) with most precipitation resulting from convectional storms during the summer growing season. Vegetation is dominated by a *Sporobolus/Pleuraphis/Scleropogon* grassland association.

Methods

Three experimental sites were established at the margins of three re-established prairie dog colonies on the Armendaris Ranch. These colonies, S-Curve (2 ha), Red Lake (8 ha), and Deep Well (1.27 ha), were established in 1998-1999. We established six to eight 50 x 50 m experimental plots (20 total) at the periphery of each colony. Burn or mowing treatments were randomly assigned to each plot, with treatments being conducted in early July 2001 just prior to the summer growing season. All burrows in each colony were georeferenced using Trimble®XXX geographical positioning system immediately preceeding and four months after treatment with differentially corrected data mapped using ArcGis 3.2 software. Vegetation cover was sampled using five 20 m line intercept transects per plot before (June 2001) and after (October 2001) treatments were established.

Prior to establishing treatments prairie dogs were trapped at each colony. Each captured prairie dog was sexed and classified as juvenile or adult, morphometric measurements obtained, and examined for presence or absence of fleas. Further, each adult prairie dog was classified as breeding or non-breeding, based on the presence of descended testes in males and on evidence of lactation in females. Burrow data, vegetation data, and trapping data were analyzed using contingency table analyses, and Wilcoxon signed-rank and Chi-square tests. Because spatial coordinates were not recorded for prairie dog captures, these data could not be examined in relation to the treatments, but instead were analyzed with regard to possible differences in population structure among the three colonies.

Results

Numbers of newly established burrows in each treatment type within each of the three colonies were used as a measure of habitat suitability. Results indicate no significant difference (P=0.59) in the number of new burrows among treatments: burned (48 new burrows at Deep Well, 25 new burrows at Red Lake, 22 new burrows at S-Curve) vs. mowed (33 new burrows at Deep Well, 41 new burrows at Red Lake, 31 new burrows at S-Curve). However, there were strong differences among colonies in the total number of new burrows (χ^2=7.70, D.F.=2, P=0.021): Deep Well 81, Red Lake 66, S-Curve 53.

There were statistically significant post-treatment increases in bare ground, litter and prairie dog feces for both burned and mowed plots (overall χ^2=5990.6, D.F.=19, P<0.001). This pattern persisted when the three colonies were analyzed separately.

Table 1—*Number of trapped prairie dogs classified by sex and breeding status (adults only) for each colony.*

	Breeding		**Non-breeding**	
	females	**males**	**females**	**males**
Deep Well	15	11	4	3
Red Lake	5	7	5	2
S-Curve	16	8	4	10

Analysis of the number of trapped prairie dogs classified by sex and breeding status for each colony were not conclusive (χ^2=10.01, D.F.=7, P=0.188). However, there were slightly more active-breeding females than males, active breeding individuals outnumbered non-breeders by about two to one, and the overall sex ratio of trapped animals was slightly female-biased (*table 1*). There were differences in age and sex structure among the prairie dogs trapped at the three colonies (χ^2=18.49, D.F.=6, P=0.005) (*table 2*); but these differences may be an artifact of our trapping protocol. Although we trapped intensively, we were not able to capture all colony members.

Table 2—*Number of trapped prairie dogs classified by age (adult vs. juvenile) and sex, ignoring breeding status.*

	Females		Males	
	Adult	**Juvenile**	**Adult**	**Juvenile**
Deep Well	19	14	14	23
Red Lake	11	6	9	8
S-Curve	20	0	18	7

Conclusion

In the short-term, fire and mowing appear to be equally effective in promoting colony expansion by black-tailed prairie dogs in Chihuahuan Desert grasslands. Based on our knowledge of the grazing behavior of large generalist herbivores, fire treated areas appeared more favorable to bison grazing than mowed areas. Fire may catalyze a self-sustaining interaction between bison and prairie dogs. Preferential bison grazing on burned areas may help to maintain post-fire reduced grass stature longer than mowing, thereby providing the potential for further colony expansion. However, persistence of these treatments and interaction will depend on prairie dog population dynamics and impact of treatments on the foraging dynamics of associated large ungulates.

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Data Collection and Fire Modeling Determine Potential for the Use of Plateau®[1 2] to Establish Fuelbreaks in Cheatgrass-Dominated Rangelands[3]

Brenda K. Kury,[4] Jack D. Alexander III,[4] and Jennifer Vollmer[5]

Introduction

Plateau® herbicide was applied on recently-burned cheatgrass (*Bromus tectorum*) dominated rangelands near Boise, Idaho, to determine if the application of up to 12 oz ac^{-1} of Plateau was a viable method to create and maintain firebreaks. Plateau has been used extensively throughout the U.S. to enhance native plant seedings and control cheatgrass. Synergy Resource Solutions, Inc. gathered data to determine biomass production, litter accumulation, and plant height in the study area on June 15 and 16, 2002. Fire behavior at each site was modeled with collected data using BehavePlus fire-modeling program (USFS v. 1.0.0). Modeling predicted that application of Plateau at rates above 6 ounces/acre would effectively reduce Bromus tectorum in fuel break areas. Between Plateau treatments, flame height increased slightly at the 12 oz ac^{-1} rate due to an increase in the number of forbs (broadleaf) species compared to a greater percentage of grass species encouraged at the lower rates of 6 oz ac^{-1} and 8 oz ac^{-1}. These data indicated that fuel breaks treated with Plateau for cheatgrass control would have lower flame lengths and rates of spread than untreated areas.

Results

Results indicated that areas treated with Plateau had substantially less cheatgrass by weight than the untreated areas, regardless of treatment level. Control treatments had substantially more Sandberg bluegrass. For the evaluated rates, cheatgrass levels

[1]**Pesticide Precautionary Statement:** This publication reports research involving pesticides. It does not contain recommendations for their use, nor does it imply that the uses discussed here have been registered. All uses of pesticides must be registered by appropriate state or federal agencies, or both, before they can be recommended.
CAUTION: Pesticides can be injurious to humans, domestic animals, desirable plants, and fish or other wildlife—if they are not handled or applied properly. Use all pesticides selectively and carefully. Follow recommended practices for the disposal of surplus pesticides and pesticide containers.

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

[3]A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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were the same in the 6 oz ac^{-1} and 8 oz ac^{-1} treatments, indicating that Plateau at 6 oz ac^{-1} would be the most cost-effective application rate.

Plots treated with Plateau had a lower flame height and rate of spread than control treatment levels. Fuel modeling data indicated that areas treated with Plateau, regardless of the treatment level, would have a lower rate of spread and flame length than control treatments.

Discussion

These data indicated that Plateau can provide cheatgrass control in areas with high fire risk. Plateau could be used to create fuel breaks around important habitat resources (i.e., native sagebrush communities, Areas of Critical Environmental Concern) in fire-prone areas. Incorporating Plateau into management strategies could protect habitat for sage grouse and other species of concern. Potential Plateau application sites include areas along maintained roads and in or around maintained fire breaks. Plateau provides an opportunity to reduce the $542 million spent by federal agencies to control wildland fires in 2001, reduce danger to life and property, and reduce destruction of wildlife and plant habitats. Finally, Plateau herbicide can be incorporated into land management plans and reduces the risk of loss of life, structures, and vegetation in areas of concern by reducing fuel loads.

Debris Flow Occurrence in the Immediate Postfire and Interfire Periods and Associated Effects on Channel Aggradation in the Oregon Coast Range[1]

Christine L. May[2] and Danny C. Lee[2]

Introduction

The freshwater rearing environment plays an important role in the life history of anadromous fishes such as endangered socks of coho salmon (*Oncorhynchus kisutch)*. Disturbances such as severe wildfire can accelerate rates of landslide and debris flow activity due to a loss of root strength and vegetative cover. These large influxes of sediment can result in substantial aggradation of mainstem river channels that provide habitat for juvenile salmonids. Our study investigated the linkages between the timing of fires and debris flows, and the associated affects on channel aggradation and fish survival in the Oregon Coast Range.

Methods

We used dendrochronology to estimate the time since the previous debris flow and the last stand-replacement wildfire in a 5 km^2 area within an unlogged Douglas fir forest (May 2001).

Results

In the thirteen streams investigated, the time since the last debris flow ranged from 4 to 144 years. Over half of these streams experienced a debris flow within 30 years of the last stand-replacement wildfire. In addition to this synchronous pulse of debris flow activity, large storms sporadically trigged debris flows in the inter-fire time period, resulting in a substantial background rate of debris flow activity. Stand-replacement fires, which are characteristic of the Coast Range, occur very infrequently. Long-term estimates of the average recurrence interval range from 230 years (Long and others 1998) to 452 years (Impara 1997). Extrapolation of the observed background rate of 0.05 debris flows per year indicates that the number of debris flows in the post-fire time period is exceeded by the background rate of debris flow activity 180 yr into the inter-fire period (*fig. 1*). Because fires occur so infrequently in this region, these results suggest that fire is not the dominant mechanism for triggering mass wasting when viewed over long time scales. However, the synchronous pulse of debris flow activity immediately post-fire can

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result in substantial aggradation of mainstem river channels and have significant biological consequences.

Mainstem rivers in the sandstone lithology of the Oregon Coast Range tend to have a thin layer of highly mobile alluvium directly over the underlying bedrock. Channel aggradation downstream of debris flow inputs creates a deformable streambed of coarse gravel and the creation of deep pools. In two streams that we studied, such streambeds were extremely porous and resulted in a dry channel as streamflow went subsurface during the summer dry season. During the mid-summer period, fish were trapped in isolated pools and perished in those that later went dry. Severe crowding occurred in the few remnant pools that persisted throughout the summer. Channels that were not aggraded formed smaller pools that were limited by the depth to bedrock. These pools had a greater likelihood of remaining wet throughout the summer because they intercepted subsurface flow traveling parallel to the bedrock surface. Mortality of juvenile coho salmon due to desiccation in dry pools was 36 percent of the initial population (estimated by snorkel surveys).

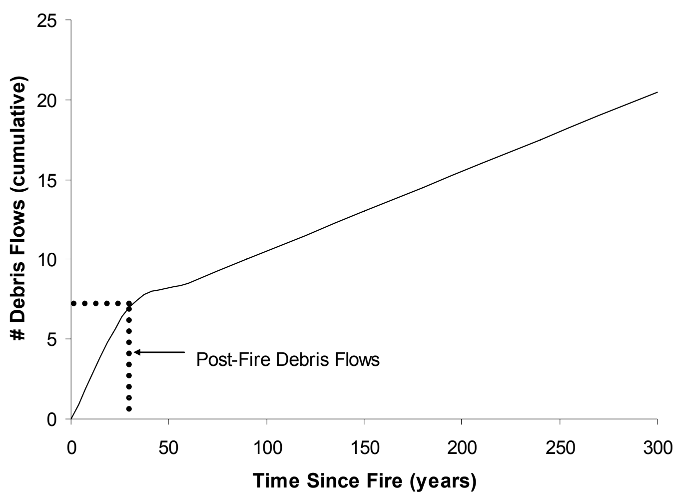

Figure 1–Debris flow activity in the immediate post-fire and inter-fire time periods.

Discussion

This study illustrates an important consequence of large influxes of sediment following severe wildfires. Because channel aggradation is directly linked to water availability and fish survival, increases in coarse sediment supply can be associated with sharp reductions in juvenile rearing habitat and productivity over the short-term. The long-term effects remain uncertain.

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LANDFIRE: Mapping Fire and Fuels Characteristics for the Conterminous United States[1]

James P. Menakis,[2] Robert E. Keane,[2] and Zhi-Liang Zhu[3]

Introduction

LANDFIRE is research and development project that will develop a comprehensive package of spatial data layers, models, and tools in support of the National Fire Plan, both at the national and local level. The project is being developed cooperatively between fire scientists at the USDA Forest Service, Rocky Mountain Research Station Fire Science Laboratory in Missoula, MT, remote sensing scientists at the USGS EROS Data Center in Sioux Falls, SD, and vegetation scientists at the RMRS Forest Inventory Analysis Laboratory in Ogden, UT.

Methods

LANDFIRE is a mid-scale project targeting map accuracies of 60 to 80 percent for the sub-watershed level (10,000 to 40,000 ac). The spatial datasets for LANDFIRE will be maintained at a 30 m pixel size. LANDFIRE is designed to be the safety net for land management agencies that do not have local-scale information, and is not a substitute for finer scale, local mapping efforts. It is intended to be scalable from sub-watersheds to a national level.

Discussion

Research and development have begun on 18,200,000 ha in two prototype areas: central Utah and western Montana. LANDFIRE Prototype is a three-year project starting in April 2002 with the prototype effort scheduled for completion in April 2005. Intermediate components/products will be available starting in summer 2002.

Conclusion

The purpose of this presentation was to introduce the audience to the LANDFIRE project. For more information about this project, please go to the LANDFIRE web page (www.landfire.gov). This web page is updated regularly with latest research information and project status. For more information about historical natural fire regimes and fire regime condition classes, please go to the following web page: www.fs.fed.us/fire/fuelman.

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Relationships Between Fire Frequency and Environmental Variables at Multiple Spatial Scales[1]

Carol Miller,[2] Brett Davis,[2] and Katharine Gray[3]

Introduction

Understanding where fires can be expected to occur and the factors that drive their occurrence is of great interest to fire managers and fire researchers. Information about the locations and perimeters of historical fires can provide insights to where fires tend to occur on the landscape. We used fire frequency maps derived from maps of fire perimeters dating back to the late 1800s for the Selway-Bitterroot Wilderness (SBW) in northern Idaho and western Montana (Rollins and others 2001). We investigated the relationship between fire frequency and environmental variables at multiple spatial scales to understand how fire-environment relationships vary within the SBW and the spatial scaling of these relationships.

Maps of fire perimeters can be digitized and analyzed in the context of a GIS. Although these digital fire atlases can be fraught with inconsistencies and inaccuracies resulting from fire reporting, they are often the best information we have about fire history for many areas. Previous research by Rollins and others (2002) used digital fire atlases for the SBW to relate fire frequency to biophysical variables. This data set covered the time period from 1880 to 1996, and included data on fires occurring in 63 different years, with a total cumulative area burned of approximately 500,000 ha. Results from that study indicated that fires occurred most often at low elevations and on dry, south-facing slopes in the SBW.

Methods

We investigated how fire-environment relationships vary within the SBW and the spatial scaling of these relationships. Building on the research by Rollins and others (2002), we examined, at five spatial scales, the relationship between fire frequency and environmental variables such as elevation, slope, aspect, potential vegetation type (PVT), temperature, and precipitation. We performed Chi-square goodness-of-fit tests to determine if fire was more common in certain topographic categories than would be expected if fire were randomly distributed among the topographic types (Manly and others 1993). We defined eight topographic categories by elevation (low and high), slope steepness (low and high) and aspect (north and south).

Results

Results from this analysis varied among three large geographic zones (269,000 to 408,000 ha) within the SBW. The most significant pattern was seen in the Montana zone,

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where fires occurred most frequently on steep north facing slopes at low elevations (<1,950 m). In the Northwest zone, fires occurred most frequently at low elevations (<1,650 m), regardless of slope steepness or aspect. The least dramatic pattern occurred in the West-Central zone, where fire tended to occur somewhat more often on steep south-facing slopes at elevations below 1,800 m.

Discussion

We offer three possible explanations for the different fire-environment relationships among the zones. First, the Montana zone, which extends from the Bitterroot crest eastward to the Bitterroot River, is drier than the other two zones because of the rain shadow effect from the Bitterroot Mountains. Fire spread and extent may have been limited by the continuity of surface fuels in this less productive zone. The relatively productive north-facing slopes may therefore have had a greater continuity of fine fuels and higher likelihood of experiencing fire. Second, differences in the range and variability of elevation among the zones may explain the differences in the fire-environment relationships we saw. Where the range of elevation was great enough to provide a gradient in the vegetation or physical conditions that drive fire occurrence, elevation emerged as the dominant predictor of fire frequency. This was the case in the Northwest zone. In contrast, much of the Montana zone comprises very high elevations (alpine and subalpine) where few fires have occurred during the period of record. As such, the range of elevations experiencing fire was much narrower. In the analysis, elevation was, in effect, more homogeneous in this zone and other factors, such as slope and aspect, emerged to explain fire occurrence. The third explanation involves differences among geographic zones in the spatial scaling of the environmental variables. The terrain in the Montana zone is much more highly dissected than in the other two geographic zones, with steep west-to-east oriented glacial valleys cutting across the Bitterroot Range, which runs from north to south. These topographic features may serve to confine fires to smaller portions of the landscape, inhibiting fire spread across more diverse and broad areas. Indeed, in any given fire year, the amount of area that burns in Montana zone is much less extensive than in the other zones, supporting the notion of topographically confined fires.

We were successful in using digital fire atlases to investigate relationships between fire frequency and the biophysical environment in the Selway-Bitterroot Wilderness. We used a multiple scale approach and found that these relationships vary within this very large landscape. As such, we suggest that for understanding landscape controls on fire regimes, smaller geographic zones (approximately 300,000 ha) make more appropriate study areas. Finally, our results underscore the importance of understanding the spatial scaling of the environmental variables influencing fire regimes.

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Effects of Summer Prescribed Fires on Taxa Richness and Abundance of Avian Populations and Associated Vegetation Changes[1]

Ken Mix,[2] William P. Kuvlesky, Jr.,[3] and D. Lynn Drawe[2]

Introduction

The plant communities of the South Texas Plains and Gulf Coastal Prairies and Marshes have been undergoing changes from grasslands to brushlands in the past century. The vegetative community once dominated by tall grasses has now become a chaparral mixed-grass community. Applying late season prescribed fires may slow shrub invasion in south Texas coastal prairies, allowing the landscape to revert to a more open grassland. Late season burns may also alter the current composition of the fauna inhabiting these landscapes. Increases in woody vegetation can reduce the abundance of grassland nesting bird species. Few studies of summer fires have documented the responses of avian communities in prairie and chaparral ecosystems. Reduction in brush as a nesting substrate may open the area for grassland nesting species and reduce the abundance of brush nesting species.

The study sites are on the Rob and Bessie Welder Wildlife Refuge located in a transition zone between the South Texas Plains and Gulf Coastal Prairies and Marshes in San Patricio County, TX, and on McCan unit of the McFadden Enterprise Ranch located in the South Texas Plains in Bee County, TX. Average yearly rainfall in the area exceeds 76.0 cm. The dominant woody vegetation is honey mesquite (*Prosopis glandulosa*) and huisache (*Acacia smallii*), and the herbaceous plant community is composed of mixed forbs and mid- to tall grasses. The study sites are approximately 1,000 ha, with each block having a control and treatment of approximately 400 ha. The Welder pastures are grazed on a high frequency, low intensity regime. The McCan pastures are grazed on a low frequency, high intensity regime. The experimental design is a repeated measures complete block with one replicate in each ranch.

Methods

Point counts were used to collect data on breeding bird species in each pasture with an unlimited detection radius. There were 15 points separated by at least 400 m. Counts began at sunrise and detection periods were partitioned into 2 minute, 3

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minute, and 2 minute intervals. Counts were conducted three times from May 1 until June 21, 2001 and 2002. Ten vegetation transects were selected from the 15 avian points. Vegetation data were categorized as woody shrubs, tress, forbs, and grasses. A 30 m line intercept was used to collect percent cover of woody and herbaceous vegetation. A vertical pole was used to collect visual obscurity data at the 20, 40, 60, 80, and 100 ft intervals and measured from a distance of 4 m. The data were collected in June and July, 2001 and 2002.

Results

ANOVA (analysis of variance) was used to analyze avian population and vegetation changes between pastures within a block. Changes in vegetation occurred in the treatment pastures of both blocks between 2001 and 2002 with significance set at $p=0.10$. In Block 1 visual obscurity (VO) changed significantly in the treatment pasture after the fire in these intervals: 0 to 1 m ($p<0.0001$), 1 to 2 m ($p=0.0669$), and total VO ($p<0.0001$). Percent woody cover was significantly different after the burn, reduced from 28.12 percent to 11.25 percent ($p=0.0249$). In Block 2 VO was significantly different in these intervals: 0-1 m ($p=0.0060$). 1 to 2 m ($p=0.0023$), 2 to 3 m ($p=0.0599$), and total VO ($p=0.0018$). Percent woody cover was significantly different after the burn, reduced from 45.55 percent to 16.27 percent ($p<0.0001$).

No changes in the nesting species have been linked to changes in nesting substrate. Most avian species showed little or no change in population in the treatment pastures from 2001 to 2002. The following species significantly changed in population in treatment pastures: Block 1, northern mockingbird (NOMO) ($p=0.0993$) and painted bunting (PABU) ($p=0.0292$); Block 2, Cassin's sparrow (CASP) ($p=0.0198$), northern cardinal (NOMO) ($p=0.0051$), olive sparrow (OLSP) ($p=0.0517$), and PABU ($p=0.0248$).

Discussion

Overall changes in avian populations were not significant. Though a few species did exhibit significant changes in population, no attribution can be made to changes brought about by summer fires, particularly since two brush nesting species exhibited positive changes in population in a treated pasture: PABU, and NOCA in Block 2. Changes in percent woody cover do not appear to affect the avian population. Though there were reductions in percent woody and visual obscurity caused by the fire, it may be necessary to reduce each of these below a certain threshold to effect changes in the avian populations.

Chamise (*Adenostoma fasciculatum*) Leaf Strategies[1]

Marcia G. Narog[2]

Introduction

Microhabitat and preceding environmental conditions may predispose chamise (*Adenostoma fasciculatum*) to a specific post-fire response. Improving predictions of post-fire response may require knowledge of the general phenological status of the chamise stand. Morphological plasticity of chamise leaves can be observed in varying degrees in seedling and resprouting individuals. Mature chamise have small, linear shaped leaves. Seedlings and initial post-fire sprouts produce complex multi-lobed leaves that are replaced with simple leaf structure over time. The occurrence of complex leaves may be instructive for developing knowledge of post-fire response. Controlled studies were designed to determine how environmental factors affect chamise leaf complexity and longevity. Determine if complex juvenile leaf morphology found on resprouting chamise after fire is affected by insolation level and water. Preliminary analysis of the effects of insolation level and hydration on chamise leaf complexity is presented.

Methods

Greenhouse grown chamise were treated with shade, water, nutrients, and pruning. After 6 months, new growth was evaluated for leaf complexity. Chamise leaves were sub-sampled from test plants to determine leaf morphology. Plants were divided into 4 equal above ground sections. Representative leaves in each third of the 4 sections were sampled and evaluated for complexity and measured (*fig. 1*). A total of 12 leaves were measured per plant. Chamise were described by the number and degree of complexity of leaves measured and by the position of the leaves along the stem. Youngest leaves were formed after the study was implemented and represent treatment effects.

Figure 1—Variability in chamise leaf morphology varies (a) among seedlings, (b) resprouting and mature plants.

Results

Chamise under shaded and well-hydrated conditions produced and retained more complex leaves. Plants with more light and less water began producing simple leaves sooner than shaded well-hydrated individuals. As time progressed after germination or following disturbance, production of leaf complexity decreased (*fig. 2*). Hot and dry

[1]A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2-5, 2002, San Diego, California.

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conditions reduced complex leaf retention and survival. Shade and hydration appear to promote greater leaf complexity in chamise.

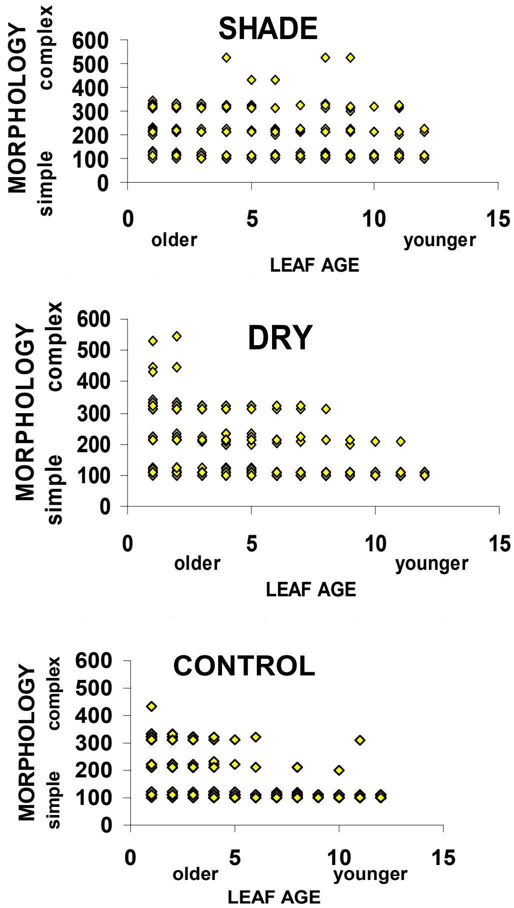

Figure 2—Leaf complexity decreased with time, yet was retained longer when a chamise was watered and shaded.

Discussion

Highly evolved complex leaves may promote growth or competitive advantage but appear to increase the plants sensitivity to environmental conditions. Evaluating leaf morphological characteristics and expected environmental conditions after a burn may be useful for determining post fire recovery or diversity in chamise or mixed chaparral.

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Stereo Photo Series for Quantifying Natural Fuels in the Americas[1]

Roger D. Ottmar,[2] Robert E. Vihnanek,[2] and Clinton S. Wright[2]

Introduction

Photo series are useful tools for quickly and inexpensively evaluating vegetation and fuel conditions in the field. The natural fuels photo series is a collection of data and photographs that collectively display a range of natural conditions and fuel loadings in a wide variety of ecosystem types throughout the Americas from central Alaska to central Brazil. Fire managers are the primary target audience of the natural fuels photo series, although the data presented will also prove useful for scientists and managers in other natural resource fields.

Methods

Phase I included 18 ecosystem types in the United States organized geographically into six volumes. Phases II and III will add volumes for ecosystem types in Hawaii (grassland, shrubland, woodland, and forest) and the northeastern United States (pitch pine, balsam fir/red spruce, and mixed hardwoods).

Discussion

Ongoing and future work will supplement already published volumes with new series in new ecosystems or additional sites in already published series. Additionally a volume has also been produced for savannah (*cerrado*) ecosystem types in central Brazil and a volume is under development for pine forests in Mexico. Ten ecosystem types have been photographed and inventoried to date with publication anticipated in the next two years.

[1]A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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Prescribed Fire Effects on California's Oaks[1]

Timothy E. Paysen[2]

Introduction

Urban encroachment, heavy use of public lands, and private land treatments create challenges for managing natural resources—in particular, California's oaks. Like many of the State's resources, the State's oaks are uniquely Californian. They give a Mediterranean flavor to the landscape that is unequalled in the country.

California oaks

California's oaks serve as havens for human recreation and use, and as buffers to disaster (*fig. 1*). They also host numerous wildlife species. Many oak species have become depleted, and some so heavily that their roles in the future of California's landscape are seriously questioned.

Figure 1—Post fire erosion problem.

Figure 2—U.S. Forest Service prescribed burning program.

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Fire, a common element in most California oak communities, can be a boon or a bane—depending upon the character of the fire and the condition of the ecosystem. A U.S. Forest Service program is underway with the objective of determining the appropriate use in the production of forest health and sustainability in California's oaks (*fig. 2*). A prime requirement is that the prescribed fire studies address effects on the complete ecosystem, as much as possible ("holistic" is the current buzz word). To do so is not a simple task. If fire can be used, then certain issues are important: what kind of fire, and when? Should the fire be "hot"? "cool? Winter? Summer? What will the effects on the entire community be?

International oak research

Interest in fire effects on California's oaks is not just limited to California. Similar kinds of oaks, growing under similar conditions, and which are subject to frequent fire, can be found in a number of countries around the world; countries with Mediterranean-type ecosystems (*fig. 3*). Thus, cooperation with scientists from various parts of the world can be beneficial—perhaps imperative.

Figure 3—Mediterranean-type ecosystems.

Study Implementation

The Prescribed fire program at the PSW Research Station, Riverside, CA is already underway. Documentation has been prepared, and a number of studies are in place (figs: 4-8).

Figure 4—Thin and understory burn to produce a shaded fuel break in a canyon live oak (*Quercus chrysolepis*) forest on the San Bernardino National Forest, San Bernardino, CA.

Figure 5—Prescribed burn of invasive grass in oak (*Quercus englemanii*) savannah on the Santa Rosa Plateau, The Nature Conservancy, Murrietta, CA.

Figure 6—Heavy fuel reduction prescribed burning in oak (*Quercus kellogii, Q. douglasii, Q. chrysolepis*) woodland at Camp Roberts,San Luis Obispo, CA.

Figure 7—Prescribed burn in mixed conifer oak forest for hazardous fuel reduction on the San Bernardino National Forest, San Bernardino, CA.

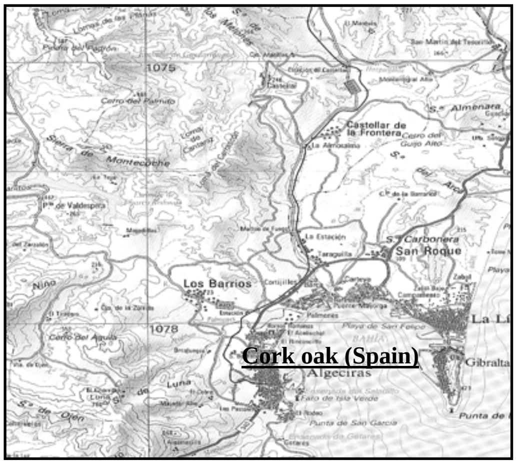

Figure 8—Understory burning in cork oak (*Quercus suber*) forests in Spain.

Discussion

Studies are in cooperation with other agencies, branches of the USDA Forest Service, other countries, and with a variety of professional disciplines. To properly carry out the necessary studies, more research disciplines must jointly direct their efforts toward the effects of prescribed fire, and the nature of factors that determine its effects—both short- and long-term. Such efforts must be deliberate, and coordinated.

Effect of Prescribed Fire on Recruitment of *Juniperus* and *Opuntia* in a Semiarid Grassland Watershed[1]

Burton K. Pendleton,[2] Rosemary L. Pendleton,[2] and Carleton S. White[3]

Introduction

The Bernalillo Watershed Protection Project was begun in 1953 following catastrophic erosion and flooding of small communities below. Although erosion control features and protection from grazing successfully increased grass cover and stabilized watershed soils, the expansion of juniper woodland (*Juniperus monosperma*) into the grassland watershed prompted concern that gains in watershed stability could be reversed. In 1995, fire was reintroduced into the grassland as a means of maintaining perennial grass cover and preventing further expansion of the juniper woodland community.

Methods

Burns were conducted on randomized 1 ha plots during November of 1995 and January of 1998. Juniper and *Opuntia* plants were censused in February of 2002. We counted all juniper plants occurring on treatment plots and measured height and crown diameter to the nearest decimeter.

Results

Burned plots had significantly fewer live juniper and significantly more dead juniper (*table 1*). Average size of living juniper on burned plots was greater, indicating a reduction in juniper recruitment.

Table 1—*Mean number and size of juniper occurring on burned and unburned plots.*

	Live juniper/plot	Dead juniper/ plot	Height (m)	Diameter (m)
Burned	7.25 a	5.0 a	1.70 a	2.14 a
Unburned	22.75 b	0.5 b	1.35 b	1.55 b
P value	<0.0005	<0.0005	0.0463	0.0445

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The three species of *Opuntia* found on the Bernalillo watershed–*O. phaeacantha, O. imbricata, and O. clavata*–were censused using belt transects. Patch area of *O. phaeacantha* and *O. clavata* patches was calculated using two perpendicular diameter measurements (cm). Both height and crown diameter were measured for cholla (*O. imbricata*). The mean number of cholla plants was significantly lower in burned plots, averaging 6 plants per plot as compared with 31 plants per plot in unburned areas. In addition, control plots averaged three patches of *O. clavata* per plot compared with zero in burned plots. The average number of *O. phaeacantha* patches was approximately equal for burned and control plots, averaging 131 and 138, respectively. However, patch size for *O. phaeacantha* was significantly reduced on burned plots (*fig. 1*). These data support the use of prescribed fire in reducing woody vegetation while maintaining cover of perennial grasses.

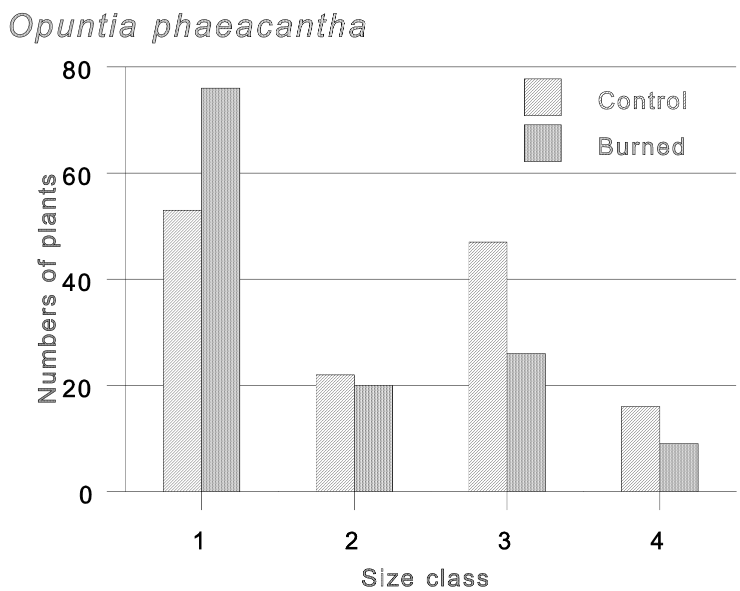

Figure 1—Size class distribution of *Opuntia phaeacantha* plants occurring on control and burned plots.

Monitoring Land Cover Change in California Using Multitemporal Remote Sensing Data[1]

John Rogan,[2] Doug Stow,[2] Janet Franklin,[2] Jennifer Miller,[2] Lisa Levien,[3] and Chris Fischer[4]

Introduction

Growing concern over the status of global and regional forest resources has led to the implementation of numerous multi-agency projects to establish long term operational systems for land cover monitoring. Land cover change (i.e., location, extent and cause) is identified as the most important and challenging research theme for many of the programs recently initiated by monitoring agencies. A key element in successfully addressing this theme is the involvement of regional and state-level management authorities to provide the necessary link between local/municipal and national/international land cover monitoring projects. Increasingly, these projects are using complex mapping procedures that require the integration of remotely sensed data, state-of-the-art image processing approaches, collateral spatial data, and georeferenced (GPS) field validation data within a Geographic Information System (GIS).

To address the challenge of forest and shrubland sustainability in the midst of rapid and widespread land cover change in California, the USDA Forest Service (USFS) and the California Department of Forestry and Fire Protection (CDF) are collaborating in the statewide Land Cover Mapping and Monitoring Program (LCMMP) to improve the quality and capability of monitoring data, and to minimize costs for statewide monitoring. Changes in forest, shrub, and grassland cover types are the primary focus in this program, but changes in urban/suburban areas are also mapped. Land cover change maps are required for regional interagency land management planning, fire and timber management, and species habitat assessment

An examination and comparison of the variety of remote sensing methods available, such as scene normalization, change feature extraction, classification, and accuracy assessment is warranted, in order to meet operational and standardization needs of the LCMMP. Faced with this task, the USFS and CDF staff associated with the LCMMP welcomed a research alliance with San Diego State University (SDSU) as a way to improve and automate change monitoring procedures. Specifically, techniques that minimize time-consuming human interpretation and maximize automated procedures for large area mapping of land cover change are being evaluated. The long-term objective of this study is statewide application of its *proof of concept* to the ongoing LCMMP.

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[1] A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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Data and Pre-Processing

Landsat TM and ETM images captures four to five years apart are geometrically registered to the UTM WGS84 projection with GCP points located at major road intersections, dispersed throughout the entire scene with less than 0.45 pixel RMSE. A nearest neighbor algorithm is used to resample the images to a 30 m output grid. These data are acquired within a three-month acquisition window (June-August) to provide operational flexibility (i.e., to minimize atmospheric contamination from cloud cover or wildfire smoke plumes), and to assure that they occur before the onset of foliage change in hardwood vegetation. These image data are then independently normalized for atmospherical illumination differences and converted to reflectance values using a dark object subtraction (DOS) approach.

The Landsat TM Multitemporal Kauth Thomas (MKT) linear transformation is used to spectrally enhance the data prior to supervised classification. MKT produces six features of interest; three features that represent change in brightness, ΔB (MKT1), change in greenness, ΔG (MKT2), and change in wetness, ΔW (MKT3), and three features that represent mean, or stable brightness (MKT4), stable greenness (MKT5) and stable wetness (MKT6) between dates. Further, ancillary data layers such as elevation, slope and aspect are included in the classification process.

Land Cover Change Classes

The land cover change classification scheme (*table 1*) describes three discrete categories of forest canopy cover decrease and two classes of canopy increase. Further, a shrub cover increase and shrub decrease class is used, along with change in developed (urban) areas and no-change (+15 to –15 percent canopy change) categories. The –15 to 15 percent change class was designed to reduce the confusion between phenological related increase and post-disturbance recovery classes. This classification scheme was developed and is currently in statewide use by the LCMMP. In situ reference data were collected for classification training and testing phases, based on a stratified random sampling scheme and was acquired using both aerial photographs and field visits.

Table 1—*Land cover change categories for the study.*

Land Cover Change Classes
+15 to –15 pct canopy change
-71 to –100 pct canopy change
-41 to –70 pct canopy change
-16 to 40 pct canopy change
Shrub/grass decrease >15 pct
+16 to +40 pct canopy change
+41 to 100 pct canopy change
Shrub/grass increase >15 pct
Change in existing developed areas

Change Mapping Approach

A univariate classification tree algorithm are used to produce tree-structured rules that recursively divide the data into increasingly homogenous subsets based on splitting criteria. At each split, the values of each explanatory variable are examined and the particular threshold value of a single variable that produces the largest reduction in a deviance measure (e.g., increase in subset homogeneity) is chosen to partition the data. Explanatory variables that have already been used in the model may be reexamined and potentially reintroduced into the tree structure. As a result, hierarchical, non-linear relationships within the data are revealed (*fig. 1*).

Spectral and ancillary variables are readily integrated and their contribution to map accuracy revealed in the hierarchical structure of the classification trees, and in the increase in accuracy when ancillary data were included in the classification. The methods used in this study were successful for mapping discrete categories of land cover change at overall accuracy levels of 72 to 92 percent. *Figure 2* shows a portion of a land cover change map of southern California from 1990 to 1996.

Conclusions

To address the concern about the amount and health of forest and shrubland ecosystems in California from accelerating land cover change, several agencies are collaborating in a land cover mapping and monitoring program. We monitored land cover change in San Diego County (1990 to 1996) using multi-temporal Landsat TM data. Change vectors of Kauth Thomas features were combined with stable Kauth Thomas features and a suite of ancillary variables within a decision tree classification. A combination of aerial photo interpretation and field measurements yielded training and validation data. Overall accuracies of the land cover change maps were high. The Kauth Thomas (brightness, greenness and wetness) and ancillary variables were important in mapping land cover change.

Predicting Patterns of Alien Plant Invasions in Areas of Fire Disturbance in Yosemite National Park[1]

Emma Underwood,[2] Robert Klinger,[2] and Peggy Moore[2]

Introduction

The location and size of Yosemite National Park means that it captures both a variety of interesting flora and fauna and also harbors many of the ecosystem processes that are characteristic to the Sierra Nevada. However, one of the increasing challenges confronting park management is the invasion of alien plant species. This is particularly problematic in areas which have experienced disturbances, such as fire or flooding (Rejmanek 1989, Mack and D'Antonio 1998). Such areas of disturbance provide ideal environments for alien species to establish; by removing the dominant species, and increasing bare ground, light and nutrients (Austin 1985).

Over the last few decades there has been an increase in both the number and mean size of fires occurring in Yosemite owing to a change in park policy which allows burning in designated areas. In order to assist park management in monitoring alien species in these susceptible areas, we conducted a community scale analysis of site collected data and also developed a landscape model to predict areas vulnerable to invasion based on their environmental envelope.

Materials and Methods

Our community level analyses assessed field data collected in 1998 and 1999 in the park (N=236). Field attributes included the identity and percent cover of alien and native plant species in each plot. This was supplemented with additional environmental data, such as slope, aspect, soil composition, in a Geographic Information System. A series of regression analyses found the best subset of these variables that explained first the presence and then the percent cover of the alien species and a TWINSPAN analysis also grouped co-occurring species to make modeling more efficient.

We then used a predictive model, the Genetic Algorithm for Rule-Set Prediction (Stockwell and Noble 1991), to extrapolate from the field collected data to the landscape scale. Models using the key environmental variables identified in the community analyses were run for each of the groups of alien species.

In order to target fieldwork to areas with the greatest probability of harboring alien species the results of from the predicted distribution were intersected with areas of greatest disturbance. We defined these as the largest and most recent wildfires that

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[1]A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

[2]Post-doc, graduate student, student, respectively. Graduate Group in Ecology, University of California, Davis, Davis, CA 95616. e-mail: eunderwoodrussell@ucdavis.edu.

had occurred in the park. We then generated a set of random points to help guide fieldwork for monitoring invasives.

Results and Discussion

The community level analyses identified elevation, slope, percent tree cover and percent shrub cover as the factors best able to explain the distribution of alien species in the site collected data. Using these key variables, the GARP model was run using a selection of site collected locations to train the model, and the remaining sites were reserved to test the accuracy of the results. The overall accuracy of the model, that is, its ability to successfully predict the invisibility of sites was 76 percent.

Our selection of burn areas with the greatest disturbance included the Ackerson, A-rock, Hoover, Leconte, and Steamboat wildfires, which have all occurred since 1990. A total of 200 random points were generated for field work in areas predicted to harbor alien species which fell within these wildfire boundaries. The number of points assigned to each burn was determined by the size of the burn, a minimum distance from the boundary of the burn was specified to avoid spurious edge effects.

Conclusions

These results provide a foundation for sampling and monitoring alien species within areas of disturbance in Yosemite National Park, and also a means to allocate limited park personnel and financial resources. Fieldwork conducted across a continuum of areas with different accuracy results will also yield important information on the way alien species respond to fire. In the long-term such analyses can be applied to other disturbances in the park, such as areas of flooding or disturbances in the park caused by human visitation, to provide a holistic monitoring plan for alien species.

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Fuel Consumption During Prescribed Fires in Big Sage Ecosystems[1]

Clinton S. Wright[2] and Roger D. Ottmar[2]

Introduction

Fuel consumption was evaluated for a series of operational prescribed fires in big sage (*Artemisia tridentata*) ecosystems throughout the interior West. Pre-burn fuel conditions (composition, loading, arrangement, and moisture content) and day-of-burn environmental conditions (temperature, relative humidity, wind, and time-since-rain) dictate fire behavior and subsequent fuel consumption. The amount of fuel available to burn, which is influenced by season of burn, site productivity, time-since-last-fire, grazing, and environmental conditions, is the most important factor controlling consumption. Fuel consumption can be manipulated to some degree, and poor burning conditions can be mitigated in some cases, by adjustments to firing techniques.

Methods

Relationships gleaned from these fuel consumption data will be incorporated into a predictive model and built into the software program CONSUME 3.0 (currently under development), a fire management and planning tool that predicts fuel consumption and emissions during prescribed and wildland fires.

Discussion

Future research will seek to refine, test, and expand on previous observations. New experiments will be designed to address questions about the effects of small changes in environmental conditions and season of burn on fuel consumption. Future work will also evaluate the effects of prescribed fire in shrub-type ecosystems composed of different species.

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[1]A poster version of this paper was presented at the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, December 2–5, 2002, San Diego, California.

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